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REVIEW

# Canadian boreal forests and climate change mitigation[1]

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**Abstract:** Quantitative assessment of Canada's boreal forest mitigation potential is not yet possible, though the range of mitigation activities is known, requirements for sound analyses of options are increasingly understood, and there is emerging recognition that biogeophysical effects need greater attention. Use of a systems perspective highlights trade-offs between activities aimed at increasing carbon storage in the ecosystem, increasing carbon storage in harvested wood products (HWPs), or increasing the substitution benefits of using wood in place of fossil fuels or more emissions-intensive products. A systems perspective also suggests that erroneous conclusions about mitigation potential could result if analyses assume that HWP carbon is emitted at harvest, or bioenergy is carbon neutral. The greatest short-run boreal mitigation benefit generally would be achieved by avoiding greenhouse gas emissions; but over the longer run, there could be significant potential in activities that increase carbon removals. Mitigation activities could maximize landscape carbon uptake or maximize landscape carbon density, but not both simultaneously. The difference between the two is the rate at which HWPs are produced to meet society's demands, and mitigation activities could seek to delay or reduce HWP emissions and increase substitution benefits. Use of forest biomass for bioenergy could also contribute though the point in time at which this produces a net mitigation benefit relative to a fossil fuel alternative will be situation-specific. Key knowledge gaps exist in understanding boreal mitigation strategies that are robust to climate change and how mitigation could be integrated with adaptation to climate change.

*Key words:* boreal forest, Canada, carbon, climate change, mitigation.

**Résumé :** L'évaluation quantitative du potentiel d'atténuation du réchauffement climatique des forêts boréales canadiennes n'est pas encore possible bien que l'on connaisse l'amplitude des activités d'atténuation, que les besoins en analysesvalables des options soient de mieux en mieux compris et qu'on observe une reconnaissance émergente que les effets biogéographiques nécessitent une plus grande attention. Une approche par système souligne les avantages respectifs des activités cherchant à augmenter le stockage du carbone dans l'écosystème, augmenter le stockage du carbone dans les produits ligneux récoltés (PLRs) ou augmenter les avantages liés à substituer le bois aux combustibles fossiles ou aux produits à plus fortes émissions. Une approche par système suggère également que des conclusions erronées quant au potentiel d'atténuation pourraient apparaître si les analystes assument que le carbone PLR est émis à la récolte ou que la bioénergie est neutre en carbone. Le meilleur avantage à court terme de l'atténuation boréale serait généralement atteint en évitant les émissions de gaz à effet serre, mais à long terme il pourrait y avoir un potentiel marqué dans les activités augmentant la suppression du carbone. Les activités d'atténuation pourraient maximiser l'absorption du carbone du paysage ou maximiser la densité en carbone du paysage, mais non pas les deux simultanément. La différence entre les deux se trouve dans le taux avec lequel les PLRs sont produits pour répondre aux besoins de la société, et les activités d'atténuation pourraient chercher à retarder ou réduire les émissions des PRL et à augmenter les avantages de la substitution. L'utilisation de la biomasse pour produire de l'énergie pourrait aussi y contribuer, bien que le moment auquel cela s'avérerait un avantage net en matière d'atténuation par rapport à l'utilisation de combustible fossile serait particulier à une situation donnée. Il existe un manque de connaissances pour comprendre quelles stratégies d'atténuation boréales robustes adopter par rapport au changement climatique et savoir comment on pourrait intégrer l'atténuation à l'adaptation au changement climatique. [Traduit par la Rédaction]

*Mots-clés :* forêt boréale, Canada, carbone, changement climatique, atténuation.



## 1. Introduction

Global emissions of carbon dioxide ($CO_2$) averaged 32.3 ± 3.2 Gt $CO_2$/year in 2000–2009 as a result of fossil fuel combustion, cement production, and land-use change (Friedlingstein et al. 2010; Global Carbon Project 2010). They reached 36.7 ± 3.3 Gt $CO_2$/year in 2010 after a record annual increase of 5.9% (Peters et al. 2012). Emissions are estimated to have caused atmospheric $CO_2$ concentrations to reach 390 ppm in 2010 (Peters et al. 2012); in comparison, these concentrations were about 280 ppm in 1750 at the start of the industrial revolution (IPCC 2007*a*). Atmospheric concentrations of other greenhouse gases (GHGs) have also increased (IPCC

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2007*a*). These increased concentrations, in combination with other human influences on climate-forcing factors (i.e., factors that affect the energy balance of the climate system, such as increased atmospheric concentrations of aerosols and changes in surface albedo), are contributing to observed changes in global climate (IPCC 2007*a*). Human societies will need to adapt to the changes as they occur (IPCC 2007*b*), but mitigation efforts that limit future growth in net GHG emissions (i.e., the sum of gross emissions of GHGs to the atmosphere plus removals of $CO_2$ from the atmosphere) will lessen the climate changes and reduce the adaptation required (IPCC 2007*c*). Mitigation efforts involving land systems can contribute both by reducing emissions and increasing $CO_2$ removals.

Forests have a significant role in the global carbon (C) cycle and Canada's boreal forests are an important part of that (Kurz et al. 2013). Globally, forests contribute a sink that averaged 9.3 ± 3.8 Gt $CO_2$/year between 1990 and 2010 and removed an estimated 30% of anthropogenic $CO_2$ emissions from the atmosphere (these estimates are based on the data collated by Global Carbon Project 2011; see also Pan et al. 2011). The anthropogenic emissions include net emissions from land-use change averaging 4.7 ± 2.6 Gt $CO_2$/year in 1990–2010 and gross emissions of 10.8 ± 1.7 Gt $CO_2$/year in 1990–2007 (Global Carbon Project 2011; Pan et al. 2011). Nabuurs et al. (2007) estimated that there is substantial potential to alter these large fluxes through forest-related mitigation activities around the globe, similar in magnitude to the mitigation potential of major sectors such as industry and transportation (Barker et al. 2007). Moreover, the technical and scientific knowledge needed to implement mitigation activities in the forest sector largely exists today, unlike the situation for many mitigation activities in other sectors (IPCC 2007*c*).

An uncertain portion of the mitigation potential could be realized in the forests of Canada's boreal zone. In this review, we first emphasize the importance of a sound analytical framework for mitigation assessment. We then synthesize the current understanding of boreal forest mitigation potential and highlight key knowledge gaps. We examine the mitigation potential of land-use change activities (afforestation, reduced deforestation) and forest management in the boreal zone, and through the use of forest biomass for harvested wood products (HWPs) including bioenergy. We also examine mitigation involving forested boreal peatlands. To date, mitigation research has focused predominantly on C and GHGs (IPCC 2007*c*) in keeping with the focus of the United Nations Framework Convention on Climate Change (UNFCCC; UNFCCC 1992) and, reflecting the state of current understanding, this review has the same focus. However, there is increasing understanding that mitigation involving land systems can affect other climate-forcing factors (e.g., Jackson et al. 2008; Anderson et al. 2011; O'Halloran et al. 2012), and we also discuss these impacts. In this paper, the boreal zone is the area defined by Brandt et al. (2013), covering 552.0 Mha, of which 270.4 Mha are defined as forest. We note for the reader instances in which information is only available by terrestrial ecozones (Ecological Stratification Working Group 1995); in such instances, the boreal zone is defined as the sum of the following ecozones: Boreal Cordillera, Boreal Plains, Boreal Shield, Hudson Plains, Taiga Cordillera, Taiga Plains, and Taiga Shield. These ecozones cover 581.9 Mha (Brandt 2009).

A large, global body of literature on climate change mitigation involving forests has developed during the past three decades, identifying possible mitigation activities, describing and addressing analytical issues, and developing estimates. Much of this literature provides insights on mitigation as relevant to the forests of Canada's boreal zone as to other forests, although the capacity of boreal forests to contribute to mitigation depends on specific ecosystem and management characteristics such as relatively low productivity, significant natural disturbance regimes, and extensive as opposed to intensive management. A growing number of site- and landscape-specific studies address forest-related mitigation in the boreal zone, but an integrated assessment has not been conducted to date. For this review, studies of forest-related mitigation in other regions and countries were assessed for their applicability to the boreal forests of Canada.

Mitigation is a global challenge because GHG emissions anywhere in the world affect global climate. It was the need for coordinated global action that led to the creation of the UNFCCC, which has an objective of "stabilization of GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" (UNFCCC 1992). The convention has been ratified by 195 countries. Working toward its objective, the UNFCCC encouraged developed countries to return GHG emissions to the 1990 level, and the 1997 Kyoto Protocol to the UNFCCC required developed country signatories to collectively limit their GHG emissions to 5% below the 1990 level in 2008–2012 (Kyoto Protocol 1998). Countries around the world have now established GHG emission reduction targets, policies, and measures for 2020 and beyond (UNFCCC 2011*b*, 2011*c*); and international negotiations are ongoing to coordinate action (e.g., UNFCCC 2011*d*, 2012*a*, 2012*b*). Countries have agreed that deep global emission reductions are required to hold the increase in global average temperature to less than 2 °C above pre-industrial levels, and they are seeking agreement on a global emission reduction goal for 2050 (UNFCCC 2011*d*, 2012*b*). Long-term reduction goals and trajectories have been suggested both in the scientific literature (e.g., IPCC 2007*d*; Allen et al. 2009; Meinshausen et al. 2009) and in international political discussions (e.g., G8 countries have supported target emission reductions by 2050 of at least 50% globally and 80% for developed countries (G8 2011)). However, the challenge of limiting the global average temperature increase to less than 2 °C has been recognized (PwC 2012; Peters et al. 2013).

Different time frames for emission reductions have been proposed on the basis of differing assessments of what constitutes dangerous anthropogenic interference with the climate system, the desired atmospheric stabilization level, the time at which emissions would need to peak, cumulative emissions, and the speed of emission reductions (e.g., IPCC 2007*d*; Meinshausen et al. 2009). Moreover, uncertainty exists about the likelihood, direction, and magnitude of feedback between climate change and the C cycle (Friedlingstein et al. 2006; Denman et al. 2007; Friedlingstein and Prentice 2010; Raupach and Canadell 2010; MacDougall et al. 2012; Le Page et al. 2013). At the global level, modelling studies and available evidence support a positive feedback effect in which warming as a result of GHG emissions has the net effect of increasing terrestrial and ocean GHG emissions (Friedlingstein et al. 2006; Friedlingstein and Prentice 2010). This adds to the uncertainty about the level of mitigation effort required to achieve the goal of the UNFCCC, but it also suggests that the earlier that mitigation occurs the less could be positive feedback effects. See Kurz et al. (2013) for a discussion of feedback effects in relation to Canada's boreal forest C.

In Canada, the federal, provincial, and territorial governments and others have established emission reduction targets (e.g., Environment Canada 2010; UNFCCC 2011*b*). Canada's announced target for 2020 is that its emissions will be 17% below the 2005 level, or 607 Mt $CO_2e$ ($CO_2e$, $CO_2$ equivalents) (UNFCCC 2011*b*; Environment Canada 2012). In 2011, Canada's anthropogenic emissions were 701.8 Mt $CO_2e$, not including the emissions and removals from forests, agricultural lands, and land-use change (Environment Canada 2013). No corresponding estimate of emissions for the boreal zone exists. However, information for emissions and removals on managed lands is available for ecozones (Environment Canada 2013) and forest-related emissions and removals in the boreal zone are described in detail in Kurz et al. (2013).

Mitigation involving forests of the Canadian boreal zone is one response to climate change. A necessary and complementary response is adaptation to the impacts of climate change (Gauthier et al., Manuscript in preparation). The impacts on boreal forest ecosystems are predicted to be significant, with consequences for human use of the forests and the values that society obtains from them (Price et al. 2013). Among those impacts will be changes in natural disturbance regimes and forest growth that will have long-term consequences for C stocks (Kurz et al. 2008*b*, 2013; Metsaranta et al. 2010), and hence for mitigation. To date, very few studies have considered how a changing climate could affect the mitigation potential of forests. There are also few published studies that examine the synergies, conflicts and linkages that may exist between mitigation and adaptation (e.g., Nabuurs et al. 2007; D'Amato et al. 2011), and none in the context of the forests of Canada's boreal zone. For example, assisted migration has been proposed as an adaptation response that could help to maintain the productivity and health of Canada's forests: it might also contribute to mitigation depending on the implications for forest C (Winder et al. 2011).

## 2. Framework for mitigation assessment

### 2.1. Defining mitigation

The Intergovernmental Panel on Climate Change (IPCC) defines mitigation as the implementation of policies to reduce GHG emissions and increase sinks (IPCC 2007*c*). This is consistent with the usage of the term in ongoing international discussions under the UNFCCC aimed at limiting global GHG emissions to reduce climate change. Halsnæs et al. (2007) defined mitigation potential as the amount of reduction in GHG emissions or increase in removals that can be achieved by a mitigation activity relative to a baseline or reference case in a given time period at a given cost per tonne. These definitions do not take into account how land-based mitigation activities can affect the biogeophysical properties of land, such as albedo, that also influence climate. Section 2.4 describes the emerging understanding that this is an important consideration in determining the mitigation potential of boreal forests. Assessment of mitigation potential typically is prospective: it is meant to provide guidance on what activities could be taken to reduce emissions or increase removals. In conceptual terms, a baseline is a projection of the emissions or removals that would occur in a business-as-usual world (i.e., in the absence of mitigation action). The business-as-usual conditions and corresponding baseline emissions and removals will change through time as environmental, technological, economic, and policy conditions evolve for reasons unrelated to mitigation, and these changes can be hard to predict. For example, activity aimed at reducing emissions caused by fires may have a real impact, but it can be very difficult to establish the baseline fire-related emissions that would occur in the absence of mitigation activity (Hurteau et al. 2008; Hurteau and North 2009). Thus, careful determination of baselines is necessary to accurately identify the potential emission reductions or removals that mitigation efforts may produce (Barker et al. 2007; Halsnæs et al. 2007) and avoid attributing existing forest sinks or business-as-usual emission reductions and removals to mitigation efforts.

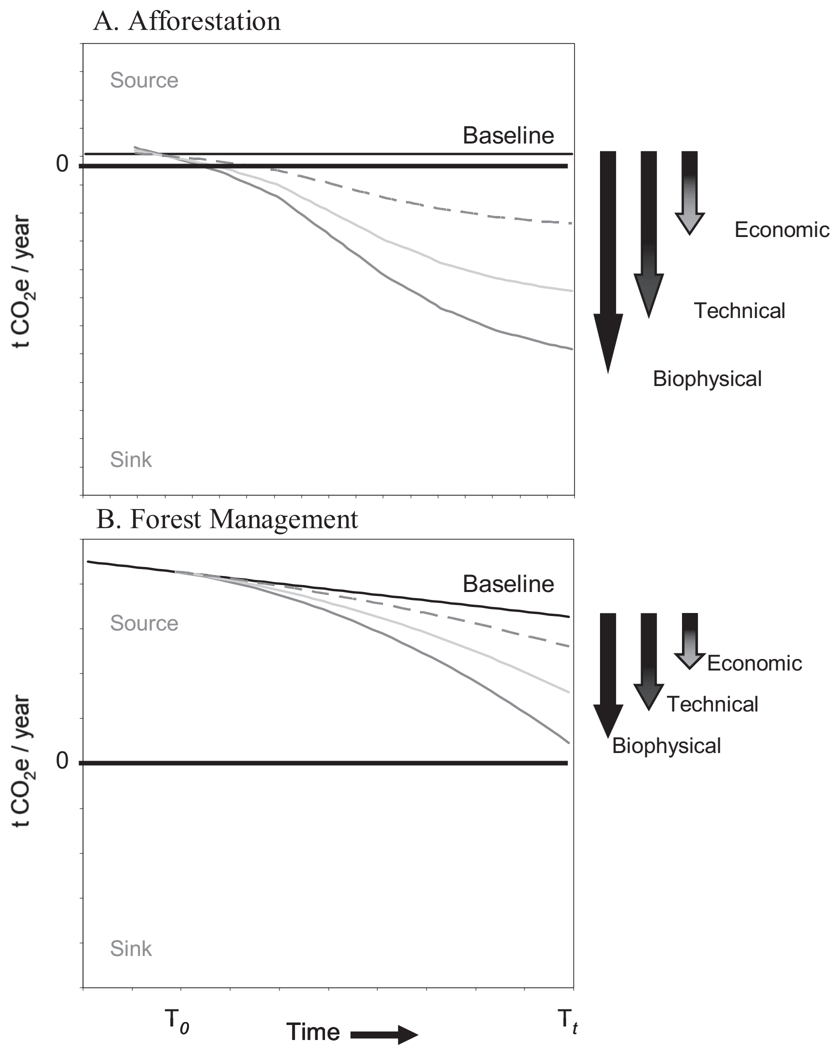
Mitigation assessment is concerned with the results of direct human action. However, analysis in biological systems is complicated by the existence of significant natural influences and indirect effects of human activities on emissions and removals with strong temporal and spatial dynamics, especially in the case of managed forests (Canadell et al. 2007; Böttcher et al. 2008). Natural factors include natural disturbance regimes and climate variability. Indirect effects of human activities can include fertilization owing to elevated $CO_2$ concentrations and nitrogen deposition (Hyvönen et al. 2007; Gedalof and Berg 2010; Girardin et al. 2011) and changes in forest productivity and natural disturbance regimes induced by anthropogenic climate change (Price et al. 2013). These factors can have long-lasting impacts on forest emissions and removals that will be greater than the impacts of management activities (Kurz et al. 2013). As well, past management activities and past natural disturbances have long-lasting effects on forest ecosystem C dynamics because of their impacts on the age-class structure of the forest (Böttcher et al. 2008). In any assessment of mitigation potential, the impact of the mitigation activity relative to the baseline activity must be isolated. This means that either the other influences on emissions and removals must be "factored out" (Canadell et al. 2007; Böttcher et al. 2008) or the effect of an influence must be included in both the baseline calculation and the assessment of mitigation activity so that it cancels out (Kurz 2010*b*).

