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Forest protection and forest harvest as strategies for ecological sustainability and climate change mitigation

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ABSTRACT

An important consideration in forest management to mitigate climate change is the balance between forest carbon (C) storage and ecological sustainability. We explore the effects of management strategies on tradeoffs between forest C stocks and ecological sustainability under five scenarios, three of which included management and two scenarios which provide baselines emulating the natural forest. Managed forest scenarios were: (a) Protection (PROT), i.e., management by suppression of natural disturbance and harvest exclusion; (b) Harvest at a higher rate removing all sustainably available wood (HHARV); (c) Harvest at the lower historical average rate of harvest, AHARV. Both harvest scenarios reflected current forest management practices in the study area, including suppression of natural disturbance and a large (>20% of total) forest area reserved from harvest. Scenarios (d) and (e) simulated "natural" forest with unsuppressed fire at higher (NDH) or lower (NDL) levels and no harvest. Ecological sustainability was evaluated using a coarse filter approach where forest age class and tree species composition were indicators of condition. The study area encompassed 3.4 million hectares of forest in northeastern Ontario at the interface between the temperate hardwood and boreal forest zones. Future forest condition for each scenario was modeled using a timber supply model (SFMM), and C stored in forests and wood products were estimated using the FORCARB-ON model. Forest protection (PROT) resulted in greatest forest C stocks, especially in the near term, but was within 95% of its maximum, becoming saturated within 30 years. Harvesting (HHARV and AHARV) resulted in less forest C stock compared to PROT, however, after 100 years of adding C in wood products to that in regenerating forests total C storage was equivalent or greater than forest C with PROT. In contrast, removing management (NDH and NDL) decreased C relative to any of the management regimes, though in NDL the decrease was delayed for 30 years compared to HHARV. Forest sustainability measured by similarity to natural forest age class was superior with HHARV and AHARV compared to PROT, although no management regime produced a fully natural result. PROT in particular largely lacked younger age classes. All management regimes produced species composition that was near or within the range of natural variation. This analysis provides an example of the types of tradeoffs that can be considered in evaluating the contribution of forests to climate change mitigation, either in a commercial forestry context or in an approach based on protected areas.

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1. Introduction

Using forests to store carbon (C) sequestered from the atmosphere has received considerable interest as a climate change mitigation strategy (Nabuurs et al., 2007). Mitigation approaches fall into two broad strategies: Protecting forests from human and/or natural disturbances to maximize their C content, or harvesting forests and storing some of the carbon in harvested wood products (HWPs) while promoting regeneration of harvested areas. However, these contrasting mitigation strategies can significantly affect not just C but, more importantly, the ecological sustainability of forest ecosystems (Lattimore et al., 2009).

A forest protection strategy avoids reductions in forest C within the area being managed, primarily by excluding harvest. It may also employ management practices to control C losses caused by

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natural disturbances, such as fire and insects. If a forest area is not already under a sustainable harvest regime, then harvesting may result in a reduction in forest C, and this carbon creates a HWP C stock that can be managed with a comparatively high degree of certainty outside the forest. Over time after harvest, the extracted C is replaced due to the forest's capacity as a renewing natural C sink. In addition to the C stock, HWP can have the added benefit of offsetting the use of alternative building products that require higher energy input and therefore produce more greenhouse gas (GHG) emissions, or harvested wood can serve as a source of bioenergy in place of fossil fuels (Lippke et al., 2011).

Determining whether forest harvest or protection provides the greater C benefit is important in discussions of how best to mitigate climate change. However, these discussions should not overlook the divergent effects such strategies can have on forest ecological sustainability (Lattimore et al., 2009). Comparisons of mitigation and ecological sustainability at the same spatial and temporal scales is needed to provide an equitable viewpoint from which to judge the relative merits of these management approaches. Forest mitigation at regional scales can be assessed based on changes in forest and HWP C stocks, while age class structure and species composition can be used as regional-scale proxies for ecological sustainability (Harvey et al., 2002; Didion et al., 2007).

Disturbances such as fires in natural forests create a mosaic of species composition and age classes that result in natural ranges of biodiversity among plant and animal species (Weber and Stocks, 1998). It is widely accepted that one goal of harvesting is to use such patterns to incorporate the beneficial ecological consequences of natural disturbance in managed forests (Attiwill, 1994; Crow and Perera, 2004). In this way, harvest can be used to emulate some of the effects of an unmanaged fire regime by artificially creating a natural age class structure and species composition (Bergeron et al., 2002; Harvey et al., 2002; Perera et al., 2004; Long, 2009). While the concept of emulating areas of natural disturbance fits well with the goal of harvesting forests for timber, in practice applying the concept depends greatly on the reliability of estimates of current rates of fire and of the natural (or target) rates of fire to be emulated (Armstrong, 1999; Perera and Cui, 2010). While natural disturbance emulation is well studied in terms of consequences to forest structure, the manner in which natural disturbance emulation affects the ability of forests to contribute to the mitigation of climate change has received less attention.

In this study, we use a modeling approach to explore the hypothesis that in the long-term harvesting regimes can produce greater carbon benefits and ecological sustainability. Ecological sustainability is defined as having age-class structure and species composition similar to those under natural disturbance regimes only (no harvesting or suppression activities). The objectives of this study are (1) to compare and assess mitigation benefits (measured using forest and HWP C stocks) using protection or harvest approaches to forest management, and (2) to compare how protection or harvest regimes affects forest age class structure and species composition, proxies for ecological sustainability.

2. Material and methods

2.1. Study area

The study area is in eastern Canada where forests were historically subject to relatively frequent stand replacing natural disturbance (e.g., a median disturbance cycle of 98–108 years (Elkie et al., 2009)), but where according to recent history, the rates of natural disturbance are now reduced by fire suppression to produce fire cycles in thousands of years (Martell and Sun, 2008). The study area comprises forests in five Crown-owned forest management units (FMUs): Nipissing, North Shore, Spanish, Sudbury, and Temagami, spanning latitudes of 45°48.5″N to 47°50.6″N and longitudes of 78°19.5″W to 83°42.5″W (Fig. 1). These FMUs encompass a total area of 4.1 million hectares in northeastern Ontario, of which 3.4 million hectares is forested.

