Spruce budworm and management effects on forest and wood product carbon for an intensively managed forest

Chris R. Hennigar and David A. MacLean

Abstract: An integrated forest management optimization model was developed to calculate potential spruce budworm (*Choristoneura fumiferana* Clemens) effects on forest and wood product carbon (C) from 2007 to 2057 and to evaluate potential C sequestration benefits of alternative management strategies (salvage, biological insecticide application). The model was tested using simulated spruce budworm outbreaks on a 210 000 ha intensively managed forest in northwestern New Brunswick, Canada. Under a severe spruce budworm outbreak scenario from 2007 to 2020, harvest volume and forest and wood product C storage in 2027 were projected to be reduced by 1.34 Mm^3, 1.48 Mt, and 0.26 Mt, respectively, compared with the levels under no defoliation. Under the same severe outbreak scenario, implementation of salvage and harvest replanning plus a biological insecticide applied aerially to 40% of susceptible forest area, reduced harvest, forest C, and wood product C impacts by 73%, 41%, and 56%, respectively. Extrapolation of these results to all of New Brunswick suggests that a future severe spruce budworm outbreak could effectively increase total provincial annual C emissions (all sources) by up to 40%, on average, over the next 20 years. This modeling approach can be used to identify to what extent insecticide application, as a forest-C-offset project, could result in additional C storage than without forest and pest management.

Résumé : Un modèle intégré d'optimisation de l'aménagement forestier a été développé pour calculer les effets potentiels de la tordeuse des bourgeons de l'épinette (*Choristoneura fumiferana* Clemens) sur le carbone (C) forestier et celui des produits du bois de 2007 à 2057 et évaluer les bénéfices de stratégies alternatives d'aménagement (coupe de récupération, application d'insecticide biologique) sur la séquestration potentielle du C. Le modèle a été testé en ayant recours à des épidémies simulées de tordeuse des bourgeons de l'épinette dans une forêt de 210 000 ha aménagée de façon intensive dans le nord-ouest du Nouveau-Brunswick, au Canada. Comparativement à l'absence de défoliation, un scénario impliquant une épidémie sévère de tordeuse des bourgeons de l'épinette de 2007 à 2020, réduirait de respectivement 1,34 Mm^3, 1,48 Mt et 0,26 Mt le volume de récolte et le stockage du C en forêt et dans les produits du bois en 2027. Dans le même contexte d'une épidémie sévère, le recours à la coupe de récupération et à la replanification de la récolte en plus de l'application aérienne d'un insecticide biologique sur 40 % de la superficie de forêt vulnérable a réduit de respectivement 73, 41 et 56 % les impacts sur la récolte, le C forestier et celui des produits du bois. L'extrapolation de ces résultats à l'ensemble du Nouveau-Brunswick indique que dans l'avenir une épidémie sévère de tordeuse des bourgeons de l'épinette pourrait effectivement augmenter jusqu'à 40 % en moyenne les émissions annuelles totales (toutes les sources) de C de la Province au cours des 20 prochaines années. Cette approche de modélisation peut être utilisée pour déterminer dans quelle mesure l'application d'insecticide pour réduire les pertes de C forestier pourrait se traduire par un stockage additionnel de C comparativement à une situation où il n'y aurait ni aménagement de la forêt, ni gestion des ravageurs.

[Traduit par la Rédaction]

Introduction

Heightened pressure to reduce Canada's greenhouse gas emissions has amplified the need for reliable information to identify forest management and protection contributions toward forest carbon (C) sequestration. Conceivable contributions include silviculture to increase tree growth and pest management to maintain C storage (i.e., to prevent tree growth loss and mortality during insect outbreaks). Trees sequester C directly; major insect outbreaks kill trees over large areas, thus reducing C storage, preventing living biomass from accumulating C, and resulting in C transfer to the atmosphere as dead trees decompose. Recent research has evaluated the effects of (*i*) gypsy moth (*Lymantria dispar* L.) outbreaks on C using eddy covariance to measure net carbon dioxide (CO_2) exchange with the atmosphere and using biometric measurements to characterize net ecosystem productivity (Clark et al. 2010), and (*ii*) hemlock wooly adelgid (*Adelges tsugae* Annand) outbreaks on C dynamics of eastern United States forests (Albani et al. 2010).