Mitigation potential can be assessed in a number of ways (Fig. 1; Halsnæs et al. 2007). Physical potential (or biophysical potential in the case of biological systems) refers to the potential for GHG emission reductions or removals, ignoring cost and other constraints. For land-based mitigation, it considers only the biophysical limits of what is possible on the basis of ecological characteristics. Closely related is the concept of technical potential, which takes into account other constraints but does not consider costs. The concepts of market potential and economic potential both include costs. Market potential takes the private-sector point of view in considering how technical potential is affected by market costs and benefits and the private discount rate, which reflects how individuals or companies evaluate saving and investing choices. Economic potential adds to technical potential a consideration of both market and non-market costs and benefits (e.g., externalities, such as environmental co-benefits of mitigation activities) and uses a social discount rate meant to reflect attitudes about investments in societal and intergenerational welfare. Mitigation activities do not always impose a cost on society: some may have negative costs (i.e., they have a net benefit), for example if energy savings pay for the investment or if they provide co-benefits (Halsnæs et al. 2007).

Technical potential can be much smaller than biophysical potential (e.g., Strengers et al. 2008), and analyses of forest-related mitigation have been criticized for often ignoring considerations that generally constrain technical, and hence economic, potential (Boyland 2006). Constraints could include lack of knowledge at the local implementation level, land manager attitudes, or practical issues. For example, a practical constraint for an afforestation effort involving planting trees over large areas might be limited availability of seedlings. Such constraints could be overcome, but it would take time and may increase costs. Constraints on mitigation efforts could also be in the form of existing regulations or policies that have been implemented to satisfy social, environmental, and economic objectives of sustainable forest management, such as maintaining habitat with certain characteristics, or goals related to rural community stability and employment (e.g., ArborVitae 2008). Existing regulatory frameworks for forest management have been implemented with these objectives in mind, not mitigation (Boyland 2006). The details of such constraints will vary across the boreal zone, depending in part on the policies and practices established by provincial and territorial governments. Although technical potential is probably much smaller than biophysical potential, it could increase in response to mitigation policy. For example, regulations or land management practices that are currently in place could be altered to allow mitigation, thereby raising technical potential. The goals of mitigation would have to be balanced with the existing goals of current regulations and practices: the extent to which mitigation could be implemented without affecting the existing objectives would need to be evaluated, and the acceptability to the various stakeholders of any effects of mitigation would have to be determined.

Economic potential is typically even less than technical potential (e.g., McCarl and Schneider 2001; Lewandrowski et al. 2004; Nabuurs et al. 2007; van Minnen et al. 2008; Strengers et al. 2008), especially if only low-cost mitigation is considered. Nabuurs et al. (2007) assumed, on the basis of an assessment of the literature,

**Fig. 1.** Schematic diagram of mitigation potential. The two panels show hypothetical examples of baselines and mitigation potential of an activity starting at time $T_0$. Economic potential is generally less than technical potential, although at high carbon prices economic potential could approach or even equal technical or biophysical potential. Technical potential is generally less than biophysical potential, although over time many technical barriers could be addressed. (A) An example of afforestation on cropland in which the baseline is assumed to be a small constant source owing to soil carbon loss. Afforestation results initially in a source higher than the baseline but then becomes a sink as the trees grow. (B) An example of a landscape-level forest management mitigation activity in which the baseline is assumed to be a declining source that reflects recovery from natural disturbances. The mitigation activity accelerates recovery.




that 20% of the technical potential for forest-related mitigation in Europe and North America could be achieved at costs up to \$20US/t $CO_2$. However, the economic potential should rise as carbon prices rise (Barker et al. 2007; Nabuurs et al. 2007), and the discussion of afforestation in section 3.1 provides a clear example of how sensitive mitigation potential can be to economic considerations.

This review focuses on biophysical mitigation potential, which represents the upper limit of what can be achieved, and provides a basis for more in-depth analysis of technical, market, and economic potential. Although not a focus here, quantification of technical potential is important because it requires identification of the constraints that mitigation policy may be able to address. Consideration of economic and market potential within and across sectors is crucial because society has finite financial resources to devote to mitigation, and lower cost mitigation options are likely to be more attractive than more costly options. For this reason, the UNFCCC specifies that policies and measures to deal with climate change should be cost effective to ensure global benefits at the lowest possible cost (Article 3.3, UNFCCC 1992). It is also for this reason that the IPCC focused in its 4th Assessment Report on the economic potential of global mitigation options (IPCC 2007*c*). Assessment of economic potential adds additional uncertainties and complexities to the uncertainties inherent in the estimation of biophysical and technical potential. For a review of issues in the analysis of mitigation costs, see Halsnæs et al. (2007); and for a review of forest economic mitigation potential globally, see Nabuurs et al. (2007). Examples of economic assessments, many of them relevant to the boreal zone, include studies of afforestation (e.g., van Kooten et al. 1999; Yemshanov et al. 2005; Yemshanov and McKenney 2008), the economically optimal harvest rotation age when C is considered along with timber (Plantinga and Birdsey 1994; van Kooten et al. 1995; Asante and Armstrong 2012), the trade-offs and co-benefits of managing for C, timber, and habitat objectives (e.g., Krcmar et al. 2005; Bourque et al. 2007; McCarney et al. 2008), and increasing the use of forest biomass for bioenergy (e.g., Stennes et al. 2009; Gautam et al. 2010; Ralevic et al. 2010). Reviews of forest C economic studies from around the world have tried to normalize diverse methodologies and assumptions and extract general conclusions (e.g., see Richards and Stokes 2004; Stavins and Richards 2005; Nabuurs et al. 2007; van Kooten et al. 2009).

## 2.2. Portfolio approach

A portfolio approach considers the timing, magnitude, and duration of the long- and short-term impacts of a set of forest-related mitigation activities, how these vary spatially depending on local or regional conditions, and costs. In conceptual terms, a portfolio could be designed such that different activities provide mitigation benefits at different points of time, taking into account the fact that the impact and cost of activities (and thus the selection of the most desirable activity) will vary from location to location. A portfolio approach can also help in distributing the risks arising from the possible failure of some activities to achieve the expected mitigation benefits.

The mitigation potential of the world's forests was comprehensively synthesized as part of the IPCC 4th Assessment Report (Nabuurs et al. 2007). As part of that report, Barker et al. (2007) suggested a global potential of 1.3–4.2 Gt $CO_2$e in 2030 at costs under $100US/t $CO_2$e (economic potential), with about two thirds of the potential being generated in developing countries, primarily from reduced deforestation of tropical forests (Nabuurs et al. 2007). This global potential is substantial and comparable to that of other major sectors (Barker et al. 2007) and roughly equivalent to about 5%–10% of current global emissions from fossil fuels and land-use change. However, the extent to which impacts of climate change on forests could affect this potential has not been evaluated. The range of forest-related activities that can be implemented for mitigation purposes is well known (e.g., Cooper 1983; Kupfer and Karimanzira 1990; Sampson et al. 1993; Schlamadinger and Marland 1996; IPCC 2000; Kauppi et al. 2001; Colombo et al. 2005; Nabuurs et al. 2007; Malmsheimer et al. 2008; Ryan et al. 2010; McKinley et al. 2011). In this paper, definitions of these activities are consistent with those used in Canada’s annual GHG inventory (Environment Canada 2013) and the methodological framework of the IPCC for estimating GHG emissions and removals (IPCC 2003, 2006). They are also consistent with those used in Kurz et al. (2013). Both activities that reduce emissions and those that increase removals can be useful; and activities generally can be divided into those involving land-use change (activity types 1A and 1B in Fig. 2), forest management at the stand (activity types 2A and 2B in Fig. 2) and landscape (activity types 3A and 3B in Fig. 2) levels, and the use of harvested forest biomass (activity types 4A and 4B in Fig. 2). Each of these is discussed in section 3.

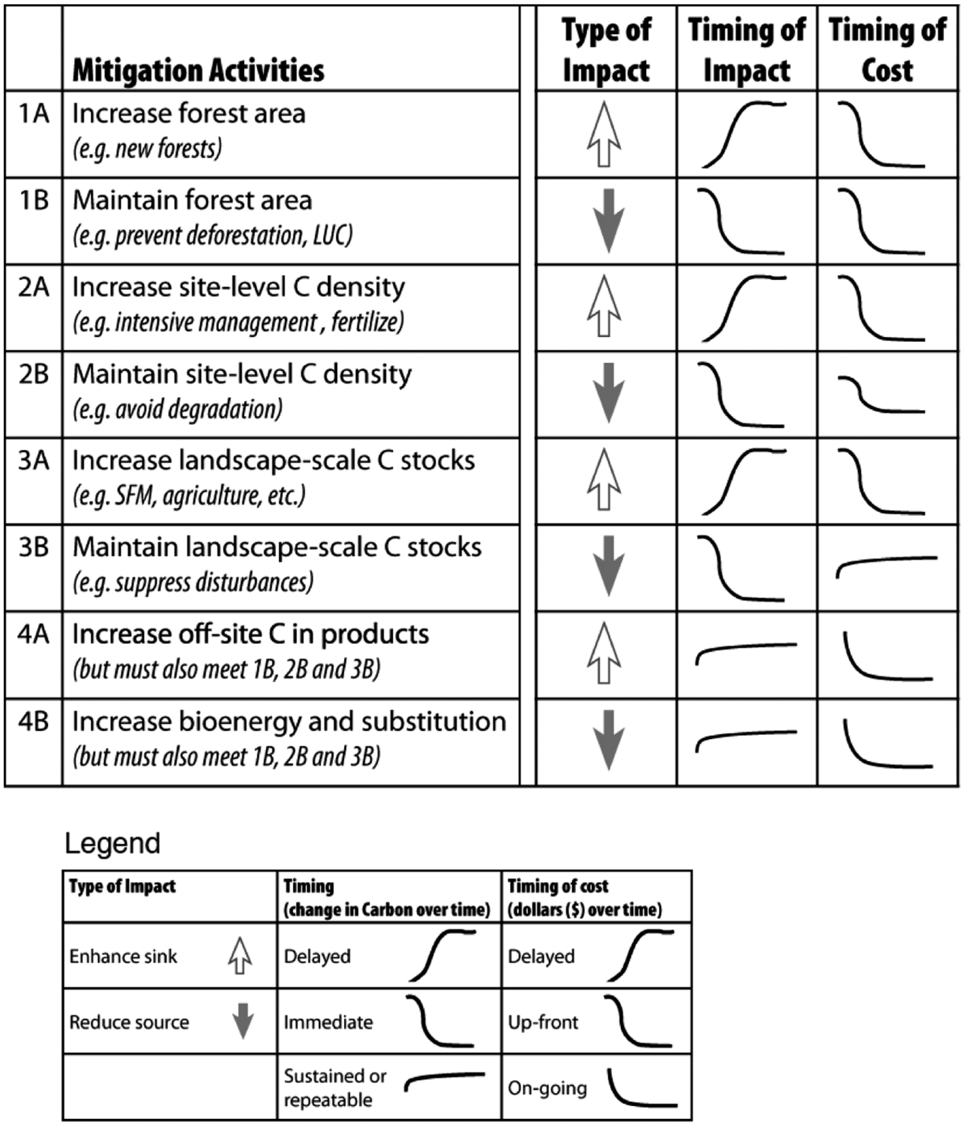
Mitigation efforts involving land-use change include increasing afforestation to increase removals of $CO_2$ from the atmosphere and reducing GHG emissions from deforestation. Afforestation is the creation of new forest through human-induced conversion of nonforest land to forest (UNFCCC 2006; Schlamadinger et al. 2007; FAO 2010) and typically involves assisted regeneration such as tree planting or seeding. The term reforestation is also sometimes used in the mitigation literature to refer to the creation of new forest, following the usage in Kyoto Protocol accounting rules (UNFCCC 2006; Schlamadinger et al. 2007); in such cases, it is distinguished from afforestation on the basis of how long the land had been without forest. For convenience, only the term afforestation is used here. Deforestation is the permanent or long-term human-induced conversion of forest to nonforest (UNFCCC 2006; Schlamadinger et al. 2007; FAO 2010). The definition of forest and methodological issues associated with ascertaining whether forest clearing is temporary or permanent affect deforestation estimates. Deforestation is distinguished from forest management: temporary removal of forest cover through harvesting, followed by regeneration of the forest, is considered part of forest management and does not result in a permanent or long-term loss of forest (IPCC 2000; Schlamadinger et al. 2007; FAO 2010; Masek et al. 2011). This distinction between permanent or long-term land cover change resulting from anthropogenic deforestation and temporary land-cover change that results from forest management is important (see Hansen et al. 2010; Kurz 2010*a*) because the two activities have quite different long-term implications for C stocks (Masek et al. 2011) and hence for mitigation assessment. Another important distinction in mitigation assessment is between gross and net deforestation: the latter is the net change in forest cover as a result of deforestation and afforestation combined (FAO 2010). Even when net deforestation is zero, emissions can be substantial owing to the different rates of C fluxes arising from deforestation and afforestation activities.

Mitigation could involve changes in forest management practices, with the mitigation impact depending on forest characteristics and details of the practice. Changes in practices to increase stand-level C density could include reducing forest degradation, intensifying silviculture, increasing fertilization, adjusting harvesting practices to reduce impacts on soil C, avoiding slash burning (and potentially removing slash for use as bioenergy), and improving regeneration. Mitigation could also involve increasing landscape-level C density, for example by reducing harvesting frequency, accelerating post-disturbance regeneration, reducing forest degradation, altering peatland management, and changing how fires, insect infestations, and other natural disturbances are managed. Finally, mitigation could involve harvested C, for example through increasing the substitution of bioenergy for fossil fuel use, increasing the substitution of wood products for more emissions-intensive materials like concrete, maximizing C retention through the use of construction techniques that use long-lived wood products, recycling, using discarded wood products for bioenergy, and minimizing emissions (especially of methane) from wood in landfills (Nabuurs et al. 2007). There are interdependencies among activities. For example, a cascading approach to the use of forest biomass has been found to have the most mitigation benefit (Werner et al. 2010): wood is first used for products, especially long-lived products that can substitute for emission-intensive materials; then recycled for other uses; and finally used for bioenergy.

The timing of mitigation impacts matter, and different activities have different impact profiles (see Fig. 2) that can be assessed relative to the time frames for mitigation objectives (e.g., to 2020 or 2050). In general, in the short run, the largest mitigation potential is in avoiding GHG emissions and maintaining C stocks, but over the longer run there can be significant potential in activities that increase removals and substitute forest biomass for more emissions-intensive products and energy sources. Mitigation objectives tend to require substantial emission reductions in the space of a few decades, but significant mitigation potential for some activities involving forests and forest C would take longer to accrue. However, the need for very long-term contributions to stabilization of atmospheric concentrations, especially if climate change has positive feedback effects (Friedlingstein et al. 2006), suggests that a role exists for mitigation activity that has an impact that builds and becomes significant only over an extended period. This is particularly true for some bioenergy-based options for which emissions typically increase initially relative to the baseline and emission reductions do not accrue until years or decades into the future (see discussion later in the paper).

Estimates of mitigation potential at multiple scales are needed to answer policy and implementation questions, but heterogeneity in forest characteristics and forest management policies creates assessment difficulties. Top-down studies use broad assumptions and generalizations to estimate potential for large regions (e.g., Chen et al. 2000). In contrast, bottom-up studies can encompass local or regional detail but may not be readily scalable (e.g., Bourque et al. 2007). Bottom-up estimates are likely to be particularly helpful for local forest managers, whereas top-down

**Fig. 2.** Impacts and costs of mitigation activities involving forest carbon (C). Activities can be divided into those that increase (1A) or maintain forest area (1B), increase (2A) or maintain (2B) stand-level C density, increase (3A) or maintain (3B) landscape C stocks, and increase C storage in harvested wood products (4A) or increase substitution of wood products and bioenergy for other more emissions-intensive products (4B) (reproduced from IPCC 2007*c*, fig. 9.4). LUC, land-use change; SFM, sustainable forest management.