Each FMU is an administrative area within which forests are managed for ecological and social values, including wood production, protection from major disturbances, and with consideration of input from Aboriginal peoples and the general public. Forest management on Crown forests is guided strategically by a provincial planning manual (OMNR, 2009) with landscape, stand, and site level implementation governed by guides of practices to conserve biodiversity and avoid harming habitat for wildlife, watersheds,

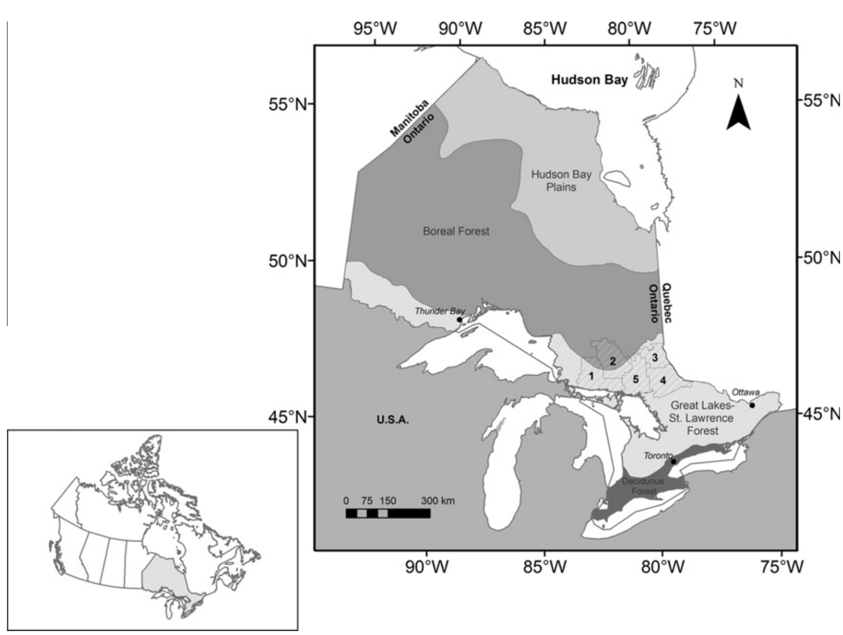

Fig. 1. Forest regions in Ontario and forest management units included in the study (hatched areas). 1. North Shore, 2. Spanish, 3. Temagami, 4. Nipissing, and 5. Sudbury.

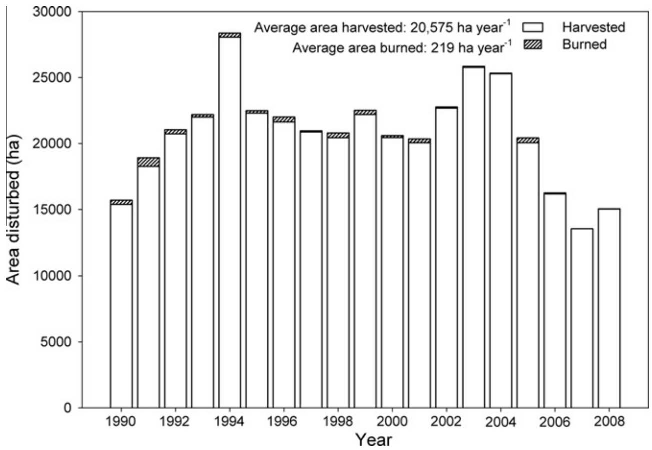

Fig. 2. Area disturbed by fire and harvest within the study area between 1990 and 2008. The total study area is 4.1 million hectares, of which 3.4 million hectares is forested.

and other societal values (e.g., OMNR, 2010). Forest management units are composed of forest "available" for harvest based on age and species composition, and forest not available for harvest (reserved) from harvest for ecological and social reasons. Both these areas are subject to protection from fire, and thus are all managed in some manner. In total, 22% of the forest area was reserved from harvest in the first decade of the simulation.

Laying at the northern edge of the Great Lakes–St. Lawrence forest biome, much of the forest in the study area shares functional and structural traits of the southern boreal forest. The forest consists predominantly of mixed stands of jack pine (*Pinus banksiana* Lamb.), black spruce (*Picea mariana* (Mill.) B.S.P.), balsam fir (*Abies balsamea* (L.) Mill.), white birch (*Betula papyrifera* Marsh.), and aspen (*Populus tremuloides* Michx.) in northern portions, with increasing proportions of hardwoods such as sugar maple (*Acer saccharum* Marshall), yellow birch (*Betula alleghaniensis* Britt.), and red oak (*Quercus rubra* L.) in southerly portions and red pine (*Pinus resinosa* Ait.) and white pine (*Pinus strobus* L.) in eastern areas. These forests have a history of human intervention since the mid-1800s (Armson, 2001). Areas of harvest and stand replacing fire between 1990 and 2008 are shown in Fig. 2.

2.2. Data sources and management scenarios

We obtained the most recent timber supply analyses of each FMU, prepared for the current forest management plans within the past 1–3 years. Timber supply analyses were performed using the Strategic Forest Management Model, SFMM (Kloss, 2002), the predominant forest management modeling tool used in Ontario. Clear-cutting was the main type of harvest system used. To determine maximum harvest over the 150-year planning horizon, first the forest area available for harvest was established in the management plan after total forest area was constrained by setting aside reserved forest to provide wildlife habitat, for parks, as riparian reserves, to meet requirements for old-growth forest, and to satisfy other non-timber objectives. Volumes available for harvest in each 10-year term in the remaining forest were then determined in SFMM using linear programming to harvest the maximum total merchantable volume over the entire planning period. In the timber supply analyses, area harvested varied over time and among forest species types.

We used outputs from the management plans' timber supply analyses for our maximum harvest scenario. Each scenario (a–e) in this study was obtained by summing the outputs (harvest volumes, areas by 10-year age class and forest unit) from the five management plan SFMM analyses, which accounts for natural processes (growth, succession, disturbance), area of reserved forest, and management constraints to control areas harvested and volumes harvested by tree species. The C stock in both available and reserved forest was determined separately but is presented as a single forest C stock for the five FMUs. Although the results are presented as composite values and the FMUs are contiguous, each FMU is administratively distinct, is managed separately, and required separate analysis as they have similar but not identical attributes for stand growth and forest succession (results for individual management units can be obtained from the authors).

At the start of the simulation, almost 22% (almost 738,000 hectares) of all forest in the study area was reserved from harvest, ranging from 16% in Nipissing to 32% in Temagami. Reserved forest increased to more than 26% of the forest land at the end of the simulation period. Although wood volume identified as available for harvest is deemed to be the "maximum harvest", because of the area of reserved forest required to meet ecological and management constraints, this volume is substantially less than would be available for harvest if all forest was eligible for harvest and subject to management to achieve maximum sustained yield (in which harvest takes place at the maximum mean annual increment in merchantable wood volume).

SFMM was used to determine forest age structure (area of forest in 10-year age classes) and species group composition over 150 years for each management unit and scenario. This study used results of the first 100 years of the planning period. In SFMM, stands can shift to a younger age class by two mechanisms – stand replacing disturbance (mainly harvest and fire) or natural succession. In the case of stand replacing disturbance the forest shifts to the youngest age class. With natural succession, stands can shift to younger forest ages (though not to the youngest age) and can either stay as the same species group or shift to a different species group.

Rates of stand replacing disturbance with fire suppression in the managed forest scenarios of the present study (Table 1) were those used in the most recent local forest management plans. These disturbance cycles were similar to those reported by Bridge (2001) and Martell and Sun (2008). Management units in this study nearer the Quebec border had rates of disturbance similar to those described for nearby forests in Quebec (Bergeron et al., 2006). As will be discussed later, these disturbance cycles do not take potential effects of future climate change on area burned into account.

The three managed forest scenarios combine forest fire suppression with no harvest, or with harvest at high or medium levels:

a. PROT – protection by excluding harvest and suppressing fire to minimize natural disturbance.
b. HHARV – higher harvest rate, minimum natural disturbance – harvest of all available tree volume combined with long fire cycles as in (a). Average annual harvest over the 2010–2110 study period was about 1.2% of total forest in the study area (counting both available and reserved forest area) with a harvest cycle among management units ranging from 88 to 165 years (Table 2);
c. AHARV – average historical harvest, minimum natural disturbance – future harvest rate at the ratio of historical to maximum available harvest volume occurring during the previous decade. We modified SFMM simulations to reduce future harvest from maximum to historical average volumes, ranging from about 38–73% of maximum harvest among FMUs, with an average ratio of 52% (Table 2).[2] Average

[2] Harvest volume data provided by Joe Maure, Ontario Ministry of Natural Resources, Sault Ste. Marie, Ontario, Canada.