Periodic population eruptions of spruce budworm (*Choristoneura fumiferana* Clemens; SBW) occur throughout much of North America's boreal, Great Lakes – St. Lawrence, and Acadian forests. These usually last 5–15 years, reoccur every 30–40 years (Royama 1984), and cause severe defoliation of balsam fir (*Abies balsamea* (L.) Mill.) and spruce

Received 1 February 2010. Accepted 20 April 2010. Published on the NRC Research Press Web site at cjfr.nrc.ca on 13 August 2010.

C.R. Hennigar[1] and D.A. MacLean. Faculty of Forestry and Environmental Management, University of New Brunswick, P.O. Box 4400, Fredericton, NB E3B 5A3, Canada.

[1]Corresponding author (e-mail: chris.hennigar@unb.ca).

(*Picea* spp.), resulting in major effects on forest management (e.g., MacLean 1980; Baskerville 1995; MacLean et al. 2002). SBW caused an estimated loss of 44 × 10^6 m^3 of timber volume per year from 1977 to 1981 at the peak of the last outbreak in Canada (Sterner and Davidson 1982). Understanding the effects (e.g., defoliation and resulting impacts on tree growth and survival) of SBW populations on hosts, in conjunction with current and potential forest-pest management during outbreaks, is fundamentally important for predicting forest C dynamics in eastern Canada.

Analysis of C dynamics in Canada's managed forests for the period 1920 to 1989 demonstrated that net ecosystem C storage was positive until 1979, but as a result of high rates of natural disturbance (wildfire and insect outbreaks) from 1970 to 1989, the forests of Canada became a net C source from 1980 to 1989 (Kurz et al. 1995; Kurz and Apps 1999). Projections of national ecosystem C for the periods 1980–2032 (Kurz and Beukema 2001) and 1990–2040 (Kurz and Apps 1995) demonstrated that future disturbance rates may have a large impact on the direction (sink or source) and magnitude of forest C stock changes. Canada's managed forests are projected to be a source of 8.2–66.8 Mt C·$year^{-1}$ (converted from CO_2 equivalents per year) during the first Kyoto commitment period, from the combined risk of fire, mountain pine beetle (*Dendroctonus ponderosae* Hopkins), increased salvage logging, projected SBW outbreaks starting between 2008 and 2011, and other insects (Kurz et al. 2008*a*). Kurz et al. (2008*b*) estimated that from 2000 to 2020 the cumulative mountain pine beetle forest C impacts will be 270 Mt. These natural disturbance impacts on forest C are potentially large enough to completely counter the C sequestration benefits of, and incentives for, improved forest management under Canada's current C accounting rules, as natural disturbance effects on C dynamics are currently reported in national forest C stock changes (Böttcher et al. 2008; Kurz et al. 2008*a*).

While forest C analyses on a national level have included estimates of human and natural disturbances, C analyses at the level of operational forest management become complicated because of difficulties in estimating (*i*) insect-caused tree mortality and growth loss for specific stand types (species, age, management); (*ii*) stand dynamics following non-stand-replacing disturbances (Kurz et al. 2003); (*iii*) forest-level effects of salvage and modified harvest scheduling; and (*iv*) subsequent effects of pest and forest management on forest and wood product C stock dynamics over time. Salvage harvesting removes trees recently killed by insects or fire, while modified harvest scheduling reflects amendments to planned stand interventions (timing, amount harvested or planted) in response to anticipated tree growth loss and mortality. Kurz et al. (2003) determined that without salvage logging, simulations of national ecosystem C accumulation were underestimated by 4.5% from 1920 to 1989 because historical insect- and human-disturbed areas were double-counted. Spruce–fir harvest impacts from SBW disturbance can be reduced by up to 30% when modified harvest scheduling with salvage is used to minimize harvest volume loss from SBW, compared with using the preplanned schedule with no modifications for SBW disturbance (Hennigar et al. 2007). These examples suggest that if the effects of harvest plan amendments during SBW outbreaks are not explicitly accounted for, the resulting future forest conditions and harvest levels may be poorly represented, and the conclusions of the effects of pest management on both forest and product C stocks will be distorted.

