Mitig Adapt Strateg Glob Change
DOI 10.1007/s11027-012-9362-x

Afforestation opportunities when stand productivity is driven by a high risk of natural disturbance: a review of the open lichen woodland in the eastern boreal forest of Canada

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Received: 14 June 2011 / Accepted: 24 January 2012

Abstract Afforestation has the potential to offset the increased emission of atmospheric carbon dioxide and has therefore been proposed as a strategy to mitigate climate change. Here we review the opportunities for carbon (C) offsets through open lichen woodland afforestation in the boreal forest of eastern Canada as a case study, while considering the reversal risks (low productivity, fires, insect outbreaks, changes in land use and the effects of future climate on growth potential as well as on the disturbances regime). Our results suggest that : (1) relatively low growth rate may act as a limiting factor in afforestation projects in which the time available to increase C is driven by natural disturbances; (2) with ongoing climate change, a global increase in natural disturbance rates, mainly fire and spruce budworm outbreaks, may offset any increases in net primary production at the landscape level; (3) the reduction of the albedo versus increase in biomass may negatively affect the net climate forcing; (4) the impermanence of C stock linked to the reversal risks makes this scenario not necessarily cost attractive. More research, notably on the link between fire risk and site productivity, is needed before afforestation can be incorporated into forest management planning to assist climate change mitigation efforts. Therefore, we suggest that conceivable mitigation strategies in the boreal forest will likely have to be directed activities that can reduce emissions and can increase C sinks while minimizing the reversal impacts. Implementation of policies to reduce Greenhouse Gases (GHG) in the boreal forest should consider the biophysical interactions, the different spatial and temporal scales of their benefits, the costs (investment and benefits) and how all these factors are influenced by the site history.

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Keywords Albedo · Black spruce · C balance · Fire frequency · Insect outbreak · Land use change · Reversal risks

1 Introduction

Implementation of policies to reduce greenhouse gases (GHG) is a worldwide challenge because GHG emissions anywhere on the planet affect global climate (Intergovernmental Panel on Climate Change; IPCC 2007). Consequently, many mitigation responses to climate change have been proposed, including land use, land-use change, and forestry policies that increase C sink functions of terrestrial ecosystems (IPCC 2007). Among them, the afforestation encouraged by article 3.3 of the United Nations Framework Convention on Climate Change (UNFCCC) Kyoto Protocol appears to be an elegant strategy to mitigate increased GHG. Afforestation, which is defined as the establishment of new stands of trees where no or few trees have existed in the last 50 years, is expected to mitigate biologically the increase in atmospheric carbon dioxide (CO_2) concentrations at a regional or continental scale over the long-term (IPCC 2003). A real power of mitigation has been shown in afforested temperate and tropical forests because of the high productivity of their soil (Lorenz and Lal 2010). Meanwhile, despite the substantial GHG mitigation potential that the boreal forest has to offer, very little afforestation has been tested and the potential areas for afforestation are still unknown in the boreal forest. The potential for the boreal forest to offset atmospheric CO_2 is of growing interest to national policymakers because boreal forests cover 14.5% of land area, containing 26% of stored carbon (C) in all the ecosystems of the planet and 31% of the amount of stored C in forest soils (Melillo et al. 1993; Dixon et al. 1994; Luyssaert et al. 2007).

Afforestation projects are rarely only a win-win scenario. Findings on C offset can vary greatly between studies due to differences in methodology, project scale, location, climate, soil types, land use and growth rates before and after afforestation, land costs, and C price (Xu 1995; Van Kooten et al. 1995, 1999, 2004; Richards and Stokes 2004; Yemshanov et al. 2005; Laganière et al. 2010). Major shortcomings are also due to the difficulty in modelling ecophysiological interactions like stand productivity versus the exposure to natural disturbance regime. In this context, open Lichen Woodland (LW) stands provide an interesting opportunity to examine the afforestation potential of C offsets within the boreal forest while considering the reversal risks. The LW are characterized by patches of open forest composed mainly of black spruce (*Picea mariana* (Mill.) B.S.P.) and jack pine (*Pinus banksiana* Lamb.), with a canopy cover lower than 25% and where the ground layer cover consists mainly of terrestrial lichens (*Cladonia* spp.) greater than 40%. The persistence of low-density of trees for more than 50 years makes them eligible for afforestation projects in accordance with article 3.3 of the Kyoto Protocol. Across Canada, the LW sparsely covers 2 million km^2 on very well drained soil (Johnson et al. 1999; Fig. 1a). In this review, we will focus on mitigation opportunities in the province of Quebec through LW afforestation as a case study. LW afforestation in the province of Quebec has generated keen interest recently (http://carboneboreal.uqac.ca/english.php) but is supported only by one study to date (Gaboury et al. 2009). Nevertheless, covering more than 1.5 million hectares (ha) in the province of Quebec, afforestation of such areas represents a significant, although theoretical, opportunity for C sequestration. In addition, afforestation of the LW in the province of Quebec represents a particular interest because the natural disturbance dynamic is well documented. This case study is useful to illustrate how interaction between site productivity and recurrent natural disturbances could interfere with the C balance and thus with any afforestation projects in the boreal forest.