Table 1
Natural disturbance dynamics either with forest management or at the lower and upper bounds of natural disturbance estimated in the pre-industrial forest. Maximum protection fire cycles are those specified in recent forest management plans.

Management unit	Fire cycle (years)			Average annual area of stand replacing fire (ha)			Published estimates of current fire cycle (years)	
	Maximum protection[a]	Natural		Maximum protection	Natural		Bridge (2001)[b]	Martell and Sun (2008)[c]
		Lower quartile	Upper quartile		Lower quartile	Upper quartile		
Nipissing	2250–3000	85	238	129	3673	1312	6739	6000
North Shore	8709	63	196	66	9170	2948	1447–>10,000	4000
Spanish	3442	69	185	245	12,239	4565	>10,000	4000
Sudbury	2600	63	170	157	6173	2288	12,315	10,000
Temagami	397–1059	69	198	444	4183	1458	919	2600
Total				1041	35,348	12,571		

[a] Ranges in some disturbance cycles for the maximum protection scenario indicate variation in fire cycle among forest types.
[b] Based on lightning and human-caused fires between 1972 and 1995. At the time of Bridge (2001), North Shore consisted of two management units that have since been amalgamated. The range of fire cycles for North Shore reflects the cycles in those two management units. Cycles >10,000 years reflect extremely low area burned during the period for which fire data was available to Bridge (2001).
[c] Values are approximations based on Martell and Sun (2008; Table 2) for lightning-caused forest fires between 1976 and 1994 in fire management compartments overlapping the management units comprising the present study.

Table 2
Harvest dynamics at two rates of merchantable volume removal.

Management unit	Harvest cycle (years)		Average annual area harvested (ha)		Ratio of historical to maximum sustainable harvest volume (%)
	100% harvest	Historical harvest	100% harvest	Historical harvest	
Nipissing	75	179	7999	3360	47.4
North Shore	93	120	9118	7103	72.5
Spanish	101	207	9996	4901	43.2
Sudbury	88	147	6246	3763	57.7
Temagami	165	388	2565	1092	38.4
Total	96	170	35,924	20,220	51.8

Note: harvest cycle is number of years to harvest an area equivalent to total forest area in the forest management unit.

annual area harvested in AHARV over the study period was about 0.7% of the total forest area (available and reserve), with a harvest cycle among management units ranging from 120 to 388 years (Table 2) and a rate of natural disturbance as in (a).

The two natural disturbance scenarios are:

a. NDH – high unsuppressed natural disturbance rate – no harvest and no fire protection, with stand replacing natural disturbance cycle from Elkie et al. (2009) at the lower quartile rate, ranging from 63 to 85 years among management units (a shorter disturbance cycle equating to a larger area burned). Under this high unsuppressed stand replacing natural disturbance rate, total average annual forest area affected is 35,348 ha (Table 1).
b. NDL – low unsuppressed natural disturbance rate – no harvest and no fire protection as in (d), with stand replacing natural disturbance cycle at the upper quartile rate, ranging from 170 to 238 years among management units (from Elkie et al., 2009), producing total average annual area burned of 12,571 ha (Table 1).

The areas of stand replacing disturbance in NDH and NDL span the middle 50% of the range of natural variation estimated by Elkie et al. (2009). The areas of natural disturbance were estimated using stochastic landscape models which simulated stand-replacing fire, spruce budworm outbreaks resulting in stand mortality, and landscape level wind events causing blowdown. Stand-replacing fire cycles were calculated by Elkie et al. (2009) using 25 replicate simulations of area burned by randomly placed disturbance events in the reconstructed historical forest. These rates of fire initiating events were based on a combination of area disturbed in historical survey line analysis (see Pinto et al., 2008) and published literature (Elkie et al., 2009). Each replicate produced a unique result of area burned based on stochastic weather conditions, forest species, fire sizes, and topography at the point of fire initiation. In addition to fire, a total area of disturbance was estimated by adding to fire disturbance the areas affected by spruce budworm defoliation and blowdown obtained from the historical survey line analysis. Fire was the dominant disturbance type and since a breakdown of the overall disturbance cycle into causal types was unavailable, we treated all stand replacing disturbance as being due to fire. The high quartile and low quartile disturbance cycles from these simulations varied among FMUs and ranged from a minimum of 63 years to a maximum of 238 years (Table 1).

These estimated composite natural disturbance cycles are of the same magnitude as the pre-industrial range of unmanaged fire cycles for the period prior to 1850 in western Quebec (Bergeron et al., 2001, 2006; Cyr et al., 2009), and for the pre-industrial period of forests to the south and west of the study area (Cwynar, 1977, 1978; Guyette and Dey, 1995; Cogbill, 1985; Cleland et al., 2004; Drobyshev et al., 2008). The question of whether such rates continue today is not settled. Although Bergeron et al. (2004b) indicate that such high rates ceased in this part of North America after the end of the Little Ice Age, Ter-Mikaelian et al. (2009) propose otherwise. However, this is only indirectly relevant to the present study, since the non-managed scenarios are used to provide an ecological benchmark of a natural forest condition, not as a prediction of current fire cycles in the absence of fire suppression.