Over the last decade, research has made progress in estimating (*i*) stand- and forest-level impacts of SBW by using a decision support system (DSS) (Erdle 1989; Erdle and MacLean 1999; MacLean et al. 2001; Hennigar et al. 2007) and (*ii*) C dynamics in the forest (Neilson et al. 2008; Kurz et al. 2009) and off-site in wood products (Apps et al. 1999; Hennigar et al. 2008*a*). In this study, we synthesize these methods (Hennigar et al. 2007, 2008*a*; Neilson et al. 2008) into a single forest management linear optimization environment (Woodstock; Remsoft Inc.) to estimate SBW disturbance effects on volume harvested and C dynamics both in the forest and in wood products for a variety of SBW outbreak and management scenarios. This approach permits a more holistic representation of SBW–human disturbance effects on forest C by combining stand-level volume and C dynamics with and without defoliation, with management (salvage, modified harvest scheduling, and insecticide application) to minimize harvest impacts.

We approached this by simulating alternative SBW outbreaks and management scenarios from 2007 to 2057 for a 210 000 ha intensively managed forest in northwestern New Brunswick. Our objectives were to provide an integrated SBW–forest C model to (*i*) forecast potential SBW outbreak effects on forest and wood product C, (*ii*) help define C stock baselines (Böttcher et al. 2008) for forest management activities with or without SBW, and (*iii*) evaluate potential C sequestration benefits of alternative pest management strategies to help guide forest C policy decisions.

Methods

Study area

The J.D. Irving, Limited, Black Brook District located in northwestern New Brunswick, Canada (47°9′51″N to 67°55′27″W), was used to demonstrate the application of our model framework. Approximately 198 000 ha are productive forest, with 139 500 ha being susceptible to SBW (defined here as the area projected to sustain defoliation or because stands have a >10% host volume composition), and 69% of the susceptible area containing ≥75% host species by volume (Table 1). Nonsusceptible area is nearly totally composed of shade-tolerant hardwoods. Common stand types and management regimes include (*i*) spruce plantations commercially thinned 2–3 times before final harvest (82 000 ha); (*ii*) uneven-aged shade-tolerant hardwood stands managed by single-tree or group selection harvest (65 000 ha), including yellow birch (*Betula alleghaniensis* Britton), sugar maple (*Acer saccharum* Marsh.), and beech (*Fagus grandifolia* Ehrh.); (*iii*) intolerant to intermediately tolerant hardwood (5000 ha) dominated by trembling aspen (*Populus tremuloides* Michx.), red maple (*Acer rubrum* L.), and white birch (*Betula papyrifera* Marsh.); (*iv*) softwood (balsam fir, spruce, cedar (*Thuja occidentalis* L.); 29 000 ha); and (*v*) mixedwood stands (18 000 ha) managed using selection, shelterwood, patch, or clear-cut harvesting. Restricted harvest (e.g., deer wintering areas) and no-harvest zones (e.g., buffers, reserves, not operable) comprise 12%

Table 1. Percentage of susceptible area (139 500 ha; area with >10% host volume) of the Black Brook District in New Brunswick, divided into spruce budworm stand impact classes* based on species composition, age class (immature (<40 years old) or mature), and management (intensive or extensive).

	% Species composition†		Susceptible area (%)			
Species class‡	% Host	% Host as FW	Immature	Mature	Intensive§	Extensive§
FW	≥75	≥75	22	7	27	2
FW-RB		50–74	8	5	8	5
RB-FW		0–49	20	7	24	3
FW-NH	50–74	50–100	2	5	0	7
RB-NH		0–49	0	4	0	4
NH-FW	25–49	50–100	1	10	0	11
NH-RB		0–49	1	0	0	1
NH	11–24	≥0	0	8	0	8

*Seventy-two stand impact classes are represented in the stand impact matrix, representing 18 species classes (those shown plus others) × two maturity classes (immature, mature) × two management types (intensive or extensive). Area with ≤10% host species was considered nonsusceptible.

†Percentage by volume measured at outbreak initiation.