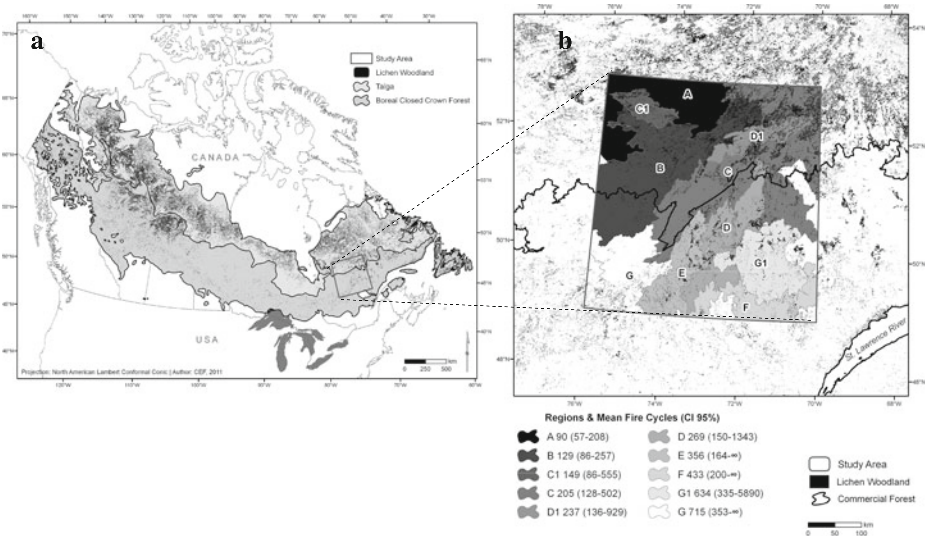

Fig. 1 a Map of the lichen woodland across Canada within the tundra and the closed-crown forest from Latifovic et al. (2004). b Case study of the lichen woodland in the province of Quebec, Canada. The afforested lichen woodlands are located within the managed forest (south of the black line). The square shows the study area used to estimate the fire cycle for the current period (1940-2006) from Mansuy et al. (2010)

The simplistic view describing the boreal forest as a unique C sink has been replaced by models describing the C forest as constantly being renewed by natural disturbances (Amiro et al. 2010; Kasischke et al. 1995; Harden et al. 2000; Taggart and Cross 2009). Wildfires are recognized as the main disturbance affecting the C balance and thus the net primary production (NPP) in boreal forests (Kasischke et al. 2003; French et al. 2004; Conard et al. 2002). Insect epidemics also distinctly alter the boreal forest C balance at a broad scale by killing or defoliating trees (Kurz et al. 2008a, b; Dymond et al. 2010). Boreal ecosystems are, however, also likely to be among the most affected by projected climatic changes (Plummer et al. 2006). Increases in forest productivity and natural disturbance frequencies are being projected for boreal forest ecosystems influencing the NPP (Flannigan et al. 2005a, b; Euskirchen et al. 2006; Kurz et al. 2008b). Therefore, a thorough understanding of C dynamics including growth potential and changes in natural disturbances in response to climate change is required. Also, the integration of the effects of land use changes is needed because the interactions between global change and boreal forest C cycles are complex (Chapin et al. 2008; Bonan 2008). Nevertheless, it remains unclear whether the combined effect of all ecophysiological processes will push these ecosystems into C sinks or sources.

In this context, this review examines mitigation opportunities in the province of Quebec through LW afforestation as a case study. The choice of afforestation of LW in the province of Quebec will help us to understand processes for afforestation where stand productivity is subject to high risk of natural disturbance. We assume that stands characterized by low production and highly vulnerable to natural disturbances are not as attractive for afforestation as strategies to mitigate climate change. The objectives were to: (1) review the existing knowledge on the growth potential of LW by examining its origins and distributions; (2) describe the NPP dynamics following afforestation driven by stand productivity versus the risks in natural disturbances (fire and insect outbreaks) for the present and the future climate; (3) underline the effects of changes in land use including the albedo on the success of afforestation projects; (4) discuss the attractiveness of afforestation in LW and lastly (5) suggest some strategies for managing C stock while minimizing reversal risks. Overall, this paper aims to inform decision makers before they start LW afforestation projects not only in the province of Quebec, but also in other potential low-density stands within the boreal forest. Mitigation strategies presented in this paper could be applied to the C balance management of boreal forests worldwide.

2 Overview of the lichen woodlands: origin, distribution and growth potential

Canadian boreal forests can be subdivided into two distinct zones: the forested-tundra towards the north and the closed crown-forest or taiga towards the south (Rowe 1972; Payette et al. 2000; Fig. 1a). The overall distribution of the vegetation is not homogenous as there are low-density (<25% cover) forest-forming open LW dominated by sparse clumps of black spruce and jack pine. Within the tundra, the wide extent of the open cover is the result of natural limiting climatic and xeric growth conditions, and is also associated with the long-lasting influence of large fires that have shaped the landscape since the Holocene (Payette et al. 1992; Arseneault and Payette 1997; Asselin and Payette 2005). In the province of Quebec, these open stands tend to spread within the commercial forest as shown in Fig. 1b and two main hypotheses have been suggested to explain their origins. Firstly, the distribution of the open stand is the result of reduced post-disturbance regeneration caused by natural and anthropogenic disturbances not necessarily associated with limited climatic conditions (Jasinski and Payette 2005; Payette et al. 2000). Successive disturbances, such as logging followed by fire or spruce budworm *Choristoneura fumiferana* (Clemens) outbreaks then followed by fire (or in the reverse order), occurring before trees reach sexual maturity or replenish the seed bank, may strongly reduce tree density (Payette and Delwaide 2003; Girard et al. 2008, 2009). Secondly, open stands can originate from edaphic constraints related to the thinness, excessive stoniness and xeric drainage of the soil, which would restrict both establishment and growth of tree species. To date, there is no evidence of natural re-densification of LW, e.g. a shift back to a closed-crown forest (Sirois and Payette 1991; Payette et al. 1992; Jasinski and Payette 2005; Girard et al. 2008). Consequently, we can distinguish three types of stable LW that persist over time as illustrated in Table 1: 1) those that have mainly climatic origins (LW1), 2) those related to the lack of seedlings due to successive natural and anthropogenic disturbances (LW2), and 3) those related to edaphic constraints (LW3). However, some stands could be derived from a possible combination of these three limiting factors. For example, many southern LW2 do not have any edaphic constraints while some LW3 could burn frequently (Jasinski and Payette 2005; Asselin and Payette 2005).