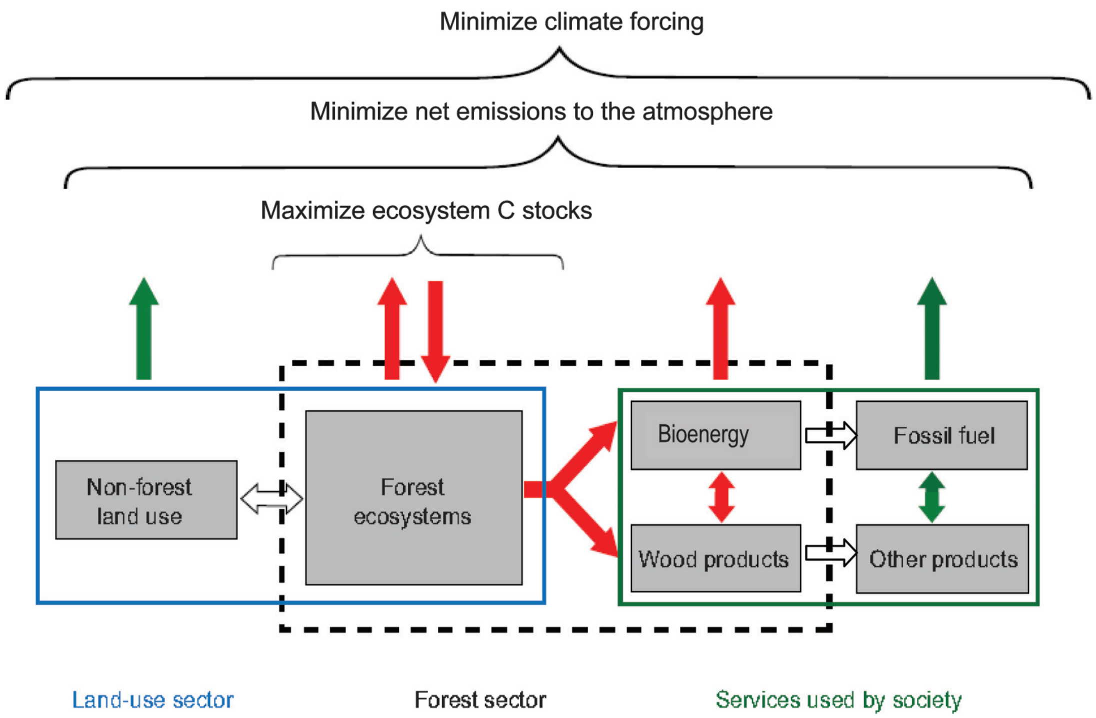



estimates may be most useful for policy-makers considering mitigation portfolios at the national or provincial level. Nabuurs et al. (2007) reviewed both types of estimates of forest mitigation potential from around the world and concluded that top-down estimates typically project much larger benefits than bottom-up estimates because they do not fully take into account variation in forest characteristics, local constraints, details of policy measures, and economic factors.

### 2.3. Systems perspective

Analyses have suggested that increases in ecosystem C stocks can be achieved by not harvesting natural forests or extending harvest rotation age, especially for forests with high C density (e.g., Cooper 1983; Harmon et al. 1990; Dewar 1991; Marland and Marland 1992; Schulze et al. 2000; Liski et al. 2001; Seely et al. 2002; Luyssaert et al. 2008; Foley et al. 2009), although the frequent occurrence of large-scale natural disturbances can affect whether this conclusion applies to boreal zone forests (Kurz et al. 1998; Colombo et al. 2005; Ter-Mikaelian et al. 2008). Researchers also have long recognized (e.g., Kupfer and Karimanzira 1990; Marland and Marland 1992; Sampson et al. 1993; Schlamadinger and Marland 1996) that assessments of mitigation must go beyond just considering the C pools in forest ecosystems: it is important to also consider C use and storage in HWPs and landfills, substitution of wood for more emissions-intensive products and fossil fuels, and land-use change involving forests. Such activities are highly interconnected, leading to the conclusion that mitigation analyses and the establishment of a mitigation portfolio need to be based on an integrated assessment of the various mitigation possibilities, in the context of other objectives for forests and the need for both mitigation and adaptation to climate change (Nabuurs et al. 2007; Canadell et al. 2010; Obersteiner et al. 2010). In the same way, it is important that assessments of mitigation consider non-$CO_2$ emissions and all forest ecosystem C pools, defined by the IPCC (2006) as aboveground biomass, belowground biomass, dead wood, litter, and soil organic matter. Nabuurs et al. (2007) emphasized the importance of a systems approach to understanding forest C mitigation potential, concluding that “[i]n

**Fig. 3.** A systems approach to climate change mitigation involving forests, showing relationships between the forest sector, land-use sector, services used by society, and the atmosphere. Red and green arrows represent the exchange of carbon or greenhouse gases between the atmosphere and components of the system or the transfer of carbon between system components. White arrows represent relationships between components that influence emissions and removals. Forest ecosystems exchange carbon with the atmosphere as part of the global carbon cycle and also emit other greenhouse gases as a result of fire and anaerobic decomposition. Expanding forest area, e.g., through afforestation, can reduce land available for agriculture and where this leads to deforestation for agriculture or intensification of agriculture the higher emissions in the sector will partly off-set sink benefits from forest expansion. Use of forest land and increases and decreases in forest area are influenced by society's use of other land types. Biomass is removed from the forest through harvesting for bioenergy and wood products that can substitute for more emissions-intensive products. A goal of maximizing ecosystem carbon storage ignores these other forest-related influences on greenhouse gas concentrations in the atmosphere. A goal of minimizing net emissions to the atmosphere takes into account the system as a whole. Greenhouse gas mitigation activities could affect the physical properties of land surfaces and taking this into account suggests an even broader goal of minimizing impacts on climate. Examining any one part of the system in isolation could misrepresent climate change mitigation potential (adapted from IPCC 2007*c*, fig. 9.3).




the long term, sustainable forest management strategy aimed at maintaining or increasing forest C stocks, while producing an annual yield of timber, fibre, or energy from the forest, will generate the largest sustained mitigation benefit" (p. 543). Figure 3 shows that focusing on the net effect of the system will shift the emphasis from maximizing ecosystem C stocks to minimizing net emissions to the atmosphere across the system.

A systems perspective is increasingly being used, and its importance demonstrated (e.g., Gustavsson et al. 2006*a*; Nabuurs et al. 2007; Hennigar et al. 2008; Ter-Mikaelian et al. 2008; Lippke et al. 2010; Ryan et al. 2010; Werner et al. 2010; Ingerson 2011; McKechnie et al. 2011; Ter-Mikaelian et al. 2011). The potential of alternative mitigation actions determined using a systems perspective may be quite different than that determined by examining only one part of the system: the choice of system boundary and time horizon for the analysis is very important in examining the GHG consequences of actions (Nabuurs et al. 2007). The systems perspective highlights the fact that interactions across sectors are important; and these can be complex and dynamic as demand and supply shift in response to shifting relative prices and costs, affecting economic mitigation potential through time (e.g., Barker et al. 2007; EPA 2009). Moreover, the systems perspective also highlights the fact that there are trade-offs between mitigation activities aimed at increasing C storage in the forest ecosystem, those aimed at increasing C storage in HWPs, and those aimed at increasing substitution benefits through the use of wood in place of other products or fossil fuels (with C storage in products and substitution in some cases going hand-in-hand). Efforts to produce one of these three types of benefits often reduce one or both of the other two types of benefits (e.g., Perez-Garcia et al. 2005*b*; Hennigar et al. 2008; Werner et al. 2010). This is a fundamental constraint that becomes apparent when comprehensive system-based analyses of mitigation options are based on an integrated assessment of each component of the system and take into account the temporal patterns of the emissions and removals for each component. This suggests that simplifying assumptions about any component should not be used in mitigation analyses because they may lead to erroneous conclusions. Examples of such simplifying assumptions, both discussed in section 3, are that C transferred out of the forest for HWPs is emitted immediately and bioenergy has no $CO_2$ emissions because sustainably managed forest growth across space or over time will offset the emissions caused by combustion.

The importance of the spatial aspect of system boundaries is illustrated by the issue of leakage, which could be either positive or negative, although most of the focus in assessments of mitigation potential is on negative leakage. An activity could have mitigation benefits within the study region, but it could produce changes in emissions outside the region (Schwarze et al. 2002). For example, harvest rates could be reduced in the boreal zone in the expectation of mitigation benefits, but the harvesting could be displaced to another area of the country (or abroad) owing to dynamic market responses and price changes (e.g., as occurred following reductions in harvest on federal lands to conserve habitat for the northern spotted owl (*Strix occidentalis caurina*) in the western United States (Wear and Murray 2004)). The extent to which displacement occurs would be affected not only by economic factors but also by policy factors (e.g., tenure arrangements and sustainable forest management practices in the case of harvesting in Canada's boreal forest). If harvest displacement occurs, then increasing the spatial boundary of analysis would reveal that the mitigation benefits would be lower than those calculated for the original study region. Although challenging to estimate, the leakage effects of forest conservation at global and national levels have been found to be potentially large (Murray et al. 2004; Sohngen and Brown 2004; Gan and McCarl 2007; Sun and Sohngen 2009), and leakage could be an important factor in determining the mitigation benefits of afforestation and reduced deforestation in the boreal zone, depending on the land use displaced by the mitigation activity and the C consequences of that displacement (IPCC 2000).

### 2.4. Biogeophysical considerations

Assessments of climate change mitigation activities involving forest C have tended to focus on the biogeochemical effectiveness of the activities in removing $CO_2$ from the atmosphere or reducing GHG emissions (e.g., Smith et al. 1993; Ryan et al. 2010). However, this focus is now understood to produce an incomplete picture, and there is increasing recognition that it is also important to examine the range of biogeophysical pathways by which potential mitigation actions affect the Earth's climate system (Marland et al. 2003; Nabuurs et al. 2007; Bonan 2008; Jackson et al. 2008; Anderson et al. 2011; Anderson-Teixeira et al. 2012). What matters most for mitigation is minimizing climate forcing (Fig. 3), and this perspective could lead to different conclusions about the mitigation potential of specific options than those that would be reached when only GHG effects are considered. Biogeophysical mechanisms determine how the physical properties of the land surface, such as albedo (the fraction of incident solar energy that is reflected by the Earth's surface) and evapotranspiration, affect climate. The biogeophysical mechanism that has received most attention in the context of mitigation is the effect of forest cover on albedo. This is especially important in the boreal zone, where the presence of trees on snow-covered landscapes leads to a marked lowering of albedo for many months during the year (Betts and Ball 1997). The presence of trees, in particular conifers, leads to greater heating of air masses over the boreal zone because of the increased absorption of solar radiation (Pielke and Vidale 1995).

The inclusion of albedo feedbacks in global modelling of climate has indicated that tree cover in the boreal zone exerts a strong, regional warming effect on climate that extends to lower latitudes and also leads to feedback effects on sea ice and sea surface temperatures (Bonan et al. 1992). Subsequent analyses that include forest-albedo feedbacks have questioned the effectiveness of afforestation as a climate change mitigation strategy in the boreal zone because albedo effects would dominate the $CO_2$ removal effects, leading to net warming rather than net cooling (Betts 2000; Claussen et al. 2001; Gibbard et al. 2005; Bala et al. 2007; Betts et al. 2007; Thompson et al. 2009; Arora and Montenegro 2011). The albedo impacts of deforestation on surface air temperatures in the boreal zone also need to be considered (Sharratt 1998; Lee et al. 2011), and the above studies imply that the calculated effectiveness as a mitigation strategy of activities to avoid boreal zone deforestation may be lower if forest-albedo feedbacks are considered in addition to GHG emission reductions.

It is important to note, however, that many models of albedo feedbacks used closed-canopy conifer forest, whereas afforestation and post-fire natural regeneration typically involve deciduous species (poplar, aspen) whose biogeophysical feedbacks differ significantly from those of conifers (Amiro et al. 2006). Compared with coniferous forests, deciduous forests exert a smaller heating effect (and in some cases, a summer cooling effect) because of relatively higher albedo and greater rates of summer evapotranspiration that may promote cloud development (Bonan 2008; Anderson et al. 2011). Thus, there are opportunities to modify practices to minimize the biogeophysical warming feedback of afforestation (e.g., by choosing deciduous species, Anderson et al. 2011; Cai et al. 2011). As well, choosing areas to afforest that have high productivity and relatively low snow cover would help. Pongratz et al. (2011) suggested that the conversion of forests to agriculture has historically been undertaken preferentially in such areas in northern regions. Deforestation of high-productivity areas with high C stocks leads to above-average $CO_2$ emissions, while deforestation of low-snow areas results in below-average albedo impacts. As a result, reversing these historic deforestation patterns with afforestation could have a cooling effect in temperate and boreal regions (Pongratz et al. 2011).

On the basis of the importance of biogeophysical feedbacks, it has been suggested that they need to be considered along with GHGs when assessing the mitigation potential of forest and other land management activities (Pielke et al. 2002; Marland et al. 2003; Thompson et al. 2009; Schwaiger and Bird 2010; Anderson et al. 2011; Arora and Montenegro 2011; Anderson-Teixeira et al. 2012). This is a challenging prospect, given the limitations of our current understanding of the complexity of the interacting biogeophysical processes. For example, the snow-albedo feedback of forests is much greater in spring than in other seasons (Bonan et al. 1992; Foley et al. 1994); a secondary feedback might be produced whereby forest-induced increases in spring temperature (Hogg et al. 2000) lead to earlier photosynthesis, extension of the growing season, and enhanced C uptake. Furthermore, future climate warming may induce earlier snowmelt, thus reducing the impact of forest cover on spring albedo in boreal regions (Bonan 2008). Depending on their significance, other biogeophysical and biogeochemical processes may also warrant consideration, such as the potential cooling effect induced by the emission of biogenic organic vapours (e.g., terpenes) from boreal forests, which produces aerosols and increases cloud cover (Kurtén et al. 2003; Spracklen et al. 2008).

The consideration of all such effects can be extended to the accounting of forest fire suppression as a mitigation strategy. The substantial GHG combustion emissions that occur as a result of fire in Canada's boreal forests (Kurz et al. 2013) have been projected to increase as the climate changes (Amiro et al. 2009; Balshi et al. 2009; Metsaranta et al. 2010). Fires could act as a positive feedback on climate warming via these emissions as well through production of aerosols and deposition on snow and ice of black carbon resulting from the burning (Randerson et al. 2006), but measurements show that fires can also contribute to climate cooling through the more immediate effects of smoke followed by post-fire changes in vegetation that profoundly affect surface energy balance (Amiro et al. 2006). Analyses have indicated that these biogeophysical feedbacks may be more important over the long term so that increases in fire could conceivably lead to negative feedbacks on climate warming (Randerson et al. 2006; O'Halloran et al. 2012). Such an interpretation is consistent with the recent analysis of climate feedbacks following fire-induced conversion of closed-canopy spruce forests to lichen-dominated woodlands in eastern Canada (Bernier et al. 2011). In turn this suggests that it will be important to consider carefully the extent to which efforts to reduce forest fires in the boreal zone reduce climate warming.

Similar considerations may also be applicable for assessing the mitigation effectiveness of programs to suppress insect outbreaks. Although the effects of tree-killing insects lead to large increases in C emissions (Kurz et al. 2008*a*; Dymond et al. 2010*b*), the accompanying increase in albedo of insect-killed stands could induce a net cooling effect, according to recent analyses by O'Halloran et al. (2012). In boreal regions with winter snow cover, post-disturbance increases in albedo are strongly affected by the rate of snag fall (O'Halloran et al. 2012), suggesting that the effectiveness of salvage harvesting as a mitigation option could be improved by the effects on albedo.

The development of methods to readily compare biogeochemical and biogeophysical effects using a common metric (Betts 2008; Anderson et al. 2011; Anderson-Teixeira et al. 2012), similar to the use of global warming potentials to combine the effects of multiple GHGs, will help in understanding the significance of biogeophysical effects. Radiative forcing, a measure of changes to the energy balance of the earth-atmosphere system that determine climate, is often used in scientific assessments of influences on climate (e.g., IPCC 2007*a*) but is not commonly used in policy discussions. In international discussions under the UNFCCC and domestic discussions around the world concerning mitigation efforts, $CO_2$e is used as the metric for policy formulation and assessment of progress. Thus, if methods to readily convert biogeophysical impacts to $CO_2$e impacts were available, it would be easier to include biogeophysical impacts in policy discussions. One challenge is the spatial difference in impacts: GHG emissions anywhere have global impacts on climate, whereas biogeophysical changes have climate impacts that can vary by location. Another challenge is the temporal difference in impacts: most GHGs produce long-lasting effects on climate, whereas biogeophysical changes have impacts that can change rapidly (e.g., seasonally for albedo) or with forest growth.