‡Host species — FW, balsam fir and (or) white or Norway spruce; RB, red and (or) black spruce; NH, non-host species.

§Intensive management includes planted or thinned stands, while extensive management refers to stand types originating from natural regeneration or single-tree or group selection harvest.

and 10% of the productive area, respectively. Detailed descriptions of historical forest conditions and management can be found in Hennigar et al. (2007).

The J.D. Irving, Limited 2007 forest estate model was built by J.D. Irving, Limited using the program Woodstock and was used as the base model formulation (no defoliation) for all simulations. Woodstock is a flexible and widely accepted modeling program developed by Remsoft, Inc., and is capable of solving complex mathematical forest management problems through the use of commercial linear optimization solvers or simulation modeling. The Black Brook District forest is managed to maximize nondeclining discounted sawlog and pulpwood harvest of spruce–fir, yellow birch, and sugar maple. While the company's main objective is profit, J.D. Irving, Limited is a vertically integrated (forest operations, mills, multiple products, retail) company, and at times, may derive more revenue from mill operations than forest operations. For this and other reasons, strategic timber supply modeling has been simplified both within forest management planning at J.D. Irving, Limited and in this paper by maximizing harvest rather than profit. At the stand level, silviculture is limited to stand conditions (tree size, products) considered profitable to harvest, from a company perspective. Other non-timber values such as water quality and biodiversity are managed using zoned harvest restrictions on water-course buffers, deer wintering areas, vegetation communities, and unique sites (J.D. Irving, Limited 2002; New Brunswick Department of Natural Resources 2005).

Spruce budworm outbreak scenarios

Rather than real-time prediction of future budworm outbreaks, the SBW DSS uses a scenario planning approach to examine "what if" outcomes of user-specified scenarios. It is not possible to predict SBW outbreak dynamics in advance of an outbreak. Once SBW populations begin to rise, pheromone traps, egg mass, and larval sampling are used operationally to estimate probable upcoming-year population and defoliation levels. Budworm scenarios typically include altering (*i*) the temporal sequence of defoliation that comprises an outbreak, (*ii*) the timing of when an outbreak begins (e.g., "What if" an outbreak began in the year 2010, or 2015?), and (*iii*) the duration of the outbreak. Scenario planning is a disciplined method for imagining possible futures that has been applied to a wide range of issues (Schoemaker 1995) and is useful in strategic planning given its ability to capture a whole range of possibilities, thereby allowing managers to see a wider range of possible futures. In that sense, scenarios can be interpreted as "if this defoliation pattern occurs, here is the impact that will result".

In this paper, we used two SBW outbreak scenarios (defoliation patterns) as developed previously from data for New Brunswick and Nova Scotia (MacLean et al. 2001). The SBW densities per square metre of foliage (log scale) from 1945 to 1965 (see fig. 1 in Royama (1984)) were converted into SBW density classes (nil, light, moderate, high, and extreme, as used by New Brunswick Department of Natural Resources — see fig. 2A in Erdle and MacLean (1999)), and in turn, were converted to current annual defoliation (%) using SBW density-to-defoliation conversions (see fig. 2B in Erdle and MacLean (1999)) — SBW density classes: nil, 0% defoliation; light, 20% defoliation when SBW populations are rising and 10% when populations are falling, due to parasite build-up in the population; moderate, 60% (rising) or 40% (falling); high, 90% (rising) or 70% (falling); and extreme, 100% (rising) or 80% (falling). The moderate SBW outbreak scenario resulted (see fig. 2C in Erdle and MacLean 1999) and is meant as a generalized pattern of defoliation observed in New Brunswick during the 1970s–1980s.