As a consequence of deficient postfire regeneration, the LW stands are generally characterized by low biomass and growth rates less than 50 m^3/ha of merchantable volume at maturity. LW stands belong to unproductive forest environments and consequently are not included in the allowable cut in the province of Quebec (Table 1; Pothier and Savard 1998). The ericaceous heath (*Kalmia angustifolia*, *Rhododendron groenlandicum* and *Vaccinium* spp.) and lichens, found in abundance in the LW, could be described as competitive species with black spruce, given their expansive and aggressive root systems (Yamasaki et al. 1998). Also, the reduced fertility of the xeric soil and allelopathic interference with ericaceous shrubs have been identified as factors limiting the growth of spruce seedlings (Mallik 1994; Yamasaki et al. 1998). Drought and water stress are also common causes of seedling mortality in the early seasons following germination (Moss and Hermanutz 2009). To date, the LW1 that originate from limiting climatic factors are not involved in afforestation

Table 1 Three lichen woodland types associated with their origins across the boreal forest in eastern Canada. We can distinguish three essential types of LW by considering their origins and latitudinal distributions: those that have climatic origins (LW1), those related to successive natural and anthropogenic disturbances in a short interval (LW2), and those related to edaphic constraints (LW3). Some LW could be derived from the combination of the three limiting factors, but not necessarily. The area covered by each type of LW is not well documented in eastern Canada or in all of Canada

Type	Potential distribution	Origins	Consequences for growth potential
LW1	Taiga forest, 52° to 56°N Large cover of open forest within the non-managed forest	Climatic origins Recurrent late frost Xeric conditions generated by repeated large fires	Decrease the viable seedlings Production difficult to germinate Very low growth rate
LW2	Closed crown forest, 47° to 52°N Patch of open forest within the managed forest	Successive disturbance origins not necessarily associated with limited climatic conditions Fires + Spruce budworm outbreaks + logging	Poor regeneration if insufficient seed stock before disturbances Absence of good seedbeds Good growth potential
LW3	Closed crown forest, 47° to 52°N Patch of open forest within the managed forest	Strong edaphic constraint origins linked to the thinness and stoniness of the soil and xeric drainage	Seed production but difficult to germinate Poor regeneration Low growth potential

projects because of the fixed nature of the environmental constraints (unfavourable climate and large fires) that have generated these forest types. By contrast, LW2 that are located in the managed forest do not have edaphic constraints per se and could be possibly re-established by plantation. Studies by Thiffault et al. (2004) and LeBel et al. (2008) have confirmed potential tree growth increases following soil scarification aimed at reducing ericaceous competition and increasing mineralization rates. However, the growth of seedlings in scarified areas only meets the lowest yield curves observed in the boreal forest of Quebec (Thiffault and Jobidon 2006). Côté (2004) showed that where the canopy opening resulted from a SWB outbreak followed by fire, survival trees had a similar growth rate to the adjacent closed-crown forest. On LW3 sites, the positive effect of scarification on water availability does not solve the nutritional deficit of poor soil (Hébert et al. 2006). Given the low nutritional status associated with ericaceous heath, adding fertilizer to improve early seedling growth has not proven effective in the long-term (Thiffault et al. 2004; Thiffault and Jobidon 2006). In addition, several studies have already reported that afforestation scenarios with low soil productivity such as LW3 (low availability of nutrients, xeric drainage) are less suitable for storage of C (Marland and Schlamadinger 1997). These scenarios are also the least economically attractive because they require an expensive investment in site preparation.

To summarize, the site history such as the origins, dynamics and growth potential of LW need to be taken into consideration in order to make a proper decision about re-establishment. In our case, only stands where low-density forest is the result of successive disturbances, and not of either edaphic or climatic constraints, may offer a potential for the re-establishment of a productive stand. In that respect, the rest of the paper focuses on the hypothetical afforestation of this type of open woodland.

3 C dynamics following afforestation under climate change

The boreal forest is likely to be especially affected by climate change because of the forest's sensitivity to high temperatures (IPCC 2007). Therefore quantifying the effect of afforestation projects in C offset in the boreal forest requires not only the understanding of C dynamics, but also its response to climate change. The NPP is the major conduit of C transfer from the atmosphere to the land surface and is thus a fundamental component of the global C cycle (Kang et al. 2006). As illustrated in Fig. 2, we will focus here on two main issues affecting the NPP: changes in forest productivity and changes in natural disturbance regimes.

3.1 Growth potential as the main input

Because the growth potential of a stand is the primary variable used for projecting C sequestration (Boyland 2006), a systematic examination of the yield curves should be carried out before use in a C model. Compared with tropical or temperate forests, where afforestation is often a success, NPP in the boreal forest is considerably lower because of the low growing degree-days and the nutrient-poor soil conditions characteristic of the high latitude biome (Luyssaert et al. 2007; Gower et al. 2001; Magnani et al. 2007; Fig. 3a). A relatively low NPP may act as a limiting factor in an afforestation project in which the time available to

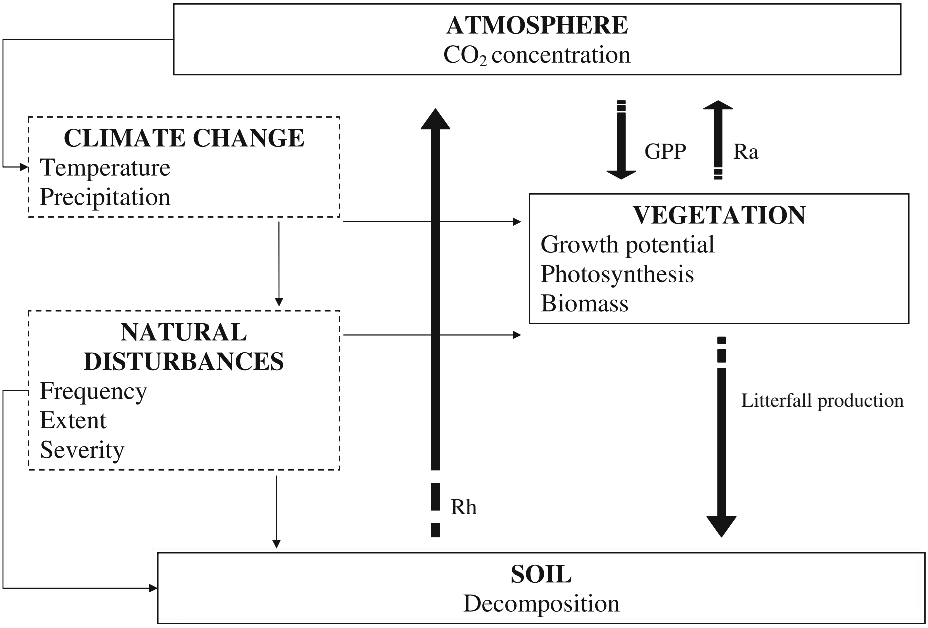

Fig. 2 Simplified conceptual model of carbon dynamics in the boreal forest under climate change. Plants absorb CO_2 from the atmosphere and build biomass through photosynthesis (gross primary production; GPP). In the boreal forest, 50 to 70% of the GPP is re-emitted as CO_2 by autotrophic respiration (Ra), which is a source of C related to growth and maintenance of plant tissue. The long-term C accumulation rates in boreal soils represent the largest C pool in the boreal forest. Soil heterotrophic respiration (Rh) is also a C source in the atmosphere, via dead organic matter decomposition. The difference between GPP and the C emitted by Ra + Rh is the net primary production (NPP). How climate change would affect the NPP via the increase in natural disturbances versus growth rate is uncertain