## 3. Current understanding of boreal forest mitigation potential

### 3.1. Afforestation

At least 5.6 Mha was afforested globally in 2005 (FAO 2010), and many studies suggest that substantially increasing afforestation rates is an important GHG mitigation option over time scales of several decades (e.g., Obersteiner et al. 2006; Nabuurs et al. 2007; Strengers et al. 2008). Afforestation could also have other benefits in the case of longer rotation plantations, such as contributing habitat and reducing forest fragmentation, although there are concerns about the albedo impacts in high-latitude regions, as described earlier in the paper, and the risk posed by natural disturbances could also be important in some cases (Mansuy et al. 2013). Historically, little afforestation has occurred in Canada. It averaged about 1.0 kha/year in 2000–2008 in the boreal zone (composed of seven ecozones), or 35% of total afforestation in Canada (Kurz et al. 2013). As a result, C removals due to afforestation in the boreal zone also have been quite small historically (Kurz et al. 2013). However, afforestation is the forest-related mitigation activity most thoroughly assessed in Canada as a whole and in the boreal zone (e.g., Dominy et al. 2010), both because it can increase soil and biomass C and because it can provide a sustainable feedstock for bioenergy (Yemshanov and McKenney 2008; Amichev et al. 2012). The GHG emissions due to plantation site development and tending (e.g., fertilization and weed control) and harvesting operations will affect the mitigation potential though many studies do not include these impacts and they will typically be relatively small compared with the sequestration (e.g., Gaboury et al. 2009).

Afforestation could occur on marginal or other agricultural lands, where issues of competition with crop production for food and forage can arise, or on non-agricultural lands. Relatively few studies have examined the latter. Lemprière et al. (2002) projected a net increase of 0.3 t C/ha/year (1.1 t $CO_2$/ha/year) over 50 years when white spruce (*Picea glauca* (Moench) Voss) is planted in the east-central part of Saskatchewan's boreal zone to replace 3 kha of degraded forest containing low-density aspen (*Populus tremuloides* Michx.) considered not sufficiently restocked after harvesting several decades in the past. The analysis did not consider impacts on soil C and assumed some C loss owing to fire and insect infestations. Gaboury et al. (2009) examined the mitigation potential of afforestation with black spruce (*Picea mariana* (Mill.) B.S.P.) in the boreal zone in Quebec where post-fire regeneration has failed and resulted in low-density black spruce – lichen woodland (potentially 1.6 Mha). Afforestation resulted in a net ecosystem increase of 1.1 t C/ha/year (4.0 t $CO_2$/ha/year) over 70 years, and alternative assumptions about low and high rates of black spruce growth and low and high estimates of annual afforestation area burned resulted in a range of 0.2–1.9 t C/ha/year (0.7–7.0 t $CO_2$/ha/year). This analysis considered all ecosystem C pools except understory vegetation such as lichens, mosses, and shrubs; the authors noted that such understory vegetation could be important. Gaboury et al. (2009) also found that total GHG emissions associated with afforestation-related operations (1.3 t $CO_2$e/ha for the 70-year period) were minor compared with the net sequestration.

Both Lemprière et al. (2002) and Gaboury et al. (2009) assumed that afforestation required removal of low-density growing forest in their baselines, which affected the net impact of the activity. In contrast, in their analysis of 4 kha of non-agricultural land in a forest management unit in northeastern Ontario (near Timmins), Biggs and Laaksonen-Craig (2006) assumed that there was little potential for sequestration in the absence of afforestation and, therefore, used an assumption that the existing C stock was in a steady state. They estimated that afforestation using 50% jack pine (*Pinus banksiana* Lamb.) and 50% trembling aspen would result in sequestration of 2.4–3.0 t C/ha/year (8.8–11.0 t $CO_2$/ha/year) over 50 years depending on whether low or high productivity sites were assumed. Like Gaboury et al. (2009), Boucher et al. (2012) examined low density woodland in boreal Quebec but assumed understory planting rather than removal of existing trees. They estimated that afforestation on low productivity sites would yield 0.8–1.4 t C/ha/year (3.1–5.1 t $CO_2$/ha/year) over 70 years depending on the species planted.

Afforestation on marginal or other agricultural land has been assessed most frequently because the biological and economic productivity of tree species on this land would be expected to be relatively good given that much of the land had been previously cleared of forest for agricultural production (Suchánek 2001; McEwen 2002). There are roughly 29.5 Mha of agricultural land in the boreal zone and hemiboreal subzone (a transitional area lying immediately to the south of the boreal zone) on the basis of analyses of satellite-based land cover classification (GEO 2009). Cropland in the boreal zone (seven ecozones) covered 10.9 Mha in 2011, or 23% of the total area of cropland in Canada, according to Canada's annual GHG inventory (Environment Canada 2013). Pinno and Bélanger (2008) found that 8 ha afforested with white spruce and Siberian larch (*Larix sibirica* Lebed.) in 1955 in the Saskatchewan Boreal Plains ecozone (Prince Albert area) and, subject to no stand tending, had gained 2.7 t C/ha/year (9.8 t $CO_2$/ha/year) in the subsequent 50 years in the white spruce plots, compared with pasture plots, and 2.1 t C/ha/year (7.8 t $CO_2$/ha/year) in the Siberian larch plots. Amichev et al. (2012) modelled afforestation of 0.4 Mha of agriculturally marginal land using short-rotation coppice willow (*Salix* spp.) in the Saskatchewan Boreal Plains, assuming that harvesting occurs every 3 years after coppicing in the first year, and removal and replanting occurs after seven harvests. After 44 years, 0.3 t C/ha/year (1.1 t $CO_2$/ha/year) more than the baseline was stored in the ecosystem, and the cumulative harvested biomass for bioenergy was 4.6 t C/ha/year (16.9 t $CO_2$/ha/year). As is generally the case with estimates of biophysical mitigation potential, Amichev et al. (2012) noted that estimates of technical and economic potential were likely to be substantially lower.

Afforestation mitigation potential partly depends on factors influencing biomass growth and accumulation of C in dead wood, litter, and soil organic C pools. Where agricultural activities have reduced soil organic C stocks to below their pre-agriculture level, afforestation of the land can not only result in higher sequestration of C in biomass than the sequestration that occurs on agricultural lands but can also restore soil organic C stocks and allow accumulation of C in dead wood and litter pools. In a meta-analysis of 33 studies from around the world of the influence of afforestation on soil organic C, Laganière et al. (2010) found that soil organic C buildup after afforestation is greater on former croplands than on pasture or grazing land, deciduous tree species seem to have a greater capacity to store soil C than coniferous species, clay-rich soils have a greater capacity to accumulate C, and reducing site disturbance before planting may increase soil C buildup. The analysis included only three boreal sites, of which one (Pinno and Bélanger 2008, described earlier in the paper) was in Canada's boreal zone. However, afforestation may initially result in net emissions for varying periods of time depending on the tree species planted, past land use, the vegetation and soil characteristics of the baseline, and assumptions about effects of the activity on pre-existing dead wood, litter, and biomass. For slower growing species planted to replace degraded forest in the boreal zone, estimates of the time for planted areas to become net sinks relative to the baseline have ranged from around a decade (Lemprière et al. 2002; Boucher et al. 2012) to 27 years (Gaboury et al. 2009). For faster growing species (e.g., coppice willow, hybrid poplar) on marginal agricultural land in the boreal zone, the time can be around 4–6 years (Arevalo et al. 2011; Cai et al. 2011; Amichev et al. 2012).

Although some studies have only examined the biophysical mitigation potential of afforestation, many have also assessed the economic attractiveness of this activity for wood products, bioenergy, and C to derive a more in-depth understanding of mitigation potential (e.g., van Kooten et al. 1999; Stephens et al. 2002; McKenney et al. 2004; Yemshanov et al. 2005; Biggs and Laaksonen-Craig 2006; Yemshanov et al. 2007; Yemshanov and McKenney 2008; Ramlal et al. 2009). Species examined have included hybrid poplar and willows, Norway spruce (*Picea abies* (L.) Karst), red pine (*Pinus resinosa* Ait.), and slow-growing hardwoods. Fast-growing, short-rotation (16–25 years) plantations of various hybrid poplar clones often show the best investment potential (van Kooten et al. 1999; Yemshanov et al. 2005, 2007). However, in some cases slower growing, longer rotation species such as red pine and Norway spruce may be financially more attractive because they involve lower establishment and maintenance costs (Yemshanov et al. 2007). Key factors that influence the economic attractiveness of short-rotation plantations include land opportunity costs, landowner risk aversion regarding woody crops (Plantinga et al. 2002; Roberts and Lubowski 2002; Smith et al. 2005), plantation establishment and maintenance costs, tree growth rates and C sequestration potential, and revenue streams for wood products and C sequestration. Risk of mortality or reduced growth due to disturbance (drought, fire, insects, and disease) also can be important in decision-making about plantations (Volney et al. 2005).

Previous analyses have shown that afforestation in the boreal zone can be an attractive investment provided an economic value can be obtained for C sequestration. For this paper, we used the Canadian Forest Service - Forest Bioeconomic Model (CFS-FBM, Yemshanov et al. 2007) to assess the economic potential for afforestation of marginal agricultural land using hybrid poplar in the boreal zone and hemiboreal subzone. Our analysis is similar to previous analyses using the model that included land in the boreal zone (see McKenney et al. 2004; Yemshanov et al. 2007), but we used a higher resolution representation of agricultural land as well as more recent estimates of establishment costs for afforestation plantations (see NRCan 2010*b* for details). Harvests were assumed to occur at year 20 with spatial variation in yields. Table 1 provides estimates of the feasible land for afforestation — the area for which it would be financially attractive to engage in afforestation — taking into account C sequestration potential during one 20-year rotation period at different C prices. Very little afforestation has occurred in the boreal zone (Kurz et al. 2013), which suggests that it has not been an economically attractive activity. Table 1 illustrates how increases in the value of C could profoundly affect the land area over which afforestation would be economically feasible and hence the potential for C sequestration. At \$20CAD/t $CO_2$, afforestation with hybrid poplar is attractive on nearly 28 Mha, or almost the entire agricultural land in the boreal zone and hemiboreal subzone, with sequestration averaging 2.1 t C/ha/year over 20 years. In Table 1, the reduction in the average sequestration rate as the C price rises reflects the fact that increasingly less productive land becomes economical for afforestation as the price rises.

There is evidence that private landowners may be reluctant to convert agricultural land to other land uses, and this reluctance has not been considered in the results reported in Table 1. The explicit inclusion of the costs of undertaking land-use change and the fact that afforestation prevents other uses of the land (owing to the multiple-year rotation lengths of woody crops) create an incentive for landowners to delay afforestation decisions (Plantinga et al. 2002; Isik and Yang 2004). Ultimately this additional cost makes afforestation less attractive and landowners are likely to remain in agriculture to take advantage of the increased flexibility associated with annual crop cycles. However, use of very short rotation species such as willow for afforestation (see Amichev et al. 2012) could overcome some of the resistance, as could higher C prices. Lack of information about the future value that could be obtained for C sequestered by afforestation, risks that the C could be emitted (an issue of the impermanence of the C storage), and volatility in agricultural commodity prices may further decrease the attractiveness of afforestation (Plantinga et al. 2002; Stevenson 2003*a*, 2003*b*; Lubowski et al. 2006; Maréchal and Hecq 2006; Yemshanov et al. 2012). Thus, although the biophysical potential of afforestation to sequester C is high, the economic potential is lower and varies substantially with the value of C. The possibility of leakage effects due to boreal zone afforestation and effects on albedo, discussed earlier in the paper, have not been taken into account in any of the studies cited in this section.

### 3.2. Reduced deforestation

Global gross deforestation was about 13 Mha/year in 2000–2010, almost all of it in developing countries, although the net forest loss was 5.2 Mha/year because of afforestation and natural forest expansion (FAO 2010). Pan et al. (2011) estimated that gross emissions from tropical deforestation for 2000–2007 were 10.34 ± 1.65 Gt $CO_2$/year. This is equivalent to about 27% of global total $CO_2$ emissions in the period (on the basis of estimates of fossil fuel combustion and cement production emissions from Global Carbon Project 2011). These emissions were offset in part by removals of 6.31 ± 1.98 Gt $CO_2$/year from tropical forest regrowth on lands affected by past deforestation. Reducing the high rate of deforestation is considered to be among the key forest-related strategies to mitigate climate change (e.g., Nabuurs et al. 2007; Keith et al. 2009). The focus has been on reducing emissions from deforestation and degradation (REDD) in tropical forests of developing countries because that is where most deforestation occurs (FAO 2010). Relatively little attention was paid to deforestation in industrialized countries until the 1997 Kyoto Protocol required developed country signatories to monitor and account for emissions in 2008–2012 from deforestation that had occurred since 1990 (UNFCCC 2006; Schlamadinger et al. 2007). Developed countries (not including the United States) reported combined gross deforestation emissions of 135.4 Mt $CO_2$e in 2009 (UNFCCC 2011*a*), a little over 1% of gross emissions from tropical deforestation.

Most simply, the biophysical mitigation potential of reduced deforestation is equal to the avoided emissions, plus any sink that would exist if the forest were allowed to grow rather than being

**Table 1.** Estimates of afforestation and carbon sequestration potential over 20 years in the boreal zone and hemiboreal subzone for different carbon prices at a 4% discount rate.

| Carbon price ($CAD/t $CO_2$) | Feasible land (kha) | Average sequestration (t C/ha/year) | Total sequestration (Mt C) | Total sequestration (Mt $CO_2$) |
|---|---|---|---|---|
| 0 | 0 | 0 | 0 | 0 |
| 5 | 1 028 | 3.0 | 61.7 | 226.0 |
| 10 | 13 466 | 2.4 | 646.4 | 2368.4 |
| 20 | 27 697 | 2.1 | 1167.5 | 4277.9 |

**Note:** These estimates are based on an annual accounting of net carbon flows (i.e., net $CO_2$ emissions are costs and net carbon sequestration is a benefit in the year in which they occur). These estimates do not consider any additional price adjustments that may occur as a result of the nonpermanence of plantation-based carbon sequestration.



permanently removed. In Canada, research has quantified the amount of deforestation and its impacts on C (Fitzsimmons 2002; Fitzsimmons et al. 2004; Masek et al. 2011; Environment Canada 2013; Kurz et al. 2013). Estimates derived for Canada's GHG inventory (Environment Canada 2013) show that gross deforestation in the boreal zone (seven ecozones) in 1990–2008 averaged 35.5 kha/year, accounting for under 0.02% of the forest area annually (Kurz et al. 2013). This low rate reflects the very large forest area and very low population density of the boreal zone. Although the overall deforestation rate is quite low, hot spots with much higher rates occur, for example in regions of forest–agriculture interface or resource development. One such area is the boreal transition region of the southern Boreal Plains ecozone (Fitzsimmons 2002; Hobson et al. 2002).

Although Canada's rate of boreal zone deforestation is small in comparison to global deforestation and the size of Canada's boreal forest, the resultant average GHG emissions of 1.6 Mt C/year (6.2 Mt $CO_2$e/year) in 1990–2008 were not insignificant (Kurz et al. 2013) and there would also be emissions over time from the 0.72 Mt C/year (2.6 Mt $CO_2$e/year) transferred to the forest products sector. In the Boreal Plains of central Saskatchewan (Prince Albert area), Fitzsimmons et al. (2004) compared the C densities of pasture and conventional till cropland cleared 35 or more years in the past with those of mature aspen (*Populus tremuloides* Michx.) forest. They were unable to quantify differences in soil C but concluded that C in aboveground live and dead vegetation greater than 10 cm in diameter would be reduced by 30–75 t C/ha (110–275 t $CO_2$/ha) over time. Grünzweig et al. (2004) found an average reduction in ecosystem C density of 112 t C/ha (411 t $CO_2$/ha) when black spruce forests in boreal Alaska were converted to agriculture. Overall, between 1990 and 2008, deforestation in the Canadian boreal zone resulted in a reduction in ecosystem C of 65 t C/ha (240 t $CO_2$/ha) (based on data shown in Kurz et al. 2013).

The causes of deforestation in Canada's boreal zone are complex and reflect economic, regulatory, and other drivers that vary across the country and across sectors. It is necessary to identify and understand both the causes and practices used in clearing forest at local to regional scale to assess technical and economic mitigation potential, which could be much lower than biophysical potential. Since 1990, the main contributors to boreal deforestation have been land clearing for agriculture, oil and gas development, episodic hydro reservoirs and infrastructure, and forestry roads and landings with small amounts due to mining, municipal development, transportation infrastructure, industry, recreation, and peat extraction (Environment Canada 2013; Kurz et al. 2013). More of the deforestation in the boreal zone occurred in the Boreal Plains than in any other ecozone (64%), with agriculture and oil and gas development as the primary causes (Environment Canada 2013). Studies of deforestation for agriculture in the Boreal Plains of Saskatchewan in the three decades before 1990 concluded that economic drivers such as agricultural support programs, commodity prices, and increases in farm size likely were important (Fitzsimmons 2002; Hobson et al. 2002). In a predictive model developed to evaluate factors that explain forest losses in the ecozone, Hobson et al. (2002) found that key factors influencing forest distribution were land ownership and land quality. Privately owned land had less forest than areas managed by the provincial government or First Nations. Deforestation occurred in all land-quality classes; but high-quality, privately owned agricultural lands were more fragmented than lower quality lands because of the higher road density.

Stopping deforestation has obvious mitigation benefits, but rather than stopping the economic activity that causes the forest loss, it may sometimes be possible to undertake the activity with less impact on the forest. For example, Schneider et al. (2003) suggested that reducing the width of the seismic lines used in oil exploration in the boreal zone in Alberta or increasing the harmonization of road building between the petroleum and forestry sectors could lessen the loss of forest. Reductions in seismic line width have already occurred and recent narrow lines (approximately 2 m) would not be classified as deforestation (Environment Canada 2013), though they would result in reductions in forest C stocks until the forest has regenerated.