Previous analyses permit some validation of this outbreak scenario for our study area. Porter et al. (2004) analyzed historical area of light (10%–30%), moderate (31%–70%), or severe (71%–100%) SBW defoliation in the Black Brook

District from 1945 to 1995. Two distinct outbreaks occurred — from 1949 to 1958 and from 1971 to 1987 — with up to 97% of the landbase defoliated in 1955 and 1957. Aggressive insecticide protection programs in the 1970s–1980s substantially reduced defoliation; in the 8 years of heaviest defoliation, more than half of the landbase was protected and nearly the entire forest was protected in 1976. Porter et al. (2004) estimated that without insecticide protection, mean area defoliated (>10%) by SBW from 1971 to 1987 would have increased from approximately 48 000 to 99 000 ha·year^{-1}, and in 1976 from 35 000 to 185 000 ha, based upon annual published records of spray efficacy (e.g., Carter and Lavigne 1994; Kettela 1995); generally, areas not sprayed would be moved into the next higher category of defoliation severity (e.g., moderate increasing to high). The predominant (estimated unsprayed conditions) defoliation pattern in the study area was 5 years of moderate to severe defoliation and 4 years of light defoliation, which is generally similar to the 5 years of ≥80% defoliation in our moderate outbreak scenario. The average percentage of cumulative defoliation during the 1950s SBW outbreak for five mixedwood stand types in the Black Brook District (Amos-Binks et al. 2010) was also similar to our moderate outbreak scenario.

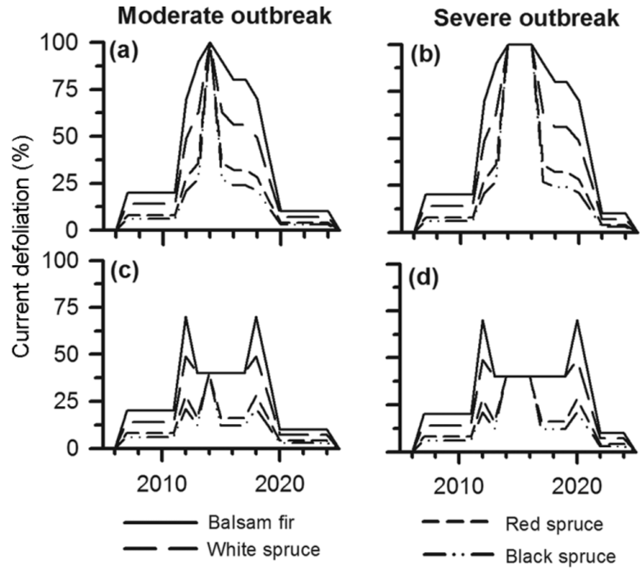
An uncontrolled SBW outbreak on Cape Breton Island, Nova Scotia, during the 1970s–1980s, where populations reached extremely high levels (Ostaff and MacLean 1989), exhibited three defoliation patterns, with the predominant one having 7 years of moderate–severe defoliation. To represent this, we created a second "severe outbreak" scenario with two more years of 100% defoliation at the peak (MacLean et al. 2001).

Gray and MacKinnon (2006) used historical records (1941–1998) of SBW defoliation across Canada to determine 27 different outbreak patterns. Our study area sustained two discrete outbreaks over this period, with no measurable defoliation from 1959 to 1970 (Porter et al. 2004). Although not precisely comparable because of different analysis periods, our moderate and severe scenarios do approximate the Gray and MacKinnon (2006) defoliation pattern numbers 5 and 6 for the 1950s outbreak alone, or numbers 12 or 18 for the 1950s and 1970s–1980s combined.

We modified the moderate and severe defoliation scenarios to include defoliation differences among host species predicted by Hennigar et al. (2008*b*), where defoliation on white (*Picea glauca* (Moench) Voss), red (*Picea rubens* Sarg.), and black (*Picea mariana* (Mill.) Britton, Sterns & Poggenb.) spruce was approximated as 72%, 41%, and 28%, of balsam fir defoliation, respectively (within ±5% of curvilinear linear models reported by Hennigar et al. 2008*b*). Norway spruce (*Picea abies* (L.) Karst.; 8460 ha of plantations) defoliation was assumed equal to white spruce levels. Outbreak scenarios were simulated to begin in 2007 and cause 50%–100% defoliation of current-year foliage of fir and spruce (Figs. 1*a* and 1*b*) from 2011 to 2020.