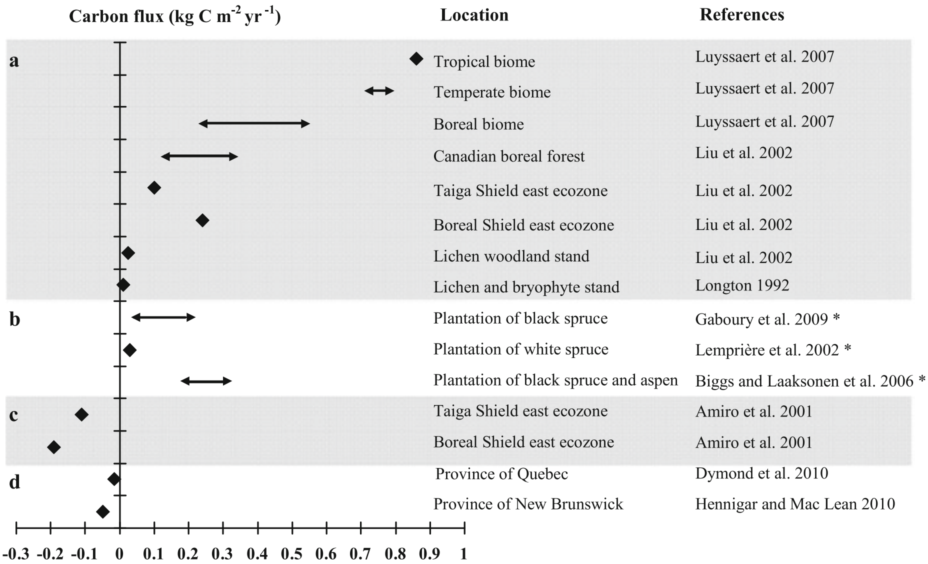

Fig. 3 **a** Net primary production for different forest regions compared with **b** Net primary production following afforestation in the boreal forest of Canada, **c** carbon release during fire for eastern Canadian ecozones and **d** carbon release during spruce budworm outbreaks in eastern Canada. For comparison purposes, all the values have been converted into the same unit (kg Cm^{-2} year $^{-1}$). Values >0 mean carbon sinks, values <0 mean carbon sources.* Values are discussed in the text

increase the C is driven by natural disturbances. Furthermore, NPP for the LW stands is even lower, between 0.02 and 0.1 kgCm^{-2} y^{-1} (Fig. 3a), because of its particularly low densities of trees (<25%) and the resulting low biomass and also because of the low nutritional status in some cases. Meanwhile, the NPP of the understory, fine root production and bryophytes could be considerably underestimated (Gower et al. 2001). As illustrated in Fig. 4a, wood biomass in black spruce stands varies considerably with the stand age and also as a function of site index. In the province of Quebec, the LW corresponds to a site index lower than to the lower site index (IS9; Fig. 4a) and therefore afforestation is expected to increase the NPP at the stand scale, through the input of standing biomass.

To date the only study that modeled net C sequestration following afforestation of LW has estimated the biological C balance at 77 tC ha^{-1} after 70 years following a hypothetical reforestation of 2000 black spruce trees (Gaboury et al. 2009). Depending on the lower and higher plantation yields simulated, the sequestration rate varies from 0.2 to 1.9 tC ha^{-1} y^{-1}. These results are similar to the few studies carried out in boreal forests across Canada (Fig. 3b). Lemprière et al. (2002) projected a net increase of 0.3 tC ha^{-1} y^{-1} over 50 years following a reforestation of white spruce (*Picea glauca*) on low-density aspen (*Populus tremuloides*) stands in boreal Saskatchewan. In northeastern Ontario, Biggs and Laaksonen-Craig (2006) estimated a sequestration of 2.4 to 3.0 tC ha^{-1} y^{-1} over 50 years following reforestation of jack pine and aspen. The wide range of responses observed in Gaboury et al. (2009) and Biggs and Laaksonen-Craig (2006) underlines the importance of the choice of the quality site index (depending on whether low or high productivity sites were assumed) used to project C sequestration. In our case study, the hypothetical growth rate of 175 m^{-3} ha^{-1} at 70 years compared with the baseline scenario of 30 m^{-3} ha^{-1} used in Gaboury et al. (2009) may be questionable. In fact, if generally the previous stand can give good information on yield for