In addition to simply reducing deforestation rates, mitigation efforts could also seek to influence the temporal pattern and magnitude of GHG emissions and reductions in C density through deforestation practices that take into account pre-clearing C density, the manner in which cleared biomass is handled (e.g., burned, left on site to decompose, or used for forest products), and the specific land-use activities following deforestation. Variations in agricultural practices on cleared black spruce forest land were shown to have a significant effect on C storage in boreal Alaska: C losses were reduced when new fields were planted rather than left fallow, when perennial rather than annual crops were cultivated, and when nitrogen fertilization was reduced where soil moisture was not limiting (Grünzweig et al. 2003). Carbon losses can be reduced by selecting relatively carbon-poor sites for deforestation and (or) by implementing carbon-preserving practices (Grünzweig et al. 2003, 2004). The feasibility of mitigation through changing how deforestation is done will be influenced by the cost relative to the baseline practice.

### 3.3. Forest management

A large number of studies conducted in other countries are relevant to the assessment of the biophysical mitigation potential of changes to forest management (see Hines et al. (2009) for a partial bibliography), although the results of such studies are not always relevant to Canada's boreal zone. In Canada, various models have been used to show the importance of biophysical mitigation potential nationally (Kurz and Apps 1995; Chen et al. 2000), provincially (e.g., Colombo et al. 2005), and for specific landscapes (e.g., Seely et al. 2002; Neilson et al. 2007; Hennigar et al. 2008; Taylor et al. 2008; Hennigar and MacLean 2010). Researchers have quantified C responses to alternative harvesting systems, scarification, fertilization, and other management practices at specific sites (e.g., Foster and Morrison 2002; Hazlett et al. 2005; Giasson et al. 2006; Jassal et al. 2008). Key considerations in management for long-term ecosystem C storage include the frequency of harvests, C density before harvesting, and frequency of natural disturbances (Kurz et al. 1998; Trofymow et al. 2008; Oneil and Lippke 2010). The impacts of climate change are also highly relevant and will influence both the baseline relative to which mitigation potential would be assessed and the result of the mitigation action. In addition to the effects of mitigation activities on ecosystem C stocks, the effects of forest operations (e.g., fossil fuel use in harvesting, silvicultural activities, and transporting biomass) on GHG emissions need to be considered (Sonne 2006; Lippke et al. 2010). The emissions might be greater in relatively remote boreal regions than elsewhere due to longer distances travelled during the operations.

The area of forest in Canada's boreal zone that is currently subject to management is about 145 Mha, about 54% of the forest in the zone and 63% of the total managed forest in Canada (Environment Canada 2013; Kurz et al. 2013). The managed forest is defined as including forests managed for timber and non-timber resources (including national and provincial parks) or where there is intensive protection against natural disturbances (Environment Canada 2013; Kurz et al. 2013). The unmanaged portion is not subject to significant direct human activity that affects C stocks, and thus it is the managed portion that is relevant because mitigation concerns human activities.

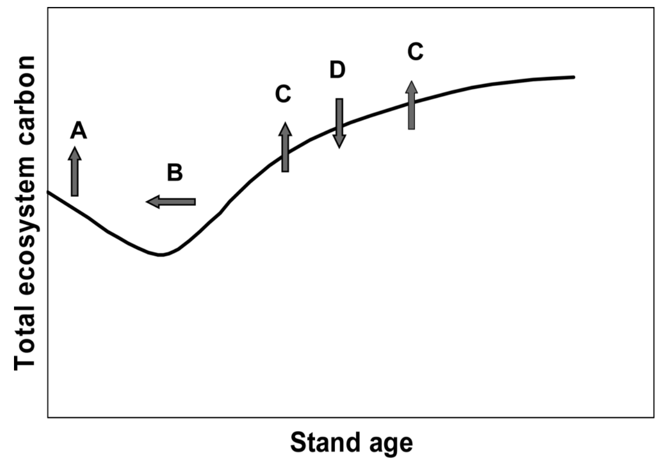
#### *3.3.1. Stand level*

Stand-level mitigation options are aimed at increasing the C density (C stored per unit of area) during the stand development and disturbance cycle. Stand-level C dynamics in the boreal forest are characterized by post-disturbance C stock declines as heterotrophic respiration losses from dead organic matter (DOM) and soil C pools exceed C uptake rates in regrowing forests. As tree growth rates accelerate and respiration losses from post-disturbance DOM decline, stand-level C dynamics revert to a C sink and the stand actively removes C from the atmosphere (Fig. 4; Kurz et al. 2013). Boreal forest stands can revert to net C sinks 10–20 years after disturbance (Litvak et al. 2003; Howard et al. 2004; Hyvönen et al. 2007; Grant et al. 2010; Coursolle et al. 2012). Boreal forests have relatively low tree growth rates, meaning that the mitigation potential of many forest management actions is likely to be less per unit area than that of actions in faster growing temperate or tropical forests.

Nevertheless, several forest management activities can be implemented to increase C stocks at different times during stand development. Carbon losses from the stand during and after harvest-related disturbances can be reduced (Fig. 4, arrow A). The amount of residual DOM left after clear-cut harvest, the treatment of residual material, and post-harvest site preparation activities can all affect the amount and composition of DOM and soil C pools (e.g., Nave et al. 2010). Decisions on the treatment of post-harvest biomass are typically guided by considerations of insect and fire risk, facilitation of seeding or planting, and creation of planting space that is otherwise occupied by slash piles. Where slash reduction (through burning or removal of slash from the forest for use as bioenergy) leads to short-term C losses from the ecosystem, these losses may eventually be offset if such treatments accelerate the rate of forest establishment, reduce fire risk, or increase tree growth rates. As always, the time dynamics of C costs and benefits need to be evaluated.

Regeneration delays can be reduced to accelerate the transition from C source to sink (Fig. 4, arrow B). Planting, seeding, site preparation (Giasson et al. 2006), and control of competing vegetation or herbivory (e.g., deer browsing) are all mechanisms through which the transition to a C sink can be accelerated (Kurz and Apps 1995). During all stages of stand development, the rate of tree growth (and thus C accumulation) can be influenced through tree species selection, through tree improvement programs (plant breeding or genetic engineering), and through silvicultural activities including fertilization (see later in the paper), management

**Fig. 4.** Schematic diagram of stand-level forest carbon dynamics and forest management mitigation actions with the following aims: (A) modify carbon density during or after disturbance; (B) reduce regeneration delay; (C) increase growth rates; and (D) conduct partial harvesting that does not increase ecosystem carbon stocks but can improve ecosystem health and drought tolerance and increase carbon stocks in harvested wood products or allow for increased product substitution including bioenergy.




of competing vegetation, and protection against insect and diseases (Fig. 4, arrow C). Finally, silvicultural activities such as thinning can help maintain forest health, reduce moisture stress, and provide biomass from recent or anticipated future mortality. This could reduce ecosystem C stocks but may allow for increased accumulation of HWP C stocks (Fig. 4, arrow D) (Briceño-Elizondo et al. 2006) or mitigation benefits through emission reductions resulting from product substitution and bioenergy use. Moreover, thinning can increase the size of individual trees, and this effect may increase the suitability of harvested material for the manufacture of long-lived HWPs and thus prolong storage of C and increase substitution benefits from wood use.

In this paper, we do not review the literature on the range of stand-level management activities that are possible (for examples, see Colombo et al. 2005; Malmsheimer et al. 2008; Ryan et al. 2010). Much of the literature is focused on determining how stand productivity and stand volume can be improved from a commercial harvesting perspective, whereas consideration of mitigation potential needs to assess the impacts on stand C density more broadly, including aboveground and belowground biomass, deadwood, litter, and soil. To illustrate these broader considerations, here we examine increased fertilization as a mitigation activity. Much of the soil of the forest in Canada's boreal zone is characterized by nitrogen limitations, and phosphorous limitation also appears to be important (Maynard et al., In press), suggesting that fertilizer could be strategically applied for mitigation purposes. Baselines for nitrogen fertilization as a mitigation activity would need to take into account atmospheric deposition that reduces nitrogen deficiency, though the growth enhancement is unlikely to be significant (Kurz et al. 2013). Low levels of nitrogen deposition on the order of 1–2.5 kg/ha/year have been recorded in relatively remote parts of the boreal forest (Chen et al. 2000; Allison et al. 2009), and Houle and Moore (2008) measured deposition of 3 and 6 kg/ha/year in south-central Quebec. Field studies demonstrate that the effect of nitrogen fertilization on boreal forest C stocks varies according to biophysical characteristics including forest type, site type, stocking level, stand developmental stage, treatment characteristic, and time period studied (Turkington et al. 2002; Newton and Amponsah 2006; Ladanai et al. 2007). Fertilization can lead to a shift in the species composition of ground vegetation (Turkington et al. 2002; Olsson and Kellner

2006; Strengbom and Nordin 2008), although the effect on C storage could be minimal, depending on vegetation characteristics (Mäkipää 1995).

Nitrogen fertilization can induce additional aboveground tree biomass growth on appropriate boreal forest sites (Maynard et al., In press). In a meta-analysis of nitrogen fertilization of jack pine and black spruce stands, Newton and Amponsah (2006) found that most stands exhibited an increase in total volume growth during 10 years post treatment. This effect is usually limited to the first decade of treatment, and even on sites with optimal characteristics and nitrogen treatment, the growth increment can vary widely, that is, from almost zero to an increase of over 30% in merchantable volume growth (Newton and Amponsah 2006). At low doses over the short term (<10 years), there is often a linear relationship between nitrogen addition and incremental growth. At higher doses and over longer time periods, fertilization has reduced benefits for stand growth and may even lead to negative effects relative to the growth of unfertilized stands (Foster and Morrison 2002; Högberg et al. 2006). Fertilization releases tree growth from the constraints of soil nitrogen deficiency; but as the stand grows, other factors can become limiting (i.e., nutrients, moisture, light). In fertilized stands where much of the growth is in the crown, the larger trees will shade out competitors and increase density-dependent self-thinning (i.e., tree mortality will increase) (Weetman et al. 1987; Foster and Morrison 2002; Newton and Amponsah 2006). This has led to recommendations to schedule stand thinning along with fertilization (Mäkipää 1995; Eriksson et al. 2007; Newton and Amponsah 2006) and focus fertilization efforts on moderately stocked stands (Weetman et al. 1987).

The complex effect of nitrogen application on the soil C pool results in poorly understood C dynamics that can reduce the net increase in ecosystem C. In many stands, soil C emissions increase because of an increase in the rate of decomposition and other microbial processes in the soil (Jandl et al. 2007; Allison et al. 2010). Increase in tree litter fall associated with increased aboveground biomass production changes soil chemistry, stimulates decomposition, and can lead to increased emission of nitrogen oxides (Jandl et al. 2007). These dynamics are dependent on forest type and site conditions and shift over time (Jandl et al. 2007): Allison et al. (2010) observed an initial decline in soil C after fertilization followed by an increase. However, most modelling of fertilization effects assumes a small net increase in soil C (Chen et al. 2000; Eriksson et al. 2007; Sathre et al. 2010).

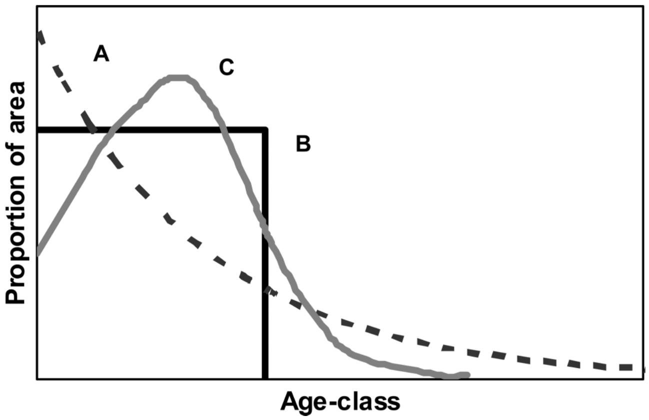
Assessment of the mitigation potential of fertilization also needs to account for indirect emissions from production, transportation, and application of fertilizer and from the volatilization of nitrogen dioxide after application. Fertilizer production is energy intensive, with emissions depending on the type of fertilizer and the production process. Wood and Cowie (2004) estimated emissions from producing primary forest fertilizers to be between 2.0 and 8.8 kg $CO_2$e/kg nitrogen, depending on the type of fertilizer. In comparison, emissions from aerial application of fertilizer are much smaller (Eriksson et al. 2007; Sathre et al. 2010); Sathre et al. (2010) estimated them to be 0.022 kg $CO_2$e/kg fertilizer. Emissions from volatilization of nitrogen dioxide after application have been estimated to range from 0% to 5% of the nitrogen applied (Maljanen et al. 2006; Crutzen et al. 2008; Sathre et al. 2010), although slow-release fertilizer produces lower volatilization emissions (Sonne 2006). How these indirect GHG emissions compare with the ecosystem C balance depends on the time period over which the emissions are calculated, the frequency and dosage of fertilizer application, and the ecosystem response. Seely et al. (2002) found that accounting for these emissions reduced the net increase in C storage (in biomass and wood products) resulting from fertilizing aspen from 13% to 9%. Where fertilization is applied to stands with the highest response, then over the medium to long term the indirect emissions will cause a relatively small reduction in mitigation potential (Chen et al. 2000; Eriksson et al. 2007; Sathre et al. 2010).

#### *3.3.2. Landscape level*

The mitigation benefits of landscape-level C density management (and their dynamics over time) need to be assessed within a systems perspective that includes C stocks in forest ecosystems and HWPs (Kaipainen et al. 2004; Hennigar et al. 2008; Werner et al. 2010) and the changes in emissions resulting from the use of wood instead of more emissions-intensive materials (see section 3.4) (Nabuurs et al. 2007). Landscape-level C density is calculated as the sum of the stand-level C stocks divided by the forest area of the landscape. Landscape-level C density is a function of the age-class structure of the landscape (Kurz and Apps 1999; Böttcher et al. 2008; Kurz et al. 2013). For the purpose of this discussion, three types of age-class structures can be distinguished: left-shifted or negative exponential distributions, "normal" or even-aged distributions, and right-shifted distributions (Fig. 5). Left-shifted or negative exponential distributions result from constant disturbance regimes with an age-independent disturbance probability, such as wildfires in the boreal forest (Van Wagner 1978; Bergeron et al. 2002). Even-aged distributions (sometimes referred to as "normal" although they are theoretical and rarely achieved) are the result of a constant harvest rate of stands that reach a specified age and are not subjected to any natural disturbances so that the forest area is evenly distributed among all age-classes up to the harvest age (Bergeron et al. 2002). Right-shifted age-class structures are a transient phenomenon arising in landscapes in which a period of higher disturbance rates is followed by a period of lower disturbance rates, thus allowing the average age of the stands and C stocks of the landscape to increase (Kurz and Apps 1999; Kurz et al. 2013). As tree longevity is finite and the combined risk of disease and various disturbances increases with stand age, disturbance rates will eventually increase or disturbance types will change. For example, the large-scale outbreak of the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) in British Columbia was in part facilitated by a large area of aging host pine species. When disturbance rates increase, the age-class structure will shift back to the left, and average age and average C density will decrease again (Kurz et al. 2013; Böttcher et al. 2008).

The theoretical maximum landscape-level C storage is achieved when all stands in the landscape are at maximum stand-level C density, which can be at maximum stand age, or earlier in stands with declining yield curves. In Canada's boreal forests, much of the landscape is unlikely to ever reach this theoretical maximum C storage because of frequent stand-replacing natural disturbances such as wildfire and some insect disturbances that result in succession to younger stands with lower C stocks (Kurz et al. 1998). However, quantifying the current age-class distribution of a managed landscape, relative to that which would exist if the landscape were subjected to the prevailing natural and management disturbance regime, allows for the determination of the potential additional C storage that could be achieved in the landscape (Keith et al. 2009). For example, a management unit with a high average forest age that has been subjected to a below-average rate of disturbances in recent years (to decades) will have a high C density but a low potential for additional C storage through management activities. In contrast, a management unit with a high rate of recent disturbances and a below-average forest age will have a lower C density but a higher potential for additional C storage in the coming years to decades. Some landscapes in which disturbances in recent years have occurred much less frequently than in prior years may in fact store more C than can be sustained in the long term because right-shifted age-class structures are transient phenomena that cannot be sustained (Kurz et al. 2013). The design of a landscape-level C management mitigation portfolio thus should take into consideration the

**Fig. 5.** Schematic diagram of three distinct forest age-class distributions: (A) left-shifted or negative exponential, (B) even-aged, and (C) right-shifted.






current age-class structure of management units relative to the average disturbance regime and assess the resulting additional C storage capacity of the management units within the landscape.