The relative differences in defoliation among fir and spruce species may be less in highly defoliated (>90% of current-year foliage) stands as a result of movement of SBW larvae from fir and white spruce to less preferred red and black spruce when foliage of the former becomes limiting (nonlinear effects observed in regression tree analyses

Fig. 1. Percent defoliation of current-year foliage on balsam fir, red, white, and black spruce for moderate and severe spruce budworm outbreak scenarios without (*a*, *b*) and with (*c*, *d*) aerial foliage protection applied in years when defoliation is >70%. Spruce defoliation was calculated from projected fir defoliation and reduced spruce defoliation probability models (Hennigar et al. 2008*b*) and was assumed to equal fir defoliation when >90%.



by Hennigar et al. 2008*b*; see also Blais 1957 and Nealis and Régnière 2004). To reflect these effects, spruce defoliation was assumed equal to fir when fir defoliation was projected to be >90% (Fig. 1). This assumption had minimal effect on mean cumulative spruce defoliation for moderate outbreak scenarios (<10% from 2012 to 2016 for red and black spruce), but for severe outbreak scenarios, cumulative defoliation from 2012 to 2021 was 20%–30% higher for red and black spruce and 10% higher for white spruce. Sensitivity of results to this spruce defoliation assumption was evaluated by simulating additional severe outbreak scenarios with (*i*) spruce defoliation equal to that of fir in highly (>90%) defoliated years, and (*ii*) spruce defoliation equal to fir in all years, as used in past SBW impact analyses (e.g., MacLean et al. 2001, 2002; Hennigar et al. 2007).

Scenarios simulated aerial application of the biological insecticide *Bacillus thuringiensis* to reduce defoliation to 40% of current-year foliage (New Brunswick's current protection efficacy target) in years when it was forecast to be >70% (considered severely defoliated in New Brunswick aerial surveys). Protecting during peak severity years increases protection program effectiveness in comparison with protecting in all years when defoliation is >40% (Hennigar et al. 2007), and therefore, protected scenarios (Figs. 1*c* and 1*d*) reflect aerial applications in only 5 and 7 years for moderate and severe outbreak scenarios, respectively. Scenarios included 0%, 10%, 20%, 40%, 70%, or 100% of susceptible forest area protected (Table 1).

A key point increasing our confidence in interpreting SBW DSS projections is that growth reduction and mortality versus defoliation relationships are consistent and repeatable across outbreaks and regions (e.g., MacLean 1980; Ostaff and MacLean 1995; Erdle and MacLean 1999). Another key point is that all SBW DSS modeling projections are based upon 5-year cumulative defoliation, in 10% classes, so that multiple current defoliation scenarios can result in the same 5-year cumulative defoliation. Hennigar et al. (2007) analyzed this with respect to outbreak severity (moderate versus severe scenarios) and various combinations of insecticide program efficacy, timing, and spatial extent of protection. Fifty scenarios simulating combinations of the above variables converged into only nine different cumulative defoliation scenarios (Hennigar et al. 2007). Given that protection essentially reduces defoliation, the Hennigar et al. (2007) results, presented as contour plots of spruce–fir harvest reductions as a function of defoliation scenario, outbreak severity, and percentage of landbase protected, can be used to infer effects of other possible outbreaks on our study area.

Prediction of defoliation impacts

The STAnd MANagement growth and yield model was used to forecast stand volume growth loss and increased mortality following SBW defoliation (MacLean 1996; Erdle and MacLean 1999). STAMAN is a stand table projection model, with tree growth and survival relationships derived from permanent sample plots, that has been calibrated for growth loss and mortality versus SBW defoliation relationships (Erdle and MacLean 1999). STAMAN calculates periodic defoliation-caused host mortality (salvageable volume), indirect non-host tree response (Erdle 1989; Hennigar et al. 2007), and ingrowth of regenerating cohorts. We reprogrammed STAMAN to accept inputs of defoliation by host species (Fig. 1) and to calculate volume differences between undefoliated and defoliated yields by tree species.