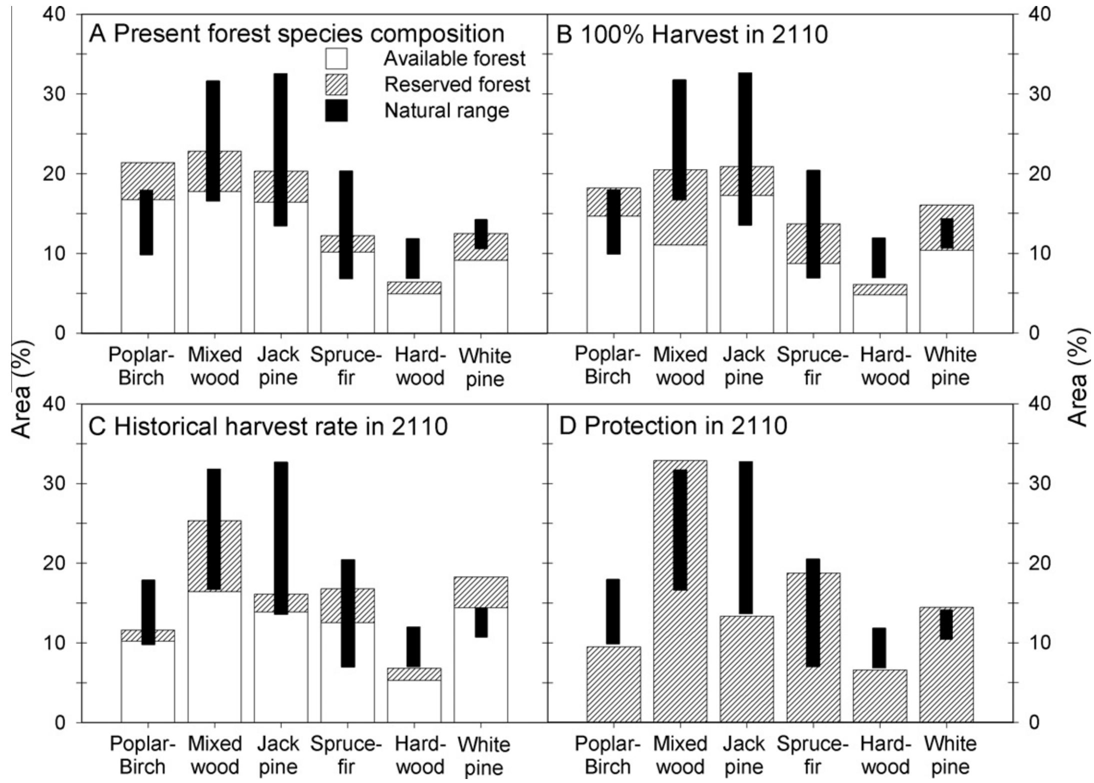

Fig. 3. Total areas of species groups (tops of stacked bars) and proportion of each species group residing in available and reserve forest, expressed as percent of total forest area for (A) the current forest (in 2010), (B) HHARV – maximum available harvest, minimum natural disturbance (in 2110), (C) AHARV – average historical harvest, minimum natural disturbance (in 2110) and (D) PROT – maximum protection (no harvest and minimum natural disturbance) in 2110. Black bars show the range of natural species group composition. The range was established based on stable composition obtained after simulating 2500 years (10000 years for Sudbury) of natural succession and natural regeneration without harvest for the upper and lower rates of natural disturbance. Where the top of the stacked bars overlaps the black bar the forest composition falls within the upper and lower natural ranges for a species group.

2.3. Assessment of forest sustainability based on age class and species group composition

Current forest age class distribution in 10-year age classes and species group composition were obtained from the provincial aerial forest inventory for each FMU. Species groups are common tree species associations (e.g., spruce-fir, aspen-birch, etc.) derived from those used in the management plan for each FMU (Fig. 3). Age class and species group composition 100 years from study start date were obtained from SFMM simulations for each FMU, for each of the scenarios described in Section 2.2.

In addition, we estimated a natural range of forest age class and species group frequencies for each FMU from simulations of scenarios (d) and (e), using the growth curves and succession rules from the forest management plan for each FMU. Non-harvest simulations were run until an equilibrium age class structure was reached for each FMU, i.e., 2500 modeling years (10,000 years for Sudbury). In these simulations, natural forest succession and a range of natural "burning" rates (scenarios (d) and (e) in Section 2.2) were used, providing us with an estimate of a natural range in forest age class and species group structure. We then compared the natural estimates to those from the 100-year simulations for managed forest in scenarios (a) to (c).

2.4. Forest and harvested wood product carbon

Carbon stocks in forests and HWP (including landfills) were estimated for each scenario and FMU using FORCARB-ON (Chen et al., 2010). FORCARB-ON uses outputs from SFMM of forest area, live tree merchantable volume, and area harvested and burned as inputs to estimate forest C. In addition to total forest C, the model provides estimates of C in seven forest pools: live trees, standing dead trees, down dead wood (i.e., logs and branches), understory vegetation, black carbon (formed during forest fires), forest floor, and mineral soil. Changes in forest C due to fire were estimated using a carbon flux matrix that accounted for C flows between pools and direct emissions due to combustion (see Table 2 in Chen et al., 2010). The large amounts of dead organic matter created when live trees are killed by fire subsequently decay according to decomposition functions contained in FORCARB-ON (Chen et al., 2010). Similarly, when forest harvest occurs in SFMM the post-harvest addition of harvest residues and its decomposition over time is accounted for in FORCARB-ON.

Furthermore, FORCARB-ON projects decadal increments of C in three types of HWP (construction lumber, other lumber, and paper). Harvested wood C is distributed in FORCARB-ON among product types taking into account conversion efficiencies from harvest-to-product and the fate of processing residues (which are used in secondary products, burned for energy, burned without energy generation, or placed in landfills). This analysis does not account for decomposition of legacy HWP in landfills at the start of the study period due to the lack of historical statistics of the volume and type of HWP produced from the forests in the study area over the previous 100 years. However, Chen et al. (2008) estimated that such an omission only slightly overestimates the net change in landfill C stocks (as legacy landfill HWP decompose); older HWP in landfills should have a relatively small effect on changes in net landfill C stock emissions, assuming about 73% of solid wood HWP remain in landfills indefinitely and the remaining nearly 27% decompose linearly over 80 years (Chen et al., 2008).

Table 3
Forest species composition under three management options compared with the natural range of variation in composition. Positive values exceed the upper bound of natural variation and negative values are below the lower bound. Rows may not sum to total due to rounding.

Scenario	Deviation of species group from natural range (%)						
	Poplar-birch	Mixedwood	Jack pine	Spruce-fir	Hardwood	White pine	Total (absolute value)
Current forest	+3.5	–[a]	–	–	−0.5	–	4.0
Protection	−0.3	+1.1	+0.2	–	−0.3	+0.2	2.2
Harvest – high	0.3	–	–	–	−0.8	+1.9	2.9
Harvest – average	–	–	–	–	−0.1	+4.1	4.1

[a] Within natural range.

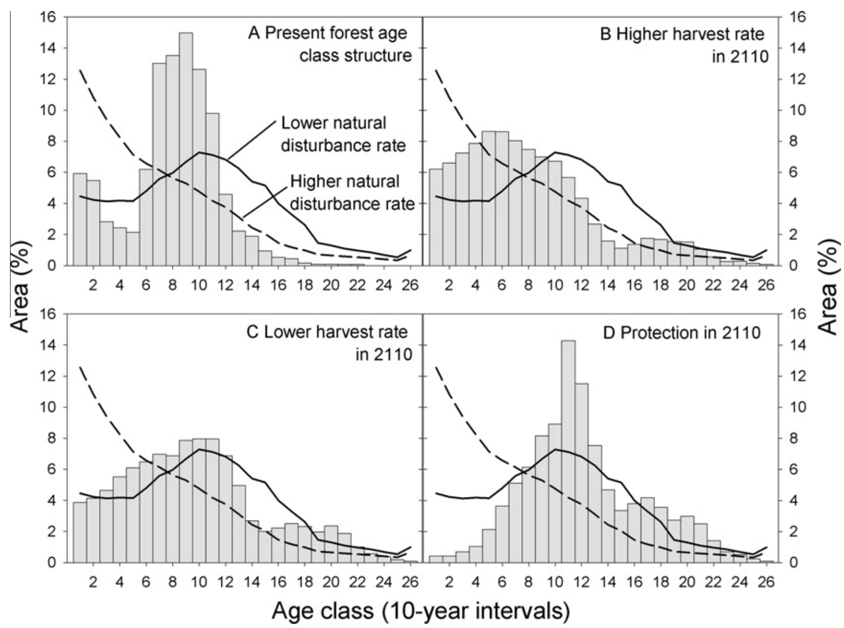

Fig. 4. Age class structures (bars) expressed as frequency distribution in 10 year increments (1 is stands 1–10 years old, 2 is 11–20 years, etc.) of (A) the current forest (in 2010), (B) HHARV – maximum available harvest, minimum natural disturbance (in 2110), (C) AHARV – average historical harvest, minimum natural disturbance (in 2110) and (D) PROT – maximum protection (no harvest and minimum natural disturbance) in 2110. The dashed line is natural forest age class distribution based on the higher rate of natural disturbance. The solid line is age class distribution for the lower rate of natural disturbance. The top of forest area bars falling between the higher and lower natural disturbance lines are within the natural range of area for that age class.