Changes in disturbance regimes, whether caused by climate change or human activity, affect the age-class structure and C dynamics of a forested landscape. Increases in the frequency and intensity of disturbances and some changes in disturbance types can reduce landscape-level C stocks during the transition period to the new disturbance regime (Kurz et al. 1998, 2013). Conversely, reductions in disturbance rates will increase C stocks during the transition to the new disturbance regime (Kurz et al. 1998; Peng et al. 2002; Fredeen et al. 2005). Changes in the rate or types of disturbances could result from fire and insect suppression efforts or from reductions in harvest rates to prolong rotations and increase average forest age. The C stock changes resulting from a change in disturbance regime occur during a finite transition period and the landscape will arrive at a new average C stock level that is higher (reduced disturbances) or lower (increased disturbances) than that in the previous landscape. Such transition periods can last many decades and while landscape-level steady-state C stocks are theoretically possible they are rarely, if ever observed, because disturbance regimes typically display high inter-annual and inter-decadal variation.

Stand-level activities, such as fuel management and fuel reduction treatments, can produce stand-level reductions in C stocks that are intended to reduce the landscape-level fire risk. Depending on the intensity of the fuel treatment and the success in reducing area burned, such strategies can increase landscape-level C storage when the increases in C density associated with the higher average age of the forest are larger than the stand-level C losses resulting from the fuel treatment (Hurteau et al. 2008; Hurteau and North 2009). As with other forms of fire or insect suppression, the mitigation potential may be difficult to quantify because the baseline (i.e., C stock changes in the absence of the treatment) cannot easily be established owing to the high and unpredictable inter-annual variability in fire disturbances and the absence of a control landscape.

Analysis of stand-level C dynamics (Fig. 4) indicates that the period of maximum C uptake from the atmosphere (sink strength) and the period of maximum C density (storage) are always separated by many years to decades (Kurz et al. 2013). Landscape-level C management strategies can, therefore, seek to maximize C uptake rates or maximize C density, but they cannot achieve both goals at the same time. A conservation strategy with no or limited harvest is expected to yield landscapes with high C density (but lower uptake rates), whereas a strategy that involves intensive management will yield a forest landscape with a lower C density but a higher C uptake rate (Kurz et al. 1998; Boisvenue et al. 2012; Colombo et al. 2012; Sharma et al. 2013). The difference between the two strategies is the rate at which harvested biomass C is provided to meet society's demands. The implications for climate mitigation depend strongly on both the C storage in harvested wood products and the substitution benefits from wood use (section 3.4). Much debate has focused on the merits of the two strategies. This debate can be informed by studies that take a systems perspective (Nabuurs et al. 2007; Werner et al. 2010) as discussed in section 2.3. Non-mitigation objectives can be relevant in the debate because, for example, older landscapes will provide different wildlife habitat and biodiversity values than younger landscapes (Venier et al., Manuscript in preparation).

### 3.4. Harvested wood products

#### 3.4.1. *Carbon storage*

The C stored in HWPs produced by Canada is large (Apps et al. 1999; NCASI 2007) and each year harvesting results in a substantial transfer of additional ecosystem C to HWPs (Stinson et al. 2011). For the boreal forest, this transfer averaged 17 ± 3 Mt C/year (62 ± 11 Mt $CO_2$/year) in 1990–2008, for a total harvest of 323 Mt C (1184 Mt $CO_2$) in the period (Kurz et al. 2013). Only about 40% of this C has been emitted to the atmosphere so far (Kurz et al. 2013). A simplifying assumption is sometimes made that all C in harvested biomass is emitted (oxidized) in the year of harvest (IPCC 1997). The validity of this instantaneous oxidation approach relies on the assumption that the stock of C stored in HWPs remains constant over time because the additions of new HWP C each year to the stock are balanced by emissions resulting from combustion or decay of HWPs manufactured previously (IPCC 1997). However, in reality, long-term storage of C in some HWPs delays emissions and, rather than being constant, total C storage in HWPs in use (e.g., in houses) or in landfills has been estimated to be increasing both globally (e.g., UNFCCC 2003; Miner and Perez-Garcia 2007; Miner 2010) and in Canada (Apps et al. 1999; NCASI 2007; Chen et al. 2008, 2010). To put it another way, the instantaneous oxidation assumption can substantially overestimate C emissions from HWPs produced in Canada (Dymond 2012; Environment Canada 2013). A reduction in C stock has been estimated in situations in which harvest rates have fallen over an extended period so that emissions from the existing HWP C stock exceed additions of C to the stock (Stockmann et al. 2012). In either case, the simplifying assumption is incorrect. Moreover, this simplification obscures the fact that improving the use of harvested biomass to increase C storage outside forest ecosystems could be a useful mitigation option. Estimating the changes over time in HWP C storage and emissions (e.g., using methodologies in IPCC 2003, 2006; see also Dymond 2012) provides both a more accurate representation of what is actually happening to the C and a better basis for understanding the mitigation potential.

Carbon in HWPs may remain stored for very long periods, depending on the type of product and how it is used and disposed of by society. For example, the default half-lives suggested by the IPCC (2003) for estimating the emissions over time of C in HWPs in use range from 35 years for sawnwood to 2 years for paper. Strategies to increase average storage times are as applicable to HWPs from the boreal zone as they are to HWPs from other regions of Canada, although many of the boreal HWPs are used and disposed of outside the boreal zone. Two possibilities are to use the harvested biomass to manufacture more products that tend to be used over extended periods (long-lived products), thus keeping the HWP C out of the atmosphere longer, and manufacture fewer products like paper that tend to be used over shorter periods (short-lived products). However, HWP production choices would still need to be based on timber supply characteristics and respond to product demand and prices in Canada and abroad. For example, foreign demand for Canadian HWP exports is important for the boreal HWP sector (Bogdanski 2008), just as it is for Canada's forest sector as a whole, and it will have a major influence on the HWP product mix.

Alternatively, the emissions profile of boreal HWP C could be influenced by changing how existing and future products are used and disposed of. A portion of used HWPs is sent to landfill and subject to anaerobic decomposition, resulting in emissions of C as methane. Increasing the rate of recycling and cascading re-use of biomass has been estimated to have mitigation benefits (Skog and Nicholson 2000; NCASI 2007; Werner et al. 2010) (e.g., recycling used lumber for other purposes and then eventually burning it for energy rather than sending it to landfill). NCASI (2007) estimated that recycling of recovered paper in Canada avoided landfill emissions of 17.3 Mt $CO_2$e in 2005.

There is uncertainty about the proportion of the HWP C that decomposes, resulting in emissions, when HWPs are sent to landfill. Studies of landfills around the world have shown a wide range in this proportion (e.g., Bingemer and Crutzen 1987; Micales and Skog 1997; Mann and Spath 2001; Ximenes et al. 2008). Barlaz (2004) estimated the fraction of degradable C in municipal solid waste entering North American landfills to be 44% for wood waste and 39% for paper (although it varied from 20% to 88% for paper, depending on the type and additives). For the purposes of annual GHG inventory reporting, Canada assumes that 50% of the organic C in purpose-built wood-waste landfills, typically operated by wood products mills, will be emitted, whereas 60% of the C in HWPs in municipal landfills will be emitted (Environment Canada 2013). In comparison, 23% has been cited as the fraction of degradable C in HWPs entering US landfills (EPA 2006).

The rate of decomposition of landfilled HWPs is influenced by a number of factors, including the types of products and the proportion of cellulose, hemicellulose, and lignin present in them, environmental factors such as moisture content, pH, landfill temperature, and ambient temperature, and landfill design parameters such as landfill depth. Landfill management practices can reduce GHG emissions from discarded HWPs by altering these environmental factors and design parameters (Pickin et al. 2002; Mohareb et al. 2004). Mitigation can also occur when emitted methane is collected and burned, thereby being converted to $CO_2$, or when it is collected for use as energy (Ayalon et al. 2001; Themelis and Ulloa 2007; Upton et al. 2008). In 2009, approximately 29% of the methane generated in Canadian municipal solid-waste landfills was captured and combusted (either for energy recovery or flared) (Environment Canada 2013). NCASI (2007) estimated that, if 95% of landfills receiving Canadian HWPs had methane collection and combustion, then the long-term emissions of methane would be reduced to the point that they would be essentially offset by the proportion of HWP C that remains in long-term storage in the landfills.

### 3.4.2. *Substitution of wood for energy-intensive products*

It can be complex to analyze the mitigation implications of increasing substitution of wood for emissions-intensive products (Gustavsson and Sathre 2010), but researchers have increasingly investigated the substitution benefits provided by the use of long-lived wood products like lumber and panels on a global scale (Miner 2010), on a national scale in other countries (e.g., Perez-Garcia et al. 2005*a*; Gustavsson and Sathre 2006; Gustavsson et al. 2006*b*; Eriksson et al. 2007; Werner et al. 2010), and nationally or regionally in Canada (NCASI 2007; Hennigar et al. 2008; Liu and Han 2009; Chen et al. 2010). Such studies have concluded that these impacts can be substantial over time. In estimating substitution benefits, researchers have sought to determine the effect of using HWPs in place of other products by comparing the full life cycle of emissions from the two sources, consistent with a systems approach to analyses of mitigation. The life-cycle emissions are determined by eight distinct processes: (1) extraction and transportation of raw materials, (2) primary manufacturing of products, (3) transportation of products to end-use site, (4) final assembly of products, (5) C sequestration in products, (6) C sequestration in landfills, (7) methane release from landfills, and (8) energy reclaimed from combustion of wood waste resulting from the production and disposal of the long-lived HWPs. For example, in a study of building materials used in US residential housing, Perez-Garcia et al. (2005*a*) found that using steel and concrete framing in place of wood-frame building systems resulted in a 26%–31% increase in life-cycle GHG emissions. In a study of four-storey apartment buildings in Sweden and Finland, Gustavsson and Sathre (2006) suggested that using wooden frames instead of concrete frames reduced life-cycle C emissions by 110 kg $CO_2/m^2$ of floor area. When end-of-life management of the apartment building included using the demolition woodwaste for bioenergy, an even greater mitigation benefit could be realized.

Sathre and O'Connor (2010*a*) synthesized data from 21 international studies in a meta-analysis of the net life-cycle GHG emission impacts of substituting wood products for non-wood materials. They calculated an average GHG displacement factor of 2.1, implying a reduction in emissions of 2.1 t C (7.7 t $CO_2$) when a generic wood product containing 1 t of C is substituted for a non-wood product. The meta-analysis showed that the displacement factors ranged between −2.3 and 15, with the majority between 1.0 and 3.0. Sathre and O'Connor (2010*a*) concluded that the negative displacement factors represented worst-case scenarios that are unrealistic in current practice, but the range does indicate that substitution benefits are context sensitive. In particular, estimates are sensitive to assumptions about the characteristics of forest growth, the products being substituted, energy conversion technologies, and the end-of-life management of the wood (Dymond et al. 2010*b*; McKechnie et al. 2011; Sathre and O'Connor 2010*a*, 2010*b*).

Although still subject to uncertainty, estimates of substitution benefits indicate that harvested wood can play an important role in mitigation when it substitutes for products whose production, use, and disposal result in higher GHG emissions. Sathre and O'Connor (2010*a*, 2010*b*) concluded that increasing the substitution of wood for other building materials produces mitigation benefits when forests are sustainably managed and construction wood waste is managed to reduce emissions. NCASI (2007) estimated a 3.7 Mt $CO_2$e substitution benefit (i.e., emission reduction) from the use of building products made of Canadian wood in new housing in Canada and the United States in 2005. A study of residential housing in the United States found that, assuming 1.5 million housing starts per year, 9.6 Mt $CO_2$e emissions would be avoided by using wood-framed building systems in all new housing instead of alternative steel or concrete systems (Upton et al. 2008). However, Eriksson (2003) estimated much higher emission avoidance of 35–50 Mt $CO_2$e if 1.7 million housing starts in Europe used wood framing. The large difference reflects the fact that wood-framed building systems are already used much more commonly in the United States than in Europe so that there is less opportunity for additional substitution in the United States.

The substitution benefits provided by current harvests in the boreal zone have not yet been estimated. The timber harvest in the boreal zone averaged 37.8% of Canada's harvest in 1900–2008 (Stinson et al. 2011; Kurz et al. 2013), and analysis of a geographic dataset of Canada's forest product mills (Global Forest Watch 2011) indicates that the mix of HWPs derived from boreal zone forests was very similar to the average mix from all of Canada's forests. This suggests that the substitution benefit noted earlier in the paper for Canadian wood in 2005 (NCASI 2007) is scalable to the boreal zone harvest, implying a benefit of 1.4 Mt $CO_2$e attributable to the use of boreal wood products in new housing in Canada and the United States in 2005.

There will be trade-offs among the goals of increasing C stored in the ecosystem, increasing storage in HWPs, and maximizing substitution benefits. Research shows that, because substitution benefits are cumulative over successive rotations whereas C storage in ecosystems and HWPs is finite, the importance of substitution increases as the time horizon of mitigation analyses increases (Hennigar et al. 2008; Sathre and O'Connor 2010*a*; Lippke et al. 2011). In a study involving the simulation of landscape-level and HWP C over a 200-year period in New Brunswick, Canada, Hennigar et al. (2008) found that the greatest GHG benefit was obtained by seeking to jointly maximize C storage in the forest, C storage in HWPs, and substitution benefits. This approach was much better than strategies aimed at maximizing either forest or HWP C storage alone.

To achieve mitigation benefits, the production of long-lived HWPs to substitute for more emissions-intensive products in construction would have to be sustainably increased. Such efforts will be influenced by construction standards and practices and the ability to produce the long-lived products needed. In addition, options for increasing the use of wood and reducing or reusing construction waste would need to be examined, forest managers and wood users (including architects and builders) would need to collaborate, and building codes that govern wood use in diverse building types would need to be examined. For example, modern engineered wood products can allow smaller dimension and lower grade lumber to be converted into long-lived products useful for a broader range of construction uses, such as commercial multi-storey buildings or sports arenas.

Alternative HWPs will involve different manufacturing emissions, directly through the use of various fuels or indirectly through the purchase of electricity. A study of the financial, socioeconomic, and environmental attributes of traditional and non-traditional HWPs and production processes provided region-specific assessments of their C life cycles using an assumption that all biomass was C neutral over time because it was sourced from sustainably managed forests (FPAC 2010, 2011; NRCan 2010*a*). The analysis included two regions (near Saguenay – Lac St. Jean, Quebec, and near Thunder Bay, Ontario) that encompass forest in the boreal zone as well as adjacent forest in the hemiboreal subzone. Key findings from the study included (1) GHG emissions vary considerably depending on the product and production process; (2) direct and indirect emissions are driven significantly by the type of fuel and C intensity of the electricity used in product manufacture; (3) solid wood products have the greatest potential for net emission reductions owing to their ability to store C over the long term; and (4) substitution of wood-based products for other more emissions-intensive products significantly reduces emissions. Regional variation largely resulted from differences in the C intensity of provincial electricity generation (e.g., coal-generated electricity in Ontario versus hydroelectricity in Quebec) and feedstock transportation characteristics.

#### *3.4.3. Substitution of wood for fossil fuels*

Biomass can be converted to solid or liquid biofuels or directly combusted to produce heat and power, collectively referred to here as bioenergy. Various conversion technologies are available. Combustion is the most mature and widely used technology to generate heat and power and provides over 97% of bioenergy production worldwide (Zhang et al. 2010*a*). Other conversion technologies such as enzymatic hydrolysis and fermentation, gasification, and pyrolysis involve thermochemical, biochemical, or biological conversions of biomass into concentrated biofuels (Szczodrak and Fiedurek 1996; Evans et al. 2010; Zhang et al. 2010*a*). In all cases, the biofuels produced may be used to generate heat and electricity or further refined into a transportation fuel (Galbraith et al. 2006). Gasification, pyrolysis, and fermentation can also be used to convert a portion of the biomass into value-added biochemicals.

Little research has focused specifically on the mitigation potential of replacing fossil fuels with bioenergy produced from biomass from Canada's boreal zone, but the substantial literature on wood-based bioenergy offers insights that often are as applicable to the boreal zone as they are to other regions. Increasing the use of bioenergy as a mitigation activity is conceptually attractive because bioenergy can substitute for energy derived from fossil fuels. Unlike fossil fuel use, which results in a one-way transfer of C from fossil sources to the atmosphere, the use of biomass for energy emits C but biomass growth removes C from the atmosphere. The concept of C neutrality of bioenergy use has attracted substantial attention, although it can be defined in a number of different ways depending on the purpose and the spatial and temporal boundaries of analysis (Malmsheimer et al. 2011; Miner and Gaudreault 2013). An assumption of C neutrality can be based on the observation that, over time, forest regeneration and C sequestration in a sustainably managed stand will eventually offset the $CO_2$ combustion emissions from burning biomass harvested in that stand if the stand is allowed to return to its pre-harvest C stock level before subsequent harvest (Schlamadinger et al. 1995; Lippke et al. 2011). This assumption does not consider the length of time that is required for sequestration to offset the $CO_2$ combustion emissions. It may take decades to occur during which time the incremental C that has been emitted to the atmosphere contributes to climate forcing. An assumption of C neutrality can also be based on the observation that C removals from growth across a forest landscape will balance the $CO_2$ combustion emissions from burning biomass harvested in the forest if the forest is managed in a way that ensures that its C stock is not decreasing. In some cases, however, continuous production of bioenergy from a forest landscape can reduce landscape-level C stocks (see McKechnie et al. 2011; Holtsmark 2012; Eliasson et al. 2013). Carbon neutrality is sometimes thought to be an assumption used in national GHG inventories because $CO_2$ emissions from burning woody biomass for energy are excluded from estimates of energy emissions in the inventories (the non-$CO_2$ emissions are included). However, this is not because of the use of an assumption of C neutrality. Instead, C contained in harvested material transferred out of forests and used for energy is implicitly assumed to be immediately emitted under methodologies developed by the IPCC (2006, 2013). These emissions are included in estimates of net emissions associated with the forests: it would be double counting to then also include the $CO_2$ emissions from burning woody biomass for energy in estimates of energy emissions.