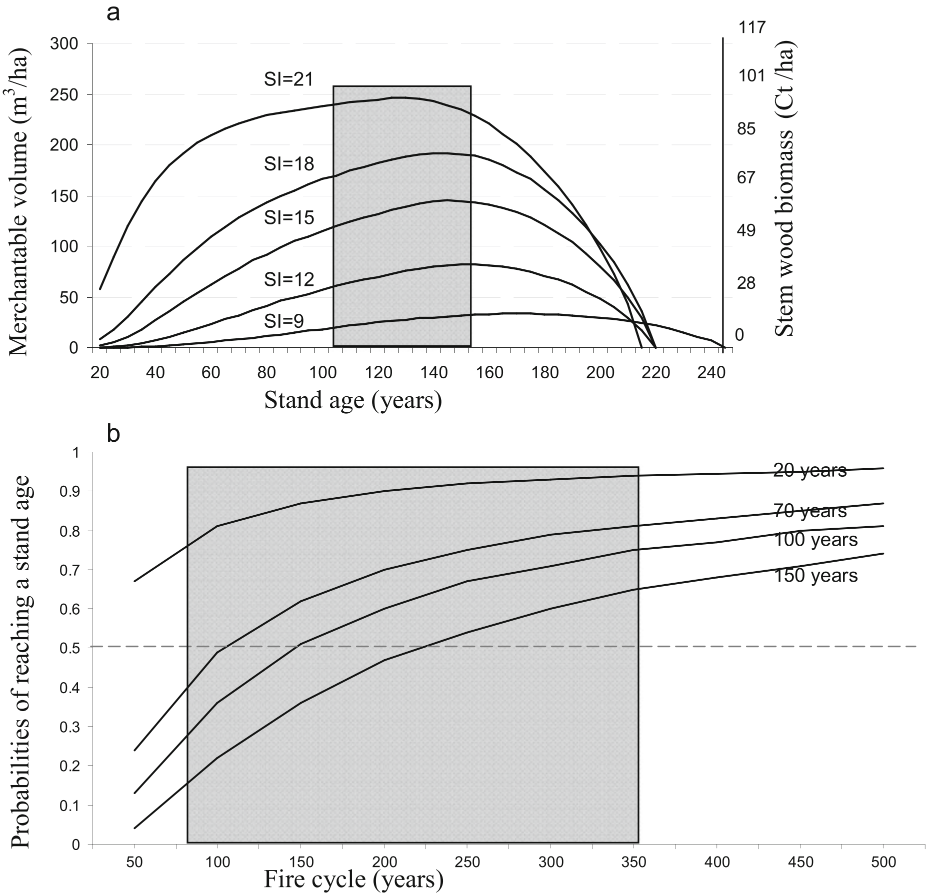

Fig. 4 **a** Volume of merchantable wood and wood biomass for high density (2,500 trees/ha; diameter at breast height > 9 cm) black spruce stands for different site indices (SI) in province of Quebec from (Pothier and Savard 1998).The stem wood biomass (left vertical axis) is estimated by $bm = a \times volume^b$ where a=1.22 and b =0.8 (from Boudewyn et al. 2007). Stem wood biomass of merchantable-sized trees accounts for the major proportion of biomass found in trees and can be thought of as the basic or primary model as all subsequent models are based on its output (Boudewyn et al. 2007). The grey square represents the time period when the volume is maximum. **b** Probabilities of reaching different ages (20, 70, 100 and 150 years) over different fire cycle lengths. The fire cycle is defined as the time necessary to burn an area equivalent to the study area. The probability of reaching maturity is derived from the negative exponential distribution $P(A) = \exp(-A/T_f)$ where A is the age considered and T_f is the fire cycle (Johnson and Gutsell 1994). In order to illustrate the potential higher fire risk, the grey square represents the lower confidence interval for fire cycle (86 to 353 years) observed in the study area in the managed forest (Fig. 1b)

afforestation, this is less obvious for the LW2. Forest growth on LW2 sites has not been tracked traditionally within operational forest inventory programs and we are left with a significant uncertainty as to what the growth trajectories of trees planted in LW2 sites are (Pothier and Savard 1998). In addition, the act of planting trees is not exclusively accompanied by C gain. During the first 20 to 30 years after the establishment of a new plantation, the stand NPP is negative because soil disturbances (such as scarification) increase C loss through heterotrophic respiration (Smith et al. 2006; Gaboury et al. 2009; Tremblay et al. 2006; Laganière et al. 2010). Afterwards, the NPP will increase for a number of years once the canopy closes and stabilize until the next onset of disturbances.

Because it takes decades to observe a net C increase, projections for growth potential in the future are needed but are also uncertain. Higher temperatures and longer growing seasons are expected to improve plant productivity and increase plant litter input and soil pools (Keeling and Phillips 2007). Boreal forest soil could possibly stock more C in warmer conditions because an increase in forest productivity will increase the amount of C input to the soil (Rustad et al. 2001). However, boreal soil that contains the largest amount of C is highly vulnerable to warming. A prolonged increase in temperature is also expected to increase soil respiration and thus C loss (Heimann and Reichstein 2008). Boreal trees and aboveground plant productivity should increase because nutrient supply increases also with temperature (Litton and Giardina 2008), but increased sequestration of C under elevated CO_2 concentrations may have little significance because over 70% of the GPP is respired (Trumbore 2006). Thus, an increase in the photosynthetic rate could have a minor effect on the NPP. Black spruce seedlings could be more susceptible to episodic soil drying and become less competitive for belowground resources (Way and Sage 2008). Deficits in the water balance, caused by extreme summer droughts, could overlap the C uptake because an increase in respiration will diminish the benefits of the increasing spring NPP (Angert et al. 2005). Consequently, despite the absence of a theoretical consensus on the response of boreal forest growth to climate change, a warming climate may not necessarily lead to higher CO_2 growing-season uptake, even in boreal ecosystems that are considered to be temperature limited (Lorenz and Lal 2010).

3.2 Effects of natural disturbances on C budget

By consuming NPP, fire and insect plays an important role in the inter-annual variability of source/sink relationships in the boreal ecosystem (Fig. 3c and d). In the Canadian managed forests, fires released 23±16 Tg C yr^{-1} directly into the atmosphere, and fires, insects and other natural disturbances transferred 52±41 Tg C yr^{-1} from biomass to dead organic matter pools, from where C will gradually be released through decomposition between 1990 and 2008 (Stinson et al. 2011). Because the amount of C released during fires or insect outbreaks is to a great extent dependent on where C is concentrated, managers should be very careful about spotting the sites to be reforested. In order to increase C stock within a minimum of time, afforestation scenarios in the boreal forest should definitely target stands where high productivity has been regularly tracked and proven, and simultaneously where vulnerabilities to natural disturbances are the lowest.