In-use HWP C stocks in this study were discounted temporally based on time-in-use curves for major HWP types either in-use or after use when disposed of by burning or placement in landfill, using the recent historical Ontario product mix described by Chen et al. (2008). Our analysis employs a production approach to assess HWP C (Skog, 2008), meaning that HWP C is accounted for in the jurisdiction in which timber was harvested, regardless where the HWP is used.

3. Results

3.1. Age class and species group composition

Current forest species composition is largely within the bounds of the natural forest (Fig. 3A, Table 3), with an overabundance of poplar-birch and a small under abundance of hardwood. The sum of deviations (ignoring sign) of current forest outside of the bounds of natural composition was 4.0%. A scenario of minimal disturbance (PROT) for 100 years produced the smallest sum of deviations from natural (2.2%), but had the greatest number (four of six) species groups falling outside the natural composition of the forest, though all by small amounts (Fig. 3D). In comparison, harvest for 100 years of all available volume (Fig. 3B) produced a small difference above the natural composition of poplar-birch and white pine and a small under abundance of hardwoods, with a sum of deviations from the natural forest for HHARV of 2.9% (Table 3). Harvest at the historical rate (AHARV) produced a natural species composition in most categories, but at the same time had the largest sum of deviations (4.1%), due almost entirely to the overabundance of white pine in this regime.

Table 4
Comparison of age class composition under three management scenarios with the natural range of variation in age classes.

Scenario	Sum of deviation from natural range (%)			Number of age classes outside of natural range (n out of 26)		
	Below	Above	Total	Below	Above	Total
Current forest	14.1	30.8	44.9	7	15	22
Protection	18.7	23.0	41.7	19	2	21
Harvest – higher	1.1	7.6	8.7	4	11	15
Harvest – average	1.5	6.9	8.4	11	5	16

The current forest is dominated by intermediate age classes (Fig. 4A), with more than two thirds of the forest between 61 and 120 years old. In general, the current forest age class composition is not representative of the natural forest (area between the solid and dashed line in Fig. 4). In particular, older forest is currently underrepresented.

In terms of age class structure, none of the scenarios were an exact match of the natural forest structure. However, harvesting at a higher rate (HHARV) (Fig. 4B), or harvesting at the historical average rate (AHARV) (Fig. 4C), produced structures that after 100 years more closely resembled a natural forest age class distribution than either the current distribution or the distribution under a protection scenario based on deviations from the natural range of age classes (Table 4). After 100 years in PROT, the forest age class distribution had an overabundance of intermediate and of some older age classes but especially a dearth of younger ones (Fig. 4D, Table 4).

3.2. Forest carbon

The forest C stock is at present 558 Tg. Of this, 40.5% (226 Tg) is in live trees and understory vegetation, 37.5% in soil (209 Tg), and 22.0% (123 Tg) in other dead organic matter (standing dead trees, forest floor, black carbon, and down dead wood). If forests reverted to a non-managed condition and were subject to uncontrolled pre-industrial rate natural disturbance, forest C stocks would generally decline from current levels (Fig. 5). In the case of the lower pre-industrial natural disturbance rate without harvest (NDL), forest C would temporarily increase, declining below current levels after 30 years. At the higher rate of natural disturbance (NDH), forest C would decrease almost 74 Tg by 2080 compared to initial C stocks.

Harvesting at the historical rate (AHARV) resulted in forest C stocks approximately the same as the lower rate of natural disturbance (NDL) – initially 3 Tg lower and later as much as 13 Tg higher. Harvesting at the maximum rate produced forest C stocks intermediate between NDH and NDL. However, only the minimal disturbance scenario (PROT) consistently increased forest C above present levels, peaking at almost 34 Tg more forest C above the current level in 2110 (Fig. 5).

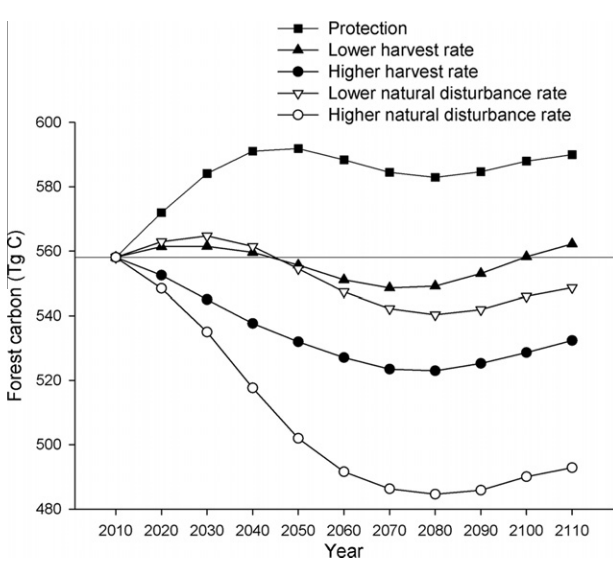

Fig. 5. Forest carbon stock projections under five scenarios: ■, PROT – no harvest and minimal natural disturbance; ▲, AHARV – harvest at the average historical (lower) rate, minimal rate of natural disturbance; ●, HHARV – harvest at the higher rate removing all sustainably available volume, minimal rate of natural disturbance; ▽, NDL –lower rate of natural disturbance without fire suppression, no harvest; and ○, NDH – higher rate of natural disturbance without fire suppression, no harvest.

The highest rates of forest C sequestration occurred in PROT during the first three decades after transition to a minimal disturbance regime (Fig. 6E). No other regime we examined matched these decadal average rates of forest C sequestration. The largest loss of C occurred during the first 50 years following transition to the NDH natural regime (Fig. 6G). Other regimes largely fluctuated between being moderate sinks and moderate sources of C throughout the 100 year modeling horizon (Fig. 6).

3.3. Harvested wood product carbon

Carbon stocks in HWP increased over time at near constant rates (Fig. 7); predictably, greater C stocks in wood in-use and landfills occurred when more volume was harvested. By 2110, in HHARV, wood in-use and disposed of in landfills contained about 27 and 42 Tg C, respectively, totaling almost 69 Tg C. In AHARV, there was almost 38 Tg C in HWP by 2110, divided between about 15 Tg in wood in-use and 23 Tg C in wood in landfills.

3.4. Combined forest and harvested wood product carbon

Forest management increased the combined forest-HWP C stock relative to the current C stock in forests (Fig. 8). However, for the first 60 years the increases were modest (AHARV) or negligible (HHARV), while forests regrew replacing the harvested forest C stocks. In both harvest regimes, after 70 years, the combined forest-HWP C stocks increased by between 0.4 and 0.8 Tg year^{-1} (Fig. 6C and D). By 2110, AHARV had stored about 9 Tg more C in forest plus HWP than HHARV (Fig. 8).