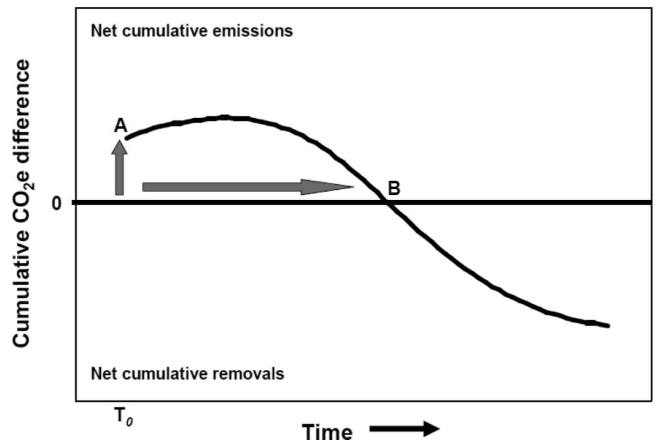
The concept of C neutrality is not directly related to mitigation potential, and bioenergy does not have to be C neutral to contribute to climate change mitigation. It merely has to be better than the baseline energy source it replaces so that it reduces net GHG emissions over a specified time period. Careful delineation of the baseline and appropriate spatial and temporal boundaries of analysis is very important for accurately determining the mitigation potential of bioenergy from a systems perspective, just as in any other mitigation analysis (see section 2.3). The $CO_2$e emissions over time associated with baseline forest and energy use must be compared with those associated with using bioenergy as a substitute (e.g., Schlamadinger et al. 1997; McKechnie et al. 2011). At the stand level, the time at which C neutrality is achieved depends on the rate of stand regeneration. In contrast, estimates of mitigation potential and the time at which a net positive mitigation benefit starts to occur at both the stand and landscape level (the break-even point) depend not only on the rate of stand regeneration but also on assumptions about feedstock sources and their characteristics (e.g., moisture content, calorific content of different tree species and tree components), fuel to energy conversion technologies, and the fossil fuel that is being substituted (Schlamadinger et al. 1997; Galbraith et al. 2006; Raymer 2006; Dymond et al. 2010*b*; Manomet Center for Conservation Sciences 2010; McKechnie et al. 2011; Ter-Mikaelian et al 2011; Zanchi et al. 2012).

Life-cycle assessments, which define spatial, temporal, and production chain boundaries of bioenergy analyses (Davis et al. 2009; Sebastian et al. 2011; Wang et al. 2011), have been used to examine the GHG mitigation potential of biomass as a fossil fuel alternative. Methodologically, the analytical boundaries and assumptions can have a large influence on the results (McKechnie et al. 2011; Lippke et al. 2011). McKechnie et al. (2011) observed that comprehensive evaluations that include detailed assessment of forest C dynamics have not been common in part because of the use of the assumption of biomass C neutrality. To be comprehensive, analyses would need to include the impacts of biomass use on forest C dynamics over time (i.e., they would not ignore the temporal pattern of ecosystem C impacts by using the C neutrality assumption (McKechnie et al. 2011; Ter-Mikaelian et al 2011; Lamers et al. 2013)). They would also need to assess the alternative uses of the woody biomass in the baseline: for example, the woody biomass might not be harvested or it might be used to produce building materials that, through substitution (see earlier in the paper), achieve higher displacement factors than if it were used for bioenergy.

While the importance of applying a comprehensive systems approach to determining the mitigation benefit of substituting bioenergy for fossil fuels has long been recognized (e.g., Schlamadinger et al. 1997), it is only recently that such analyses have become more common. The consequences of applying a comprehensive approach can be most clearly seen at the stand level: the initial impact of bioenergy use on the atmosphere is typically a net increase in $CO_2$ emissions compared with the impact of the alternative (baseline) energy source (Fig. 6). This difference has been referred to as an initial C debt and reflects the fact that the energy density of biomass is typically lower, and in some cases much lower, than that of fossil fuels. Thus, to produce the same amount of energy, larger quantities of biomass $CO_2$ have to be released into the atmosphere. The debt is smallest where biomass substitutes for coal or other fossil fuels with low energy density and it is highest where it substitutes for high-density fossil fuels, such as natural gas. As the forest stand that provided the biomass regrows, the C sequestration will reduce the C debt to the point that net emissions will reach the break-even point with the alternative energy source. From that point on the bioenergy alternative will achieve a mitigation benefit as ongoing removals in the regrowing forest continue to lower $CO_2$ in the atmosphere. While these effects are clear at the stand level, they have also been shown in estimates of the GHG impacts of continuous production of bioenergy that draws on biomass from a managed forest landscape (e.g., McKechnie et al. 2011; Ter-Mikaelian et al 2011; Holtsmark 2012; Zanchi et al. 2012; Lamers et al. 2013).

Most life-cycle assessments of biofuels have focused on agricultural feedstocks (e.g., Davis et al. 2009); whereas life-cycle assessments of electricity generation have included woody biomass, agricultural residues, and energy crops (Froese et al. 2009; Evans et al. 2010; Sebastian et al. 2011; Zhang et al. 2010*b*; McKechnie et al. 2011). Zhang et al. (2010*b*) found that 100% utilization of wood pellets in power generation in Ontario had a very significant mitigation impact, reducing GHG emissions by 91% and 78% relative to baseline coal and natural gas combined cycle systems. However, this analysis used an assumption of C neutrality, and results that incorporate forest C dynamics over time are likely to be different, as discussed earlier in the paper. For example, with an assumption of C neutrality, McKechnie et al. (2011) found that 20% co-firing with pellets from logging residues decreased GHG emissions from Ontario electricity production by 18% over 100 years compared with coal-only operation. When McKechnie et al. (2011) incorporated forest C dynamics (for the hemiboreal and temperate forests of the Ontario Great Lakes - St. Lawrence region that grow faster than boreal forests) in the analysis, they found that a short-term increase in emissions meant that the overall reduction in GHG emissions due to the use of logging residues was 13% rather than 18%.

**Fig. 6.** Schematic diagram of stand-level cumulative net greenhouse gas emissions after wood-based bioenergy is substituted for a baseline fossil fuel at time $T_0$. The curve shows the difference between the baseline and the bioenergy alternative. The initial net emissions (A, the initial debt) depend on the fossil fuel that is being substituted, the efficiency with which the wood and fossil fuel feedstocks are converted to energy, and the fact that the energy density of wood is lower than that of fossil fuels. The length of time required to reach a net positive mitigation benefit (B, the break-even point) depends on what would have happened to the forest carbon in the baseline, the initial debt, and the sequestration rate of the regenerating forest that supplied the wood for bioenergy. The shape of the curve is influenced by the type of feedstock (e.g., harvest residues, whole tree harvest, or salvage harvest) and what would have happened to it in the baseline. Carbon neutrality occurs later than the break-even point and is reached when the forest carbon sequestration offsets the carbon emissions from bioenergy use.




The different biomass-to-energy conversion technologies influence GHG mitigation potential. For example, gasification and pyrolysis used for electricity generation result in lower GHG emissions than direct combustion because the feedstock is used more efficiently (Galbraith et al. 2006). For combustion, the form of the wood (sawdust, pellets, briquettes, etc.) and its moisture content influence conversion efficiency (Raymer 2006) and pelletization of biomass can improve the handling, storage, and energy density of biomass (Stelte et al. 2011*a*, 2011*b*). Conversion technologies need to be assessed not only with respect to GHG emissions but also with respect to other environmental, social, and economic factors (Evans et al. 2010).

The source of feedstock strongly influences the level and timing of net mitigation benefits. Slow growth rates of boreal forests mean that the break-even point can be many decades in the future, especially when tree stem or whole tree harvests are used (Ter-Mikaelian et al. 2011; Bernier and Paré 2013; Holtsmark 2012). For example, in the study discussed earlier in the paper, McKechnie et al. (2011) investigated the break-even point of continuous bioenergy production to displace coal with bioenergy from harvest residues (16 years) and standing tree harvests (38 years) and displace gasoline with bioenergy from residues (74 years) and standing tree harvests (not achieved in the 100 year analysis period). This analysis was for central Ontario forest so the break-event points likely occur earlier than for slower-growing boreal forests. In contrast to such long break-even periods, use of biomass from fast-growing plantations such as willow and poplar that allow for a harvesting cycle of as little as 3 or 4 years (Allen et al. 2011;

Amichev et al. 2012) would result in net mitigation benefits much sooner. Such time-dependent impacts on mitigation potential can be very important in the context of GHG emission reduction targets at specific points in time, such as 2020 or 2050.

Also important for the timing of mitigation benefits is the baseline use of the feedstock if it is not used for bioenergy. Harvesting of actively growing forests that in the baseline would continue to remove C from the atmosphere would lead to longer time to reach the break-even point than if other sources of biomass were used that would otherwise decay, be burned without having their energy captured, or be disposed of in landfills. Examples of other sources of biomass include black liquor from pulp mills, hog fuel from sawmill operations, construction and demolition waste, wood waste diverted from landfills, wood from slash piles containing harvest residues, and in some cases wood removed from forests in fuel treatments designed to reduce fire risks. Burning harvest slash piles at roadsides releases GHGs immediately, so conversion of those residues into bioenergy to replace fossil fuel use produces a rapid, if not immediate, mitigation benefit (because GHG emissions would have occurred anyway in the baseline). If harvest residues are left on site to decompose, they will emit C at a much slower rate than if they are used for bioenergy, so the benefit takes longer to occur if instead of leaving the residues to decompose they are used for bioenergy. Mill and processing residues can be used to produce wood products such as particle board or medium-density fibreboard that store C for years or decades and can have substitution benefits, whereas using these residues for bioenergy results in quick emissions (Dymond 2012).

Dymond et al. (2010*b*) listed three main sources of woody biomass for bioenergy: (1) mill and processing residues (e.g., bark stripped from logs, chip rejects, sawdust, slabs, end-cuts, trimmings, shavings, flour, sander dust, and flawed dimension lumber); (2) residues produced during harvesting, thinning, or silvicultural activities (e.g., tops, branches, and foliage); and (3) deadwood (e.g., standing dead trees resulting from natural disturbances such as insect infestations, fires, and disease outbreaks). Other feedstocks could include urban wood waste (e.g., demolition and construction waste), purpose-grown plantations, and agricultural residues. In addition, the harvesting of whole trees for bioenergy may be of interest in specific areas where energy costs are high.

The major focus of Canadian interest in wood-based bioenergy to date has been on the use of industrial residues and deadwood. There is a positive relationship between the available supply of industrial residues and lumber demand, as mill residues are a by-product of the lumber industry. For example, from 2004 to 2009 lumber demand in North America decreased substantially, resulting in a drop in Canadian mill residues from 21.2 million oven-dried tonnes (odt) in 2004 to 10.9 million odt at the end of the period (Bradley 2010). Although industrial residue availability reflects economic factors, deadwood availability reflects natural disturbance regimes. Climate change is expected to increase tree mortality because of drought, pest infestation, and wildfire events in boreal forests (Price et al. 2013), which will likely increase the quantity of potentially salvageable deadwood feedstock for bioenergy. The mountain pine beetle (*Dendroctonus ponderosae* Hopkins) infestation in central British Columbia since the late 1990s (Safranyik et al. 2010), although not in the boreal zone, provides a good example of this possibility. The infestation created interest in the use of mountain pine beetle salvage material as a potential bioenergy feedstock (Stennes and McBeath 2006; Kumar et al. 2008; Lamers et al. 2013). The potential spread of mountain pine beetle into jack pine (*Pinus banksiana*) in boreal forests (Safranyik et al. 2010) or the emergence of other major insect infestations could provide substantial but uncertain future feedstocks (Dymond et al. 2010*a*, 2010*b*). However, assessments of the mitigation benefit of salvage operations would need to take into account the post-disturbance forest C dynamics in both the baseline case where no salvage occurs and the case where salvage occurs.

Using a system similar to that used to classify differing measures of mitigation potential, Smeets and Faaij (2007) have defined alternative categories of bioenergy feedstock volumes: theoretical potential (the maximum amount biologically available), technical potential (the amount that operationally can be obtained when technological limitations are taken into account, such as limitations on the use of machinery in remote or inaccessible areas), economic potential (the affordable amount given current costs and prices), and ecological potential (the amount that can be removed from the forest without negative impacts on environmental sustainability, such as loss of soil productivity owing to nutrient and biomass removal). Differences among the categories can be substantial. Ralevic et al. (2010) used the Biomass Opportunity Supply Model (BiOS) (Cormier and Ryans 2006) to estimate the biomass available for bioenergy use in three boreal zone sites north of Kapuskasing, Ontario, and compared these estimates with actual field measurements. The model estimated potentially available post-harvest residues to be 49%–65% of the aboveground biomass, but field samples revealed technically available harvest residues to be between 2% and 25%. Operational limitations and cost considerations (a function of the value of the residues for bioenergy or other uses) related to collecting small, low-quality, and dispersed residues constrained the technically available amount.

Most studies for Canada have focused on theoretical estimates of harvesting and mill residues. These estimates have been based on similar roundwood harvesting statistics but have differed in terms of the proportion of aboveground tree biomass that is considered to constitute harvest residues (Dymond et al. 2010*b*). Table 2 (modified from Ralevic et al. 2008) illustrates the range of estimates of woody biomass currently available for energy in Canada. In one of the most detailed studies to date on feedstock potential in Canada, Dymond et al. (2010*b*) estimated both the theoretical and ecological potential from harvest residues and deadwood from fire and insect disturbances in Canada's managed forest, applying a 50% discount factor to the theoretical potential to estimate the ecologically sustainable feedstock potential. These researchers included 215.2 Mha of managed forest south of 60°N. For the portion of this area in the boreal zone, they estimated the ecological potential of harvest residues and deadwood for 2005 and 2020 as 9.0 ± 0.1 and 26 ± 9.0 Tg/year, respectively, or roughly 50% of the Canadian total (C. Dymond, personal communication, 2011). The standard deviation represents the uncertainty associated with annual harvest volumes (e.g., uncertainties associated with policy, sustainability, and economic conditions) and natural disturbance patterns (e.g., uncertainties associated with predicting future forest fires or pest outbreaks).

A key question is the ecological impact of more intensive use of forests for bioenergy, including the effects on hydrology, site productivity, and biodiversity. The issue of the amount of biomass that can be sustainably removed from sites has been of interest for decades in Canada and elsewhere, and research has suggested ways to manage this removal for ecological sustainability, although there remain many gaps in knowledge (Lattimore et al. 2009; Thiffault et al. 2010, 2011; Maynard et al., In press). Lattimore et al. (2009) identified five areas of major environmental concern: soil, water, site productivity, forest biodiversity, and GHG balances. Negative impacts may be a result of organic matter removal or site disturbances (e.g., soil compaction or forest floor scraping) owing to the effects of machines. The long-term sustainability of forest resources is a prerequisite for widespread support and market acceptance of using harvest residues for bioenergy, whether for mitigation or other purposes. Lattimore et al. (2009) and Stupak et al. (2011) explored how existing sustainable forest management programs address forest fuel harvesting and proposed sustainable forest management principles, criteria, indicators, and information for use in forest bioenergy certification systems.

**Table 2.** Estimates of woody biomass in Canada currently available for energy.

| Study | Quantity (M dry t/year) | Energy equivalent (EJ/year)[a] | Percentage of Canada's primary energy (%)[b] |
|---|---|---|---|
| Love (1980)[c] | 72.90 | 1.31 | 8.2 |
| Robinson (1987)[c] | 98.30 | 1.77 | 11.1 |
| Wood and Layzell (2003), low estimate | 51.26 | 0.92 | 5.8 |
| Wood and Layzell (2003), high estimate | 97.13 | 1.75 | 11.0 |
| Mabee et al. (2006)[d] | 20.00–33.00 | 0.36–0.59 | 2.3–3.7 |
| Wetzel et al. (2006) | 76.00 | 1.37 | 8.6 |
| Ralevic et al. (2008) | 97.40 | 1.75 | 11.0 |
| Bradley (2008) | 20.23 | 0.36 | 2.3 |
| Sidders et al. (2008) | 44.26 | 0.80 | 5.0 |
| Dymond et al. (2010*b*)[e] | 71.00 | 1.28 | 8.0 |
| Ralevic and Layzell (2006)[f] | 11.02 | 0.20 | 1.3 |

[a]Assuming energy content of woody biomass is 18 GJ/dry t.