3.2.1 Changes in fire frequency and area burned

Fire frequency strongly affects the pattern of forest succession, which in turn controls forest growth and thus the storage of C in new plantations (Litvak et al. 2003; Bonan and Shugart 1989; Weber and Flannigan 1997). As fire frequency is driven by climatic variables, some regions are more prone to fire than others and even a low fire risk region could experience some hot spots of fire risk because very dry surficial deposits are prone to burn (Mansuy et al. 2010). For example, mean fire cycle varies between 129 and 715 years in the area where the LW2 are afforested in the province of Quebec (Fig. 4b). However, if we look at the lower confident interval, the fire cycle may vary between 86 and 353 years only. Therefore, the spatial and temporal variability of fire should be accounted for in afforestation scenarios because the probability of reaching maturity in boreal forest is strongly driven by fire frequency. In a dense black spruce forest, the wood biomass increases depending on the site index to reach a maximum between 100 and 150 years before decreasing to stand senescence (Fig. 4a). So, under a fire cycle less than 100 years, the probability of reaching 100 and 150 years old (when the maximum volume stand is assumed) is lower than 0.35 and 0.20; the probability of reaching 70 years is lower than 0.50 and of reaching 20 years (generally minimum age when a plantation starts to act as a net C sink) is much higher between 0.60 and 0.80 (Fig. 4b). Fire cycles longer than 200 years seem not to be a constraint in reaching 70 and 100 years (probabilities > 0.5) but is one for reaching 150 years (probabilities < 0.5). Moreover, even at maturity the wood biomass can be particularly low for the less productive sites (SI9 and SI12; Fig. 4b). The poorest sites reach maturity later and their low capabilities to increase C stock are therefore exposed to the fire risk for a longer period.

In addition, fires that have generated the LW2 are expected to be more critical in the future because higher temperatures may increase fire frequency and geographical expansion of severe fires in the boreal biomes (Plummer et al. 2006; Amiro et al. 2010). Consequently, in an area submitted to a particular fire cycle, the probability of new plantations reaching maturity within coniferous forests could be lower for the future decades than for the current period. As a result, any increase in fire frequency in the future will tend to lower long-term NPP and C storage (Thornley and Cannell 2004). In eastern Canada, the Fire Weather Index is expected to decrease in the southern boreal forest but increase in the northern boreal forest (Girardin and Wotton 2009; Bergeron et al. 2010). Depending on the area, simulations done for $2XCO_2$ and $3XCO_2$ scenarios across the boreal forest of the province of Quebec suggest an increase in fire frequency but also in extent with a longer fire season sustained by drier combustibles (Le Goff et al. 2009; Flannigan et al. 2005a, b; Girardin et al. 2010; Metsaranta et al. 2010). However, these results should be interpreted with caution because they are hardly representative of a particular locality. As a result, C stored in the new plantation would likely be released into the atmosphere and the desired effect of the atmospheric CO_2 offset would possibly be neutral. This is particularly true for the less productive sites because they are more exposed to fire risks. Moreover, the unpredictable extent and the extreme inter-annual variability of the area burned makes also the success of afforestation on the richest sites uncertain.

3.2.2 Changes in the distribution of spruce budworm outbreaks

Recovery of strong C sinks in Canada’s managed forest will be delayed until at least the 2030s because of insect outbreaks, even if predicted increases in area annually burned do not occur (Metsaranta et al. 2010). Therefore, taking spruce budworm (SBW) into consideration in LW2 afforestation projects in the province of Quebec is justified by the geographical extent and severity of the expected outbreaks (Dymond et al. 2010) and also by the fact that some of the LW2 were the product of past intense SBW outbreaks (Simard and Payette 2001; Fleming et al. 2002). The SBW causes considerably more damage than any other insect in eastern Canadian boreal forests (Volney and Fleming 2000). Defoliation reduces growth and survival of balsam fir (*Abies balsamea*) and spruce by up to 90% (Erdle and MacLean 1999). Although the preferred host of the SBW is balsam fir, recent studies have shown a large decrease in the growth of black spruce, a preferred species for plantation programs in the boreal forest (Simard and Payette 2005). Studies show that recent warming episodes might accelerate the seasonal development of some insects, and might also cause catastrophic changes in the spread of outbreaks, thereby influencing boreal forest C dynamics on a broad scale. Using CBM-CFS3 (Carbon Budget Model of the Canadian Forest Sector), Dymond et al. (2010) simulated moderate to severe SBW defoliation from 2011 to 2024 across susceptible areas in eastern Quebec. They suggested that similar outbreaks to the last one could make a forest switch from a C sink to a source in the near future (Fig. 3d). Hennigar and MacLean (2010) obtained similar conclusions regarding the next outbreak in New Brunswick (Fig. 3d). In addition to forest fire, future outbreaks could possibly reduce photosynthesis and thus limit C offset in sites afforested with black spruce or in proximity to fir stands.

4 Effect of land use changes on C dynamics

The Good Practice Guidance for Land Use, Land-Use Change and Forestry (IPCC 2003) recommends the inclusion of C emissions from forest operations induced by the afforestation project. Many studies concluded that C emissions arising from afforestation-related operations (seed farms, seedling growth and maintenance, current harvesting operations, site preparation, planting, forestry road construction, housing and accommodation, monitoring, and transportation) are generally below the rate of C sequestered by plantations (Karjalainen and Asikainen 1996; Berg and Karjalainen 2003; Johnson et al. 2005; Gaboury et al. 2009). However, transportation from the forest to the factory (including road construction to access the resources) is the activity that emits the most C. Consequently, afforestation of LW2 in the commercial forest area, where stands are easily accessible by an existing road network, should not lead to a loss of C. Conversely, projects that seek to reforest remote areas where infrastructures are poorly developed should include the costs of C emissions from road construction and transport.

Changes in albedo must also be taken into account in order to assess the global C sequestration benefits following afforestation (Jackson et al. 2008; Schwaiger and Bird 2010). A change in land use has a direct effect on solar reflectivity, which in turn influences the local radiation budget. Radiative forcing is a measure of how the energy balance of the Earth-atmosphere system is influenced when factors that affect climate are altered. A boreal forested landscape generally has a lower surface albedo than open land, particularly in conditions of snow on the ground when short-wave radiation is trapped by multiple reflections within the forest canopy (Betts et al. 2007). A shift from yellow lichen ground cover to evergreen black spruce or pine canopy offers a dramatic contrast in reflectance. As a result, planting forests could help stabilize global atmospheric CO_2 but may accelerate climate warming regionally, further speeding the loss of snow and ice cover (Betts 2000; Randerson et al. 2006). Of particular interest to our case study, Bernier et al. (2011) showed that the conversion of lichen woodland to closed-crown forest in Quebec may generate a warming effect to the atmosphere. Considering that the projected future climate change has been linked to increased fire activity, the generation and maintenance of LW would provide possible negative feedback to climate change (Bernier et al. 2011). However, how the increase in biomass versus the reduction of the lichen cover may affect the net climate forcing of the plantations is not known.