When HWP C was accounted for, natural forests C stocks were usually less than C stocks in managed forests (Fig. 8). An exception was during the first few decades, when C stocks in unmanaged forests with a low rate of natural disturbance (NDL) contained about 4 Tg more C than HHARV. Management that minimized disturbance (PROT) stored almost 100 Tg more C after 70 years than NDH (Fig. 8). Nearly all the increase in forest C in PROT occurred in the first few decades following exclusion of harvest (Fig. 6E); forest C increased at an average rate of about 1.1 Tg year^{-1} over the first three decades, with a maximum forest C stock of nearly 592 Tg reached in 2050 (Fig. 8). Combined forest-HWP C in AHARV exceeded that in PROT after 90–100 years. Based on trends in C accumulation, forest plus HWP C in HHARV would exceed that in the minimal disturbance scenario by 2120 (Figs. 6D and 8).

4. Discussion

This study shows that eliminating harvest from forests combined with low rates of natural disturbance has the potential to increase forest C. However, it has also shown that protection from disturbance in disturbance-prone forests can have negative ecological side-effects due to development of an overly old forest age structure, similar to the results of Scheller et al. (2005). Where natural disturbance rates are low in disturbance-adapted forests, forest managers may be able to redress some of these structural effects through judicious use of harvest (Bergeron et al., 2004a), but the balance between protection and harvest affects not just forest structure, but also mitigation of climate change through C storage in forest and wood products. The goal of integrating mitigation objectives and concerns about sustainability may lead forest managers to seek what Canadell and Raupach (2008) call sustainable mitigation.

A management approach where harvest substitutes for fire is premised on the assumption that current rate of burning is less

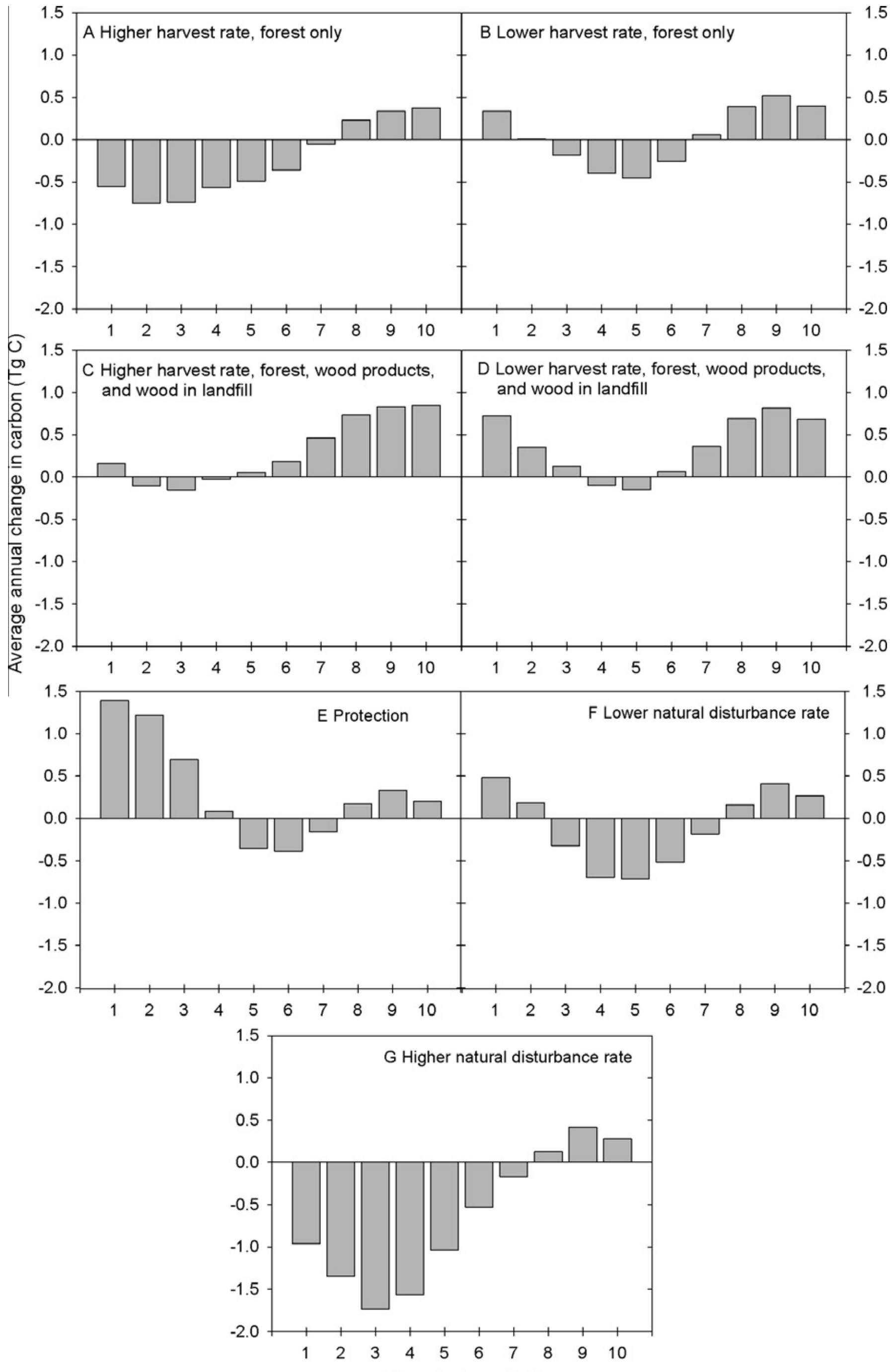

Fig. 6. Change in carbon by decade with different management approaches and considering different carbon pools: (A) and (C), HHARV – harvest at the higher rate removing all sustainably available volume (A is forest carbon only, C is total of forest, wood product and landfill carbon); (B) and (D), AHARV – harvest at the average historical (lower) rate (B is forest carbon only, D is total of forest, wood product and landfill carbon); (E), PROT – minimal natural disturbance and no harvest (forest carbon only); (F) NDL – lower rate of natural disturbance without fire suppression (forest carbon only); and (G) NDH – higher rate of natural disturbance without fire suppression (forest carbon only).

than the natural fire cycle (Le Goff et al., 2005). Otherwise, harvest plus the current rate of burning would disturb more area than the natural fire cycle, and the resulting age class structure would contain an overabundance of younger age classes (Bergeron et al., 2006). Conversely, higher harvest rates would be needed to create a natural forest structure if current rates of burning are less than the natural fire cycle (Bergeron et al., 2004a), or, in the absence of harvesting, some forest could be allowed to burn that might otherwise have been limited by aggressive fire suppression (Martell and Sun, 2008). Indeed, current rates of burning are generally considered to be less than the long-term historical rate in managed forests in eastern Canada (Bergeron et al., 2006), though the attribution of cause varies from fire suppression (Ward et al., 2001), to a change in climate featuring less severe droughts (Bergeron et al., 2004b), to alterations in forest structure, such as fragmentation (Bergeron et al., 2006) and larger areas of less flammable hardwood species (Drever et al., 2008).