Stand growth for each of over 11 000 forest inventory plots (stand tables) in New Brunswick was simulated using STAMAN with and without defoliation under four outbreak-protection scenarios (Figs. 1*a*–1*d*). Percent volume reductions (differences in live volume and periodic mortality projections with and without defoliation) by species were aggregated into 72 stand impact classes and averaged. Impact classes were based on (*i*) species — percentage of balsam fir, white or Norway spruce, red or black spruce, in combination with nonhost shade-intolerant and shade-tolerant hardwood, or other softwood; (*ii*) age — mature (≥40 years old) or immature; and (*iii*) management — whether the stand was intensively managed (planted or thinned). This resulted in a table of relative volume impacts (percentage of live or salvageable volume multipliers) averaged by stand impact class, species, time since outbreak initiation, and outbreak-protection scenario, referred to as a stand impact matrix (Erdle 1989; MacLean et al. 2001; Hennigar et al. 2007).

The stand impact matrix was incorporated into the forest estate model. A dynamic linkable library was developed using C++ and referenced from Woodstock to allow real-time assignment of stand impact class to each development type based on species volume composition, maturity, and management at time of outbreak initiation (Table 1). In Woodstock, area constraints were formulated to force the assignment of stand impact class in the first planning period (2007–2011) to susceptible areas when an outbreak occurred. By assigning an outbreak-protection scenario and stand impact class to each development type, base (undefoliated) yields could be linked to the stand impact matrix to calculate forest live and salvageable growing stock available for harvest under SBW defoliation and protection scenarios.

Prediction of salvageable volume and stand deterioration

Tree mortality during a SBW outbreak generally begins 5–7 years following the onset of severe defoliation and is nearly complete after 10–12 years (MacLean 1980). In the J.D. Irving, Limited Woodstock model, each period or model iteration represented 5 years. Therefore, salvageable volume was calculated by period for each stand and defoliation scenario from 2007 to 2057. Since defoliation of >40% was specified in the scenarios to begin in 2011 (Fig. 1), the volume of dying trees would be concentrated near the end of the 2012–2016 period and start of the 2017–2021 period. Therefore, the volume dying in 2012–2016 was assumed to be salvageable as mostly live volume, whereas in 2017–2021 it was assumed to be dead or dying volume. The volume dying in 2017–2021 was considered to be inoperable from 2022 to 2026, since most died before 2020 and the volume would have deteriorated (Sewell and Maranda 1979).

While the stand impact matrix quantifies direct SBW impacts on stand volume, there can be additive indirect effects from increased susceptibility to windthrow from canopy disruption, pathogens, and other insects during or following the outbreak (Raske 1980; Taylor and MacLean 2007). For 106 spruce–fir and fir–spruce permanent sample plots >50 years old in northern New Brunswick, measured from 1985 to 1993, the probability of tree mortality nearly doubled 3–5 years following cessation of moderate to severe defoliation (>30%) compared with stands with less or no defoliation (Taylor and MacLean 2007). Wind-related mortality peaked at 6.5 merchantable $m^3·ha^{-1}·year^{-1}$ 11–15 years following cessation of moderate to severe defoliation, and ranged from 5 to 10 $m^3·ha^{-1}·year^{-1}$ for most of the 20 years following cessation of severe defoliation (Taylor and MacLean 2009). Given these reported rates of stand decline following outbreak, typically only 100–200 $m^3·ha^{-1}$ in such stands, we assumed that mature (≥40 years old) highly susceptible (≥75% host volume and ≥50% host composed of fir) stands severely impacted (>50% host volume loss) from defoliation would succumb to near total loss of merchantable volume 10–15 years post outbreak. We simulated stand breakup by constraining stands with significant canopy disruption (assumed here as >50% host volume reduction from defoliation) so they transitioned to advanced regenerating conditions (same stand type without SBW impact at 15 years old) 15 years following cessation of moderate to severe defoliation (Baskerville 1975). The effects of these stand breakup and transition assumptions on harvest and C were explored via sensitivity analyses.