[b]Total primary energy use in Canada in 2010 was 15.95 EJ (Statistics Canada 2012).

[c]These estimates are outdated and combine various categories of biomass (forest and agricultural) but are shown for the purpose of comparison.

[d]This estimate includes forest harvest residues, biomass from naturally disturbed areas, and precommercial thinnings.

[e]A 50% discount factor was applied to net-down theoretically available biomass quantities. This estimate includes clearcut harvesting residues and biomass from naturally disturbed areas.

[f]This is an assessment of the availability of wood killed by the mountain pine beetle over 20 years. It is a temporary resource and, therefore, is not included in the sequential order by year of publication.



Titus et al. (2008) summarized current research on the environmental impacts of forest biomass removal across Canada on the basis of a range of trials with treatment comparisons for over 50 field sites across Canada. More than half of these sites were located in the boreal zone, particularly in Ontario, Quebec, and Newfoundland. The main focus of the research was on the impact of harvesting on soil or stand productivity: trials examine the impact of whole-tree harvesting, whole-tree harvesting with forest floor removal, stem-only harvesting, and various soil compaction treatments, with many of the research sites having been established for more than a decade. Although it is difficult to make generalizations, as the studies examine a range of different tree species (e.g., black spruce, poplar, balsam fir, and jack pine) and geographic sites, the results do emphasize the complexity of soil and stand productivity in the boreal zone. Harvest activities that remove a significant portion of the aboveground slash do negatively affect soil nutrients and microbial activity; however, responses in stand productivity are also affected by factors such as tree species, the mineral content of parent soils, and atmospheric deposition (Belleau et al. 2006; Thiffault et al. 2010, 2011).

### 3.5. Peatland management

Forested peatlands are abundant throughout the Canadian boreal zone (Webster et al., Manuscript in preparation). Western peatlands extend over 36.5 Mha and are concentrated in northern and north-eastern Alberta, north-eastern Manitoba, and along the north-eastern shore of Lake Winnipeg. The majority (over 60%) of these peatlands are fens, and the remainder are bogs (Vitt et al. 2000). Approximately 75% of the bogs are underlain by permafrost (Vitt et al. 2000). Eastern Boreal peatlands occupy 12.5 Mha of the Hudson and James Bay Lowlands area known as the Clay Belt (Lefort et al. 2002). The area is known for its extensive black spruce stands on peatlands, with mixedwood forests interspersed in the better drained sites. The formation of peaty forests is predominantly an autogenous process involving interactions between flat topography, poor drainage, a growing moss layer, and successional dynamics (Webster et al., Manuscript in preparation; Harper et al. 2003).

The forested peatlands of the boreal zone are highly vulnerable to natural and anthropogenic disturbances. Natural processes capable of removing large quantities of C from peatlands include permafrost thaw and deep-burning forest fires (Frolking and Roulet 2007; Turetsky et al. 2010*a*). Direct anthropogenic influences on peatland C in the boreal zone include commercial forestry, petroleum extraction (oil, gas, and coal-bed methane), mining (bitumen, coal, peat, ore, and diamonds), agriculture, major hydrologic construction projects and peat harvesting (Webster et al., Manuscript in preparation; Foote and Krogman 2006), and the drainage and road building associated with these developments. By mid-2009, an estimated 23.7 kha of peatlands in boreal Alberta had been affected by bitumen extraction operations alone (Lee and Cheng 2009). Also as of 2009, approximately 10.5 kha of peatlands had been drained for peat extraction currently or at some point in the past in the boreal zone (seven ecozones), representing about 45% of the peat draining in Canada (adapted from information used in Canada's national GHG inventory report (Environment Canada 2013)).

The GHG effects of these disturbances are both direct and indirect. The direct effect is transfer of C from a long-term solid pool (peat) to the atmosphere owing to forest fire (Flannigan et al. 2009) or physical removal (peat harvesting or surface mining) and subsequent decomposition (Waddington et al. 2002; Cai et al. 2010; Grant et al. 2011). Direct losses of soil C also occur in the dissolved form, as a result of drainage (Waddington et al. 2008) or when water infiltration is impeded on compacted organic soils, leading to runoff and erosion. Secondary effects of disturbances, especially large ones, occur following the removal of the insulating moss layer; exposing the soil leads to permafrost melting, enhanced microbial activity and conversion of peat C to $CO_2$, methane, and nitrous oxide (Turetsky et al. 2010*b*; van Groenigen et al. 2011). In contrast, small-scale forest disturbances in eastern Clay Belt peatlands (e.g., careful logging, low-temperature fires) open the canopy without completely removing the moss layer, leading to a buildup of the *Sphagnum* moss layer and reducing organic matter mineralization and tree productivity (Lefort et al. 2002; Lavoie et al. 2005; Simard et al. 2007). The impact of such processes on the C balance at the landscape level has not been quantified.

Little is known about the potential contribution of peatland management to mitigation in the boreal zone. Mitigation could seek to reduce the GHG impact of disturbances or seek to restore or rehabilitate peatlands after human activity has ended. Forest management practices to reduce disturbance and compaction of organic soils have existed for decades. However, these practices have been developed primarily to ensure post-harvest forest regeneration and little is known about their effect on minimizing peatland C losses. Improved understanding of the paludification process in eastern boreal forested peatlands indicates that silvicultural strategies could be designed to favour C sequestration either in aboveground biomass (through the removal of the moss layer, drainage, and other site preparation techniques that improve tree productivity) or in soils (such as reduced-impact harvesting techniques) (Lavoie et al. 2005; Lafleur et al. 2010*a*, 2010*b*). Choi et al. (2007) suggested that tamarack would respond to peatland drainage more strongly than black spruce through improved water use efficiency and growth. To understand the usefulness of such practices for mitigation, the net effect on C losses and gains in both soils and biomass over time would need to be determined.

The effects of large industrial activities on GHG emissions from boreal zone peatlands could be mitigated in part by limiting hydrological disturbances as much as possible in the surrounding landscape. Nutrient influxes can be reduced by retaining adequate buffer zones around large complex peatland areas when uplands are disturbed through anthropogenic activities. Active restoration after the decommissioning of peat extraction sites can re-establish C sequestration capacity and other ecosystem functions (Lucchese et al. 2010). Peatland restoration requires re-establishment of pre-disturbance water tables and a live cover of peat-forming species such as *Sphagnum* (Schouwenaars 1993). Raising the water table also renews methane dynamics. Post-restoration methane emissions tend to be site-specific and vary with physical conditions (depth of water table, average temperatures) and biological factors (vegetation type, rooting depth, productivity) (Glatzel et al. 2004; Waddington and Day 2007; Mahmood and Strack 2011). However, restoring ecological functions such as peat accumulation is problematic when landscape hydrology and soil chemistry have been deeply altered, such as after surface mining of oil sands (Trites and Bayley 2009). Vitt et al. (2011) reported that, after oil and gas development, it may not be possible to establish peatland plant communities into plant assemblages that resemble natural analogues and have comparable species richness, community structure, and C-sequestering capacity. Hence, novel approaches and new ecosystem research will be needed for rehabilitation of severely disturbed landscapes (in the sense of reinstating some, but not all, characteristics and functions of pre-disturbance landscapes) (Purdy et al. 2005; Johnson and Miyanishi 2008). In all cases, accumulation of pre-disturbance peatland C stocks is not possible given that the initial stocks represent thousands of years of accumulation (Frolking and Roulet 2007).

The impact of climate change itself on the functioning of boreal peatlands (Price et al. 2013; Kurz et al. 2013) is a major consideration in the development of mitigation strategies. Climate-driven permafrost thaw is expected to increase across the boreal zone, compounded by more severe forest fires in the western boreal (Price et al. 2013). Schuur et al. (2009) found that recent permafrost thaw along with enhanced nitrogen availability could stimulate the net primary productivity of moss, resulting in faster rates of peat accumulation that in the short term more than compensate for C losses owing to thawing, in turn resulting in net ecosystem C uptake. However, precipitation in the western boreal zone is predicted to decrease. Concerns that warming and drying will enhance respiration at the expense of photosynthesis in peatlands have not always been verified: in a western boreal treed fen, gradual warming and drying over 5 years was found to interact with vegetation succession and stimulate net ecosystem primary productivity (Flanagan and Syed 2011). It appears therefore that, under a changing climate, forested peatlands could become either a source or a greater sink of C, depending on site characteristics, the disturbance regime, and their interactions with successional dynamics.

Key elements of climate mitigation strategies for boreal peatlands include a long-term perspective, a landscape approach, and integrated ecosystem assessment. A long-term perspective is necessary to avoid investing in strategies whose short-term mitigation benefit could be reversed at a later successional stage (Schuur et al. 2009). Effective strategies should be designed at a landscape level, taking into account the feasibility of manipulating the disturbance regime and incorporating natural processes and management objectives into the design of mitigation action (Le Goff et al. 2010). For example, such an approach could involve facilitating soil C sequestration in topographical lows where paludification occurs naturally, and aiming to optimize biomass productivity in upland sites that are less prone to paludification. Finally, a focus on either biomass or soils alone could be misleading, given the complex interactions that occur between different plant forms, water table levels, soil respiration, and nutrient cycling in peatlands. In particular, integrating the influence of moss layers on peatland hydrology, soil thermal regime, and biogeochemical cycling is likely to be essential to successful mitigation strategies (see the discussion of bryophytes in Kurz et al. (2013)).

## 4. Knowledge gaps

An overall quantitative assessment of the climate change mitigation potential of Canada's boreal forests is not yet possible. A recent blueprint for Canadian forest C science in 2012–2020 identified mitigation as one of four major forest C policy issues requiring scientific attention and proposed increased assessment of the biophysical and economic implications of mitigation options (Canadian Forest Service 2012). Forest-related mitigation activities in the boreal zone will have payoffs that differ in time scale and by location, and future analyses, therefore, should seek to develop a portfolio of such activities that can be considered in the context of national and regional goals for GHG emission reductions. While not discussed in this paper, the economics of mitigation options is likely to have a significant influence on the portfolio.

Mitigation is an important policy response to climate change but so is adaptation. Very little is known currently about the relationship between these responses in the boreal forest zone, making this a key research question as interest in both responses intensifies. As well, mitigation choices must be made in the context of sustainable land and forest management, which is characterized by diverse management objectives of which mitigation would be but one. Thus, where feasible, researchers should attempt to evaluate the consequences of mitigation activities for other environmental, economic, and social objectives (e.g., objectives related to sustainable livelihoods, habitat preservation, and energy security).

The assessment of mitigation actions requires careful determination of baselines to accurately identify the potential. Uncertainties due to climate change add difficulties to what can already be a somewhat subjective process. Future analyses should address the fact that Canada's boreal zone is expected to be substantially affected by climate change. Although examination of mitigation potential in the short term can reasonably ignore the effects of a changing climate, longer term assessments and conclusions about mitigation potential are likely to be compromised if climate change impacts are not factored into the analyses. A key area in which research is needed is examination of the impacts of mitigation strategies under alternative climate scenarios to identify those strategies that may be most robust in terms of their mitigation impact. This is particularly important for strategies aimed at maintaining or increasing forest C stocks.

Although the biophysical potential remains uncertain and the economic potential in many cases remains even more uncertain, the types of forest-related mitigation strategies that may be useful in the boreal zone are increasingly well understood. Also understood are the importance of using a systems approach, careful consideration of analytical boundaries, and the avoidance of simplifying assumptions that can lead to misleading conclusions about how to minimize GHG emissions to the atmosphere. Despite the recent increase in the number of studies on the topic, substantial knowledge gaps remain concerning the potential biogeophysical effects of forest management practices and land-use change on climate at the local, regional, and global scale. Mitigation policy development would thus benefit from an expansion of integrated research into the biogeophysical mechanisms affecting forest–climate interactions and how these ultimately influence the effectiveness of alternative mitigation strategies in the boreal and other zones. Methods that allow comparisons of biogeochemical and biogeophysical effects of mitigation activities using a common metric will be useful for understanding which strategies will minimize climate forcing.

Reducing deforestation could be a key way to reduce GHG emissions relatively quickly. There has been no published assessment of the mitigation potential of reducing deforestation in Canada and such assessments are challenging given the very diverse sectors and proximate causes of deforestation involved. Additional research is needed to better understand specific deforestation drivers, how policy could address them, and leakage implications if deforestation activity shifts elsewhere. Research focused on possibilities to reduce the GHG impact of deforestation when it occurs, for example by changing deforestation practices or making different choices about where deforestation occurs, may be particularly useful.

Overall, afforestation is the mitigation activity that has been most thoroughly analyzed for Canada's boreal zone (and hemiboreal subzone). There is biophysical potential to sequester significant amounts of C over periods of many decades, but the biogeophysical impacts in particular need to be better understood. Afforestation could occur on marginal or other agricultural lands where the biological productivity of new forest could be relatively high, although those lands could also be needed for crop production; in such cases, issues of leakage could be created if the crop production shifted elsewhere. Relatively few studies have examined afforestation on the non-agricultural lands where such competition is less likely to arise. Research is needed to understand the characteristics, costs, and benefits of afforestation designed to maximize climate change mitigation benefits (i.e., taking into account biogeophysical issues) in the boreal zone while minimizing potential for leakage.

The benefits of implementing forest management practices for mitigation purposes in the boreal zone need to be assessed within a systems perspective that includes both C stocks in forest ecosystems and HWPs and the substitution benefits of HWPs. The temporal pattern of benefits arising from alternative strategies is likely to vary depending on the characteristics of the forest, forest management practices, and HWPs. Research from a systems perspective is needed to quantify and compare the impacts of alternatives on the components and how these vary across the boreal zone. It will be particularly valuable to incorporate the possible impacts of climate change on the boreal zone into such analyses because these impacts suggest that long-term efforts to maintain or increase average forest C density in the boreal zone for mitigation purposes are likely to be compromised by increases in natural disturbances.

Increasing C storage in HWPs by increasing the use of long-lived HWPs, reducing emissions by substituting wood for fossil-intensive products, and efficiently using biomass as bioenergy are all viable strategies by which biomass from the Canadian boreal zone can aid GHG mitigation. If wood and paper waste can be diverted from landfills and used to substitute for fossil fuels, additional substitution benefits can be realized through the resultant reduction in landfill methane emissions and extension of landfill operating life. Each year, harvesting results in a substantial transfer of additional ecosystem C to HWPs, and research is needed to improve estimates of the changes over time in HWP C storage and emissions and assess strategies to reduce the emissions including the use of cascading approaches that involve multiple, sequential uses of biomass through recycling. Research is also needed to better quantify the influences on substitution benefits of Canadian HWPs used in Canada and abroad and the potential to increase these substitution benefits.

Little is known about mitigation involving peatland management. Site studies are playing an important role in revealing complex interactions between hydrology, climate, and vegetation in boreal peatlands. Although an essential first step, understanding site-specific interactions is insufficient for the development of mitigation strategies at a scale commensurate with the human interventions on the landscape. A systems perspective emphasizes that mitigation research should adopt a broader perspective and examine the landscape-level potential to support human needs while minimizing net GHG emissions: this is especially true for boreal peatlands, where a broader perspective focused on mitigation at the landscape level over the long term is likely to be fundamentally important.

## 5. Conclusions

The forests and forest biomass of Canada's boreal zone can contribute to mitigating climate change although only some components of the potential have been quantified. A considerable and growing amount of research has focused on forest-related mitigation, providing a sound basis for understanding concepts that are important to mitigation analyses. Foremost is the key realization that the spatial and temporal analytical boundaries of mitigation assessments can profoundly affect the results and conclusions of these assessments. To quantify mitigation potential and identify activities that will have the greatest mitigation impact, a systems approach should be used to assess both the baseline and the mitigation activity. Mitigation analyses using a systems approach should include the impacts of each action on (1) C stored in forest ecosystems, (2) C stored in harvested wood products and landfills, and (3) substitution benefits. Moreover, mitigation analyses should avoid the simplifying assumptions that wood supply for bioenergy is C neutral and harvest material is instantaneously oxidized at the time of harvest because such simplifications can result in unintended mitigation outcomes. Analyses that estimate the actual C stock changes in forests and HWPs and emissions across the relevant sectors (Fig. 3) will lead to effective mitigation portfolios. It can be challenging to use a systems approach for analyses, but such an approach can reveal trade-offs in mitigation effects across the components of the forest system. It can also clarify the possibilities for leakage that reduce the benefit of mitigation activities.

In general, the largest biophysical mitigation potential in the short run will be achieved by avoiding GHG emissions and maintaining C stocks, such as by reducing deforestation, but over the longer run there could be significant potential from activities to increase removals and substitute forest biomass for other more emissions-intensive products and energy. This potential will vary spatially in the boreal zone, depending on differences in the forest, forest management, and the HWPs produced; and the biogeophysical impacts of alternative mitigation strategies remain a significant uncertainty. In addition to the variations in the impact of biophysical mitigation, there will be variations in economic considerations. This review has focused primarily on biophysical mitigation potential; but the most useful analyses for policy discussions will also assess technical and economic potential, factors that constrain the potential, and how those constraints can be overcome.

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