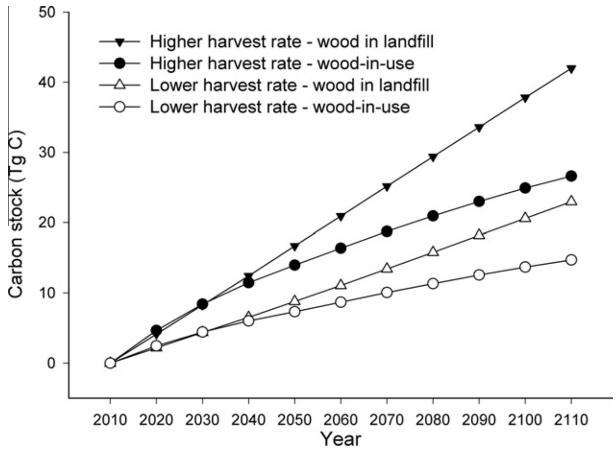

Fig. 7. Cumulative carbon stocks of wood in-use (●, ○) and wood disposed of in landfills (▼, △) for forests in five forest management units harvested at the historical (lower) rate (open symbols) or at the higher rate removing all sustainably available volume (solid symbols).

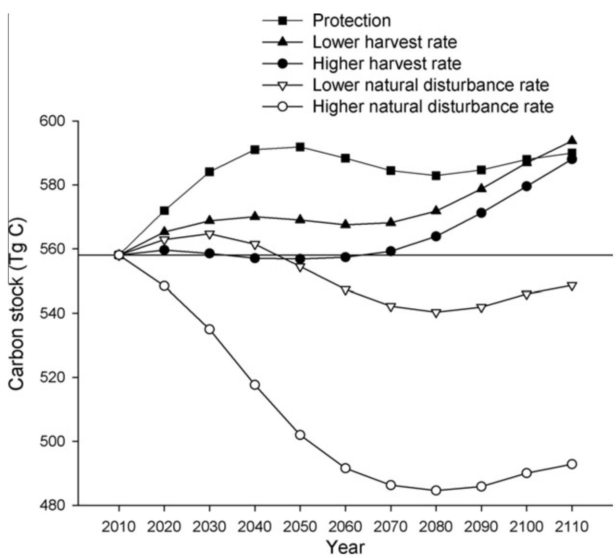

Fig. 8. Total carbon stock combined in forests, wood products in use, and wood products disposed of in landfills, projected under five scenarios. Symbols are as in Fig. 5.

4.1. Fire suppression and harvest exclusion as a climate change mitigation strategy

The managed and natural forest scenarios produced markedly different forest age structures as well as substantial differences in forest and HWP C stocks. In the southern boreal/northern temperate forests in this study, older stands generally contain more C than younger stands. In the absence of stand replacing disturbance, live trees in older stands in this forest area eventually decline and die, with tree replacement by shade tolerant understory species. One management option for using forests to mitigate climate change is therefore to allow an older age structure to develop by eliminating harvest and attempting to suppress large scale natural disturbances. This approach was reflected in the protection strategy, in which the highest levels of forest C occurred.

However, forests with minimal disturbance for 100 years, although having greater C stocks for most of the study period, acquired an age class distribution least representative of natural forests. Older and intermediate age classes were overrepresented and age classes younger than 60 years were highly underrepresented, compared to the natural range of age classes. Removing both harvest and fire from these disturbance-prone forest ecosystems not only removed an important element of variability in natural disturbance regimes (Bergeron et al., 2002), but almost eliminated younger forests. This may have important ecological impacts if an area managed to exclude all disturbance is very large, since plant species and functional types dependent on young forest originating from a stand-replacing disturbance, and animal species needing young forest stands for habitat or forage, will have difficulty finding suitable ecological conditions, reducing their occurrence compared to natural forests (Schulte et al., 2006).

In the protection strategy, the forest C stock ranged from about 585 to 590 Tg C over most of the simulation period. This approximates the biological potential of the forest C stock in this study area over the next 100 years. With a starting forest C stock of 558 Tg, current forests in this area are within about 95% of maximum achievable forest C over the next century. This contrasts with the findings of Canadell and Raupach (2008), which identifies a large global potential to increase C density using forest management. However, in our study area, the forests are already within 5% of maximum achievable C, likely a result of older average forest age due to a history of fire suppression and low harvest rates (Rehmtulla et al., 2009).

Results of this study depend to a large extent on assumed rates of natural disturbance. The natural forest scenarios (those with unsuppressed natural disturbance rates and no harvesting) provided baselines for comparisons of forest structure and carbon stocks under potential management scenarios. We consider the unsuppressed natural disturbance rates used in this study to be appropriate for evaluating the effects of emulating the natural forest, as these rates are similar to those reported for pre-industrial forests within the past 150–250 years (1–2 tree life spans). However, we do not imply that such rates would necessarily occur at present in the absence of fire suppression.

The management scenarios in the present study, especially the protection regime, are dependent on the premise that current observed long fire cycles are not temporary. This premise depends in part on the principle that fire suppression reduces area burned (Larsen, 1996; Ward et al., 2001; Cumming, 2005; Podur and Martell, 2007), though we recognize that suppression effectiveness continues to be debated (Bergeron, 1991; Johnson et al., 2001; Bridge, 2005). Fire cycles with suppression in the current study deviate little from those calculated based on actual area burned since the 1970s, with cycles measuring in thousands of years in the study area (Martell and Sun, 2008). In addition to fire suppression, current low rates of area burned in much of the eastern Canadian boreal forest are attributed to several other factors, including a change in climate after the end of the Little Ice Age (Bergeron and Archambault, 1993), an increased area of less flammable deciduous forests (Cleland et al., 2004; Drever et al., 2008), and increased forest fragmentation (Johnson et al., 1998).

4.2. Harvesting as a component of a climate change mitigation strategy

Harvesting for 100 years at either the historical rate or the maximum rate, when combined with substantial area of reserved forest and effective fire suppression, produced age class frequencies and species group composition reasonably similar to our projection of the natural forest. While harvesting clearly differs from fire in terms of site conditions after disturbance (McRae et al., 2001), it is likely the only practical means of obtaining a natural landscape structure in the study area, since allowing fires to burn uncontrolled would be problematic given the proximity of communities and infrastructure. Harvesting at the historical rate for 100 years produced a forest where the youngest age classes were slightly underrepresented, while many age classes were within or slightly above the natural range. Harvesting all available volume for 100 years also produced an age class structure and species group composition reasonably similar to the natural range, with the largest exceptions being overrepresented intermediate age classes (40–90 years old). Therefore, in terms of emulating natural forest structure, the protection management strategy appears less desirable, while harvest at historical rates or increased to the maximum level, by producing age classes and species group composition largely within the range of natural variation, may be more desirable.

Forest age in the study area is currently at a semi-steady state regulated by the fairly constant historical harvest rate and natural mortality, as was also found by Hurtt et al. (2002) for forests in the US. The relatively low historical harvest rates combined with protection from harvest of about one quarter of all forests have produced forest C stocks in the study area that are currently high and, as a result, there is only a moderate potential to increase the forest C stock by transitioning to a protection strategy. Sink size in the current study is limited because in the absence of disturbance, many species of trees in this region undergo natural mortality that leads to succession to younger trees with lower C levels.