5 Lichen woodland afforestation attractiveness

Considering all the cumulative factors reviewed above, it becomes rather difficult to accurately validate atmospheric CO_2 offset following LW afforestation and its permanence over time. The potential net C mitigation following afforestation in the boreal forest is subject to multiple reversal risks, i.e. the intentional or unintentional release of C back into the atmosphere due to fire, pests, land use decisions, and many other factors (Galik and Jackson 2009). This is particularly true for the LW2 stands. Indeed, the theoretical LW2 growth rate to model C sequestration used in Gaboury et al. (2009) may be overestimated. Besides, the LW2 reforested over the past 10 years are still not included in the calculation of allowable cut because the potential growth has not been proven yet (Paillé et al. 2007). However, increases in NPP at the landscape level would not be significant compared with C losses from global increased natural disturbance rates (Kurz et al. 2008b). Under warmer conditions, an increase in the growth potential in the boreal forest will not necessarily enhance the NPP due to the increase in soil and tree respiration. In particular, the productivity for dry conifer areas such as LW could be reduced by more frequent fire activity (Kang et al. 2006). The risks of successive disturbances (fire+SBW) could interfere more frequently and to a wider extent than in the past, and shift new plantations back to open woodlands, releasing the C they had stored up until that date. Moreover, the expected benefit of afforestation of LW2 on climate could be neutral or even negative, if we consider the decrease in albedo related to land use change (Bernier et al. 2011).

Given the impermanence of C stock linked to the reversal risks, it is relevant to question the economic issues regarding the LW2 afforestation. If we consider only the current C price around 9$CAD/t (www.pointcarbon.com, December 2011) and an average net sequestration of 0.2 to 1.9 tC $ha^{-1}y^{-1}$ estimated for 2000 seedlings of black spruce in Gaboury et al. (2009), 1 ha of LW plantation could yield 1.8 to 17.1$CAD y^{-1}. The basic costs of forestry operations necessary for re-establishing the production of LW2 reviewed in Table 2 compared with the current low price of C makes this scenario not cost-attractive. Moreover, to date much of the basic information required for economic analyses for LW afforestation: yield, biomass/C ratio, establishment costs, land opportunity costs, potential sales of harvest

Table 2 Basic costs relative to the lichen woodland afforestation project in eastern Canada

Operations	Average cost[1]	
	Estimates ($/ha)	Planning and monitoring ($/ha/year)
Land cost	0	0
Scarification	500	27
Purchase of 2000 plants[2]	600	42
Fertilization	459	27
Total[3]	1 559	96
Road construction[4]	20 000 to 100 000 $/km	Nd

[1] MRNFQ (2010) in CAD$

[2] Planting of 1 ha includes about 2000 plants

[3] The values are given as indicative because the price of a plantation varies depending on the region and silvicultural treatments required. Van Kooten et al. (1999) identify an average cost of $1,500/ha

[4] Forest road construction could eventually be included considering the remoteness of certain lichen woodland stands. According to Desautels et al. (2009), costs related to forest road construction vary greatly depending on the proximity of raw materials (gravel) as well as different work requirements (excavation, grubbing, etc.). Nd means no data

wood as well as the natural disturbance regime include significant levels of uncertainties that make it difficult to draw conclusions. Current limitations in our ability to map site suitability also affect the economic issues. Therefore, the current low profitability of LW2 afforestation could represent a serious obstacle to large-scale afforestation projects in the boreal forest. A retrospective of the pan-Canadian afforestation initiative reveals that even if fast-growing species (hybrid poplar on private croplands) can provide a range of C benefits, their returns are generally not high enough to induce large-scale investments (Dominy et al. 2010). Otherwise, Van Kooten et al. (2004) highlight the influence of discount rates on the time value of C and the marginal value of afforestation. The time value of C depends on the path of marginal damage; in other words, the value of mitigating GHG emissions today depends on when the damage being avoided is expected to occur and how much it will cost. Also, ignoring biophysical interactions (such as changes in albedo) could result in millions of dollars being invested in a mitigation project that provides few climate benefits or, worse, is counter-productive (Jackson et al. 2008).

6 Lessons learned and strategies for managing C stock in the boreal forest

To respond to the growing interest for the boreal biome to offset atmospheric CO_2, some useful strategies can be suggested in order to best manage C stock, while simultaneously minimizing reversal risks. Mitigation strategies of forest around the world have been widely documented (Nabburs et al. 2007). Here we briefly discuss activities in the boreal forest that could reduce emissions and those that could increase removals. These strategies could be incorporated into forest management planning to assist in climate change mitigation efforts.

Given the uncertainties raised in this paper, we suggest that mitigation opportunities through afforestation in the boreal forest may be determined by site history, mainly successional stage and site productivity linked to the natural disturbances regime. Afforestation scenarios in the boreal forest should only target stands where high productivity has been regularly tracked and proven, and also where vulnerabilities to natural disturbance are the lowest as shown in Fig. 5. Nevertheless, even a highly productive afforested stand with low vulnerabilities to natural disturbance (considered as the best opportunity to increase C stock) may be not successful because a large proportion of the forest could be hit by a small percentage of the number of events (Cooley et al. 2012). This means that just one single large fire event could ruin decades of investment. Since this type of catastrophic events are expected to be more frequent in the future, an alternative to excluding stands vulnerable to fire would be to include a fire risk threshold in the planning phase as suggested by Savage et al. (2010) and Raulier et al. (submitted). Despite the unpredictability of the natural disturbance events, the growing literature about natural disturbance vulnerability mapping in the boreal forest should be included in C management. Pooling diverse plantations or careful geographical diversification seem to be good approaches to minimizing the impact disturbance on C stock (Laurikka and Springer 2003; Hultman 2006). More research is needed before including afforestation as mitigation policy in the boreal forest.

Reducing deforestation is considered among the most efficient mitigation strategies around the world (Nabburs et al. 2007; Miles and Kapos 2008), but little research has been done on its impact on C in the boreal forest. Meanwhile deforestation affected only 0.02% (56 000 ha) of Canada's forests in 2005 (Natural Resources Canada 2008) and may not significantly affect the global C forest budget at the country-wide scale.