4.3. Applicability of these results to other forest regions

In the current study, natural disturbance regimes D and E (i.e., no protection and no harvest) resulted in less forest C during the ensuing 100 years (at the lower rate of pre-industrial disturbance forest C stocks initially increased before decreasing after 30 years). It may seem counterintuitive that stopping removal of C by harvest would lead to decreased forest C. However, removal of C by harvest was more than offset by increased C removal by fire after the suspension of fire suppression. The resulting net decrease in forest C was attributable mainly to changes in forest age due to the managed disturbance cycle being longer than the natural one. Price et al. (1997) projected a similar outcome for western Canadian boreal forests when the harvest cycle was longer than the natural fire cycle. In contrast, when Krankina and Harmon (1994) modeled the effect of a transition from natural boreal forest to a managed scenario they projected decreasing C, which reflects the longer natural fire cycle they assumed (250 years) along with a shorter modeled harvest cycle of 60–100 years. Thus, the applicability of the present results to other forest areas is dependent on natural fire cycle, rate of forest harvest, and starting forest age class composition.

Kurz et al. (1998) projected greater C in natural than managed forest, even when the frequency of disturbance was the same. Their result reflects removals due to harvest causing greater average reductions in forest C compared to fire, since their modeled fire disturbance affected all age classes equally, some of which have low biomass C, while harvest affects only stands with high biomass C. In the present study, fire was similarly applied to all age classes equally and harvest targeted higher biomass stands. But these effects on forest C were offset due to a higher area of natural disturbance in the non-managed scenario compared to the area affected by harvest.

The finding in this study that management (suppressing fire and replacing natural disturbance with harvest) over a long time period can create a larger forest C stock, while maintaining a natural forest structure in the relatively short-lived tree species in the northern extremity of the Great Lakes – St. Lawrence forest, may not be the case in forests with longer-lived tree species in forest landscapes and long natural disturbance cycles. For example, in the Pacific Northwest of North America, natural forest will contain more C than harvested forests, given a harvest cycle considerably shorter than the natural disturbance cycle (e.g., Harmon et al., 1990). More generally, Keith et al. (2009) found that forests around the world growing in relatively cool climates with high precipitation and subject to minimal natural disturbance, have higher C stocks than if such forests are harvested on short cycles. Similarly, the low natural disturbance rate of tolerant hardwood forests of northeastern North America, combined with the long-lived nature of many tree species occurring there, means these forests likely store more C in the long-term if they are protected rather than harvested (Hurtt et al., 2002; Meng et al., 2003; Nunery and Keeton, 2010).

It is important to note that although the maximum or historical harvest rates produced a forest structure that tended toward that of a natural forest, it remains to be seen whether similar results would occur in other parts of the managed boreal forest. The methodology applied is one possible means forest managers could use to evaluate whether their unique forest conditions combined with local management constraints would emulate the natural forest.

This study did not account for potential effects of climate change on forest growth, succession, and fire occurrence (Flannigan et al., 2005), factors which as time went on could increasingly affect harvest rates and carbon storage. However, Bergeron et al. (2004a,b) found that while a doubling or tripling of global atmospheric CO_2 concentrations would increase future burn rates in the general vicinity of this study, increases would not result in the high historical pre-industrial fire cycles and might be partly offset with greater investments in fire suppression.

4.4. The role of wood products in mitigation using harvest

Accounting for C stocks in HWP gives a more complete comparison between C stocks in managed and natural forests. Harvested trees create a C stock that can accumulate in wood in-use or, after use, when wood is transferred to landfill (Chen et al., 2008). In the present study, a protection strategy that minimized disturbance by fire suppression and excluded harvesting produced the greatest forest C stock, but produced no stocks of C in HWP. Ultimately, the greatest combined C stock (forest plus HWP) occurred when natural disturbance was minimized and harvesting was carried out at the historical rate, although it took nearly 100 years of harvesting to achieve it.

An added benefit of managed forest C versus natural forest C is the diversification of C stocks in locations outside the forest. Storing C in multiple HWP in use and after disposal provides a measure of security not found when the entire C stock is in the forest, subject to natural disturbance (Apps et al., 2000; Neilson et al., 2006).

Due to long-term uncertainty, we did not estimate emissions of methane from wood decomposing in landfills. Methane released from the decomposition of organic material like HWP is potentially significant as it has a global warming potential approximately 25 times that of CO_2. Current methane emissions from wood in landfills offset a large portion of the wood products C stock; according to Lippke et al. (2011), it offsets 55% of the landfilled wood C stock and 135% of C in paper in landfills. However, it seems probable that methane emissions will be reduced substantially in future. Landfill methane release from wood may become insubstantial if HWP are diverted to recycling before entering landfills, burned for bioenergy in place of fossil fuels, or flared to convert the methane to CO_2 (Eggers, 2002; Mohareb et al., 2008; Lippke et al., 2011). For this analysis, all C emissions from landfills were counted as CO_2 and C emissions from HWP life cycles (harvesting, silviculture, transportation, and manufacturing) were ignored, while also ignoring emissions reductions due to product displacement using wood. This is the same as assuming HWP emissions and avoided emissions are in balance. Based on Werner et al. (2010), this assumption may underestimate the mitigation benefits of product substitution.

5. Conclusions

Forest protection as a mitigation strategy resulted in the forest becoming C-saturated within 30 years, which results in low additional carbon benefits after the first 30 years. This scheduling of mitigation benefits may or may not be preferable, so the result demonstrated that the trajectory of mitigation benefits needs to be a consideration when evaluating forest carbon sequestration strategies. Planned forest management in the study area, consisting of substantial areas of harvest combined with an area of 20–25% protected forest, tended towards a natural age structure, provided an ongoing capacity to sequester C, and gave protection against inadvertent release of C stocks to the atmosphere by diversifying stocks between the forest and HWP.

Moreover, the large-scale, long-term protection of forests prone to stand replacing natural disturbance produced a forest with an age structure skewed towards older stands, with less young forest than occurs within the natural range of the landscape. It also produces a forest with more shade tolerant species and fewer species that regenerate following disturbance than are found in a natural landscape. If the area managed in this manner were large, then having fewer young stands and fewer disturbance-origin stands in the landscape compared to historical variability may mean that certain biological and ecological functions provided by these types of forests will be underrepresented (Didion et al., 2007) and species dependent on such forests for habitat or forage may be negatively affected (Rempel et al., 2007).

Despite the fact that fire is a natural process, avoiding uncontrolled stand replacing fires is preferred in this part of Canada due to the proximity of people and human infrastructure. Although harvesting is an imperfect tool for emulating the effects of fire (McRea et al., 2001; Perera and Cui, 2010), coupled with prescribed burning it offers forest managers a more controlled means of emulating some of the landscape structural effects of fire (Long, 2009). In the present study, we contrast a protection strategy which excludes all disturbance with harvesting strategies that protect a substantial area (about 20% of the forest was reserved from harvest). The harvest strategies (with their substantial areas of protected forest) were projected to be able to provide for ecological sustainability (by promoting a more natural forest structure), socially acceptable fire risk, supporting a wood products industry while contributing to climate change mitigation.

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