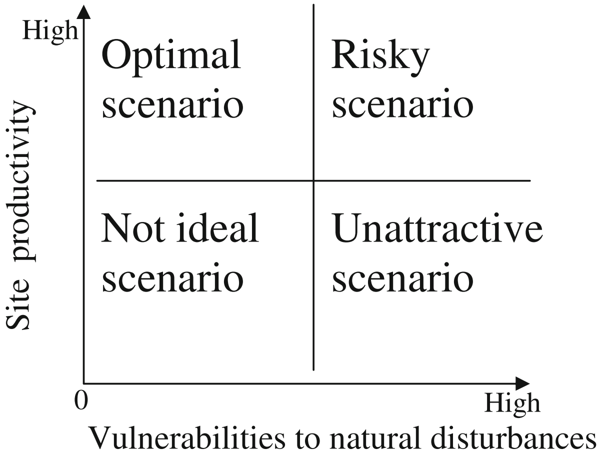

Fig. 5 Concept of afforestation opportunities in the boreal forest based on the ratio of site productivity versus vulnerabilities to natural disturbances. A potential highly productive site with low vulnerabilities to natural disturbance may represent the optimal scenario for afforestation in the boreal forest. Conversely, a low productivity site with high vulnerabilities to natural disturbance represents an unattractive scenario. Because the growth potential of a stand is the primary input used for projecting C sequestration, a low productivity site is not an ideal scenario even if vulnerabilities to natural disturbances are low. Moreover, the unpredictable extended and the extreme inter-annual variability of the area burned also makes the success of afforestation on the highly productive sites risky

However, some particular location (construction of a hydroelectric dam or mining, for example) could experience much higher rates and thus need more consideration.

At the landscape scale, mitigation strategies could involve lengthening harvesting rotation, reducing forest fragmentation, peatland management, enhanced fire and insect protection. However, fire suppression as a mitigation strategy is not so obvious because of the extent and the inaccessibility of the forests as well as the characteristics of the disturbances. Only 3% of large fires contribute to 97% of the area burned in Canada (Stocks 1991). In addition, an increase in C stock due to fire suppression represents much accumulated fuel, which over time can lead to catastrophic C emission under particularly dry conditions (Tilman et al. 2000). At the stand level, a number of practices exist to increase the C density (mass per hectare) such as increasing fertilization, improved regeneration with genetically modified trees, intensifying silviculture, promoting thinning cut instead of clear cut, avoidance of slash burning and postfire logging. In other words, efforts could be made to maximize C densities with stands characterized by low disturbance risks (human or natural) and conversely, efforts could be made to maintain low C stocks with stands characterized by high disturbance risks.

Considering the potential conflicts arising between forest stands, natural disturbances, land use changes, and C management, policymakers should ensure that forest offset strategies do not provide singular incentives to maximize stocking levels at the expense of risk management. Instead of providing credit only for the amount of C stored on-site, an alternative approach could also award credits for activities that reduce reversal risks, such as fuel reduction activities, to address wildfire risk in fire-prone areas (Hurteau et al. 2008, 2009; Galik and Jackson 2009) or pest management in outbreak-prone areas (Slaney et al. 2009). Encouragement of forest sector activities could also provide climate benefits by substituting fossil fuels with bioenergy, or indirectly by using wood products in place of more greenhouse gas-intensive materials like steel, aluminum and concrete (Schlamadinger and Marland 1996).

Ultimately, managers should monitor the periodic estimates of expected C sequestration versus loss, allowing adjustments for achieving quantifiable objectives in order to be accounted for in the national GHG inventories. Implementation of policies to reduce GHG in the boreal forest should consider the different spatial and temporal scales of their impacts, the costs (investment and benefits) and how all these factors are influenced by the site history and its location.

7 Conclusion

Despite the considerable C sequestration potential that afforestation offers, it is still unclear how the boreal forest C sink can be managed to mitigate atmospheric CO_2 build-up (Canadell and Raupach 2008). Uncertainties surrounding LW2 afforestation in the province of Quebec as a mitigation scenario are wide and make it an unattractive scenario. Afforestation projects in the boreal forest should target high productive stands where the low density cover is not the result of both edaphic or climatic constraints and simultaneously where vulnerabilities to natural disturbances are minimal. So, optimism of C net sequestration only based on hypothetical yield curves must be tempered. This paper pinpoints nonexhaustive factors with possible negative feedback for C offset to be considered: (1) Boreal forest C dynamics are influenced by past and present disturbance impacts. The impacts of recent disturbances will continue to influence forest NPP for many years as killed biomass decomposes and successional pathways are altered or reset. (2) With ongoing climate change, increases in NPP at the landscape level could in most cases not be significant enough to offset C losses from global increased natural disturbance rates. (3) The need to parameterize the counteracting effect of evapotranspiration and albedo in C sequestration models. (4) A complete economic analysis and the associated potential for a large-scale investment is needed.

Whatever the scope of such afforestation programs may be in the boreal forest, the level of reversibility is likely to increase, thereby affecting the sustainability of these projects. Therefore conceivable C management options will likely have to be directed toward minimizing the reversal impacts of the increasing natural disturbance activity rather than focusing solely on maximizing C stocks. Mitigation scenarios in the boreal forest, such as climate protection strategies, must incorporate the uncertainties, risk vulnerabilities and the increased probability of extreme events, as well as the planned periodic comparison of projections with the evolving reality so as to provide regular updates on targets (Bernier and Schöne 2009). Meanwhile, the social, economic and biological criteria must also be considered to sustain the key ecosystem services. Finally, research into afforestation in the boreal forest is still relatively new and requires major progress to support national policy development in compliance with international goals such as the UNFCCC. Sound science-based policy would help optimize the climatic benefits of C management in the boreal forest while reducing its counterproductive effects.

Acknowledgements This research was made possible thanks to the financial and logistical support provided by the Université du Québec à Montréal and the Canadian Forest Service. We acknowledge the contribution of Pamela Cheers who carefully edited the text. We thank the Centre for Forest Research for their logistical support. We also thank Pierre Bernier and Jerôme Laganière for their helpful suggestions and revision of a previous version of the manuscript.

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