Timber supply and spruce budworm impact management

In the event of a SBW outbreak, it is probable that J.D. Irving, Limited will endeavor to minimize forest harvest losses from defoliation. Thus, we modified the objective function of Woodstock to minimize the defoliation-caused harvest reductions from base (undefoliated) levels, as described by Hennigar et al. (2007). The solution attained was then resolved to maximize discounted harvest volumes with the lower spruce–fir harvest bound set equal to levels attained during the previous run. Specific methods used here, including objective equations, solve steps, salvage, and protection treatments, were discussed in detail for the same forest by Hennigar et al. (2007). Nondeclining harvest restrictions were removed from the model until 2022, as this constraint was impossible to maintain with an outbreak. Harvest flows of hardwoods and non-host softwood were constrained to vary <5% in any one period from base levels, for all scenarios. All other constraints (e.g., reserves, stand operability, old-growth retention) in the forest estate model were retained. The amount of susceptible area protected (0%–100%) was restricted using area constraints formulated in Woodstock.

Forest carbon pools and spruce budworm impacts

The C Budget Model for the Canadian Forest Sector (CBM-CFS3; Kurz et al. 2009) was used to convert stand projections of merchantable volume ($m^3 \cdot ha^{-1}$) contained in the J.D. Irving, Limited Woodstock model into carbon per hectare stored in (*i*) living biomass (stem wood, foliage, stumps, branches, bark, coarse and fine roots), (*ii*) standing deadwood (stems, branches), and (*iii*) forest floor deadwood pools (litter, forest floor and soil detritus, downed coarse woody debris). Stand C projections for each of the above three aggregated pools were represented as age-dependent yields in Woodstock, allowing simulation or optimization of C in the forest. Detailed descriptions of CBM data inputs, parameters and calibration (Kurz et al. 2009), and methods to develop stand projections of C in live biomass and dead organic matter for use in forest estate models (Neilson et al. 2008) have already been presented. Our use of CBM-CFS3 to generate on-site C projections followed methods described by Hennigar et al. (2008*a*).

To quantify forest C impacts from defoliation, all current Black Brook District area data on stand type, age classes, and their associated merchantable volume yields with and without defoliation impacts were imported into CBM-CFS3 and simulated for 80 years. CBM-CFS3 allows hardwood and softwood volume projections in stands to be modeled separately but cannot differentiate softwood host and non-host components. Therefore, species-specific impacts from the stand impact matrix were applied to undefoliated yields prior to importing into CBM-CFS3 to create defoliated yields by outbreak-protection scenario for each stand-type × age class combination. Percent differences between stand C projections with and without defoliation for each stand type, age class, and outbreak-protection scenario were calculated for live biomass, standing deadwood, and forest floor C pools, averaged by stand impact class and represented as a stand C impact matrix (time-dependent stand C pool % multipliers) in Woodstock. Representing stand C yields and a SBW C impact matrix in Woodstock significantly reduced simulation time to calculate forest C budgets, compared with simulating forest development totally in CBM-CFS3, and more importantly, allowed treatment scheduling decisions to be replanned optimally to minimize harvest or C losses from budworm effects.

Accounting for carbon in wood products

The C contained in merchantable log products exported from forest to mills was accounted for independently of CBM-CFS3 through direct conversion of harvest volume (softwood log, hardwood log, or pulpwood) to C (m^3 × specific gravity of species x × 50% C per oven-dry mass of product). A wood product C fate model — Carbon-Object Tracker (CO_T; Hennigar et al. 2008*a*) — was used to simulate product C transfer to alternative pool states through time, e.g., C in harvested roundwood, conversion to wood products, and transfer to landfills. Mill roundwood utilization statistics and wood product retention and landfill decay rates for parameterization of CO_T were adapted from the Forest Product Sector model extension of CBM-CFS2 (Apps et al. 1999) by Hennigar et al. (2008*a*). The fate of C stored in products through manufacturing and use, deposition in landfills and release to atmosphere was tracked separately for construction lumber, other lumber (flooring, particle board, panels), and paper products. Methane emissions from wood and paper decay in landfill were excluded, as was the potential for displacing coal and natural gas emissions from using wood waste or methane at landfills for energy.

One tonne of C for each raw product was simulated in CO_T for 200 years, allowing C storage over time in paper, lumber, and landfill pools to be tracked and represented as separate harvest product life-cycle yields in Woodstock (Hennigar et al. 2008*a*). Since the quantity of C stored in wood product pools depends on harvest level, and since SBW impact on harvest volume was already considered with integration of the stand impact matrix, SBW off-site C impacts could be tracked in this analysis and minimized using optimization.

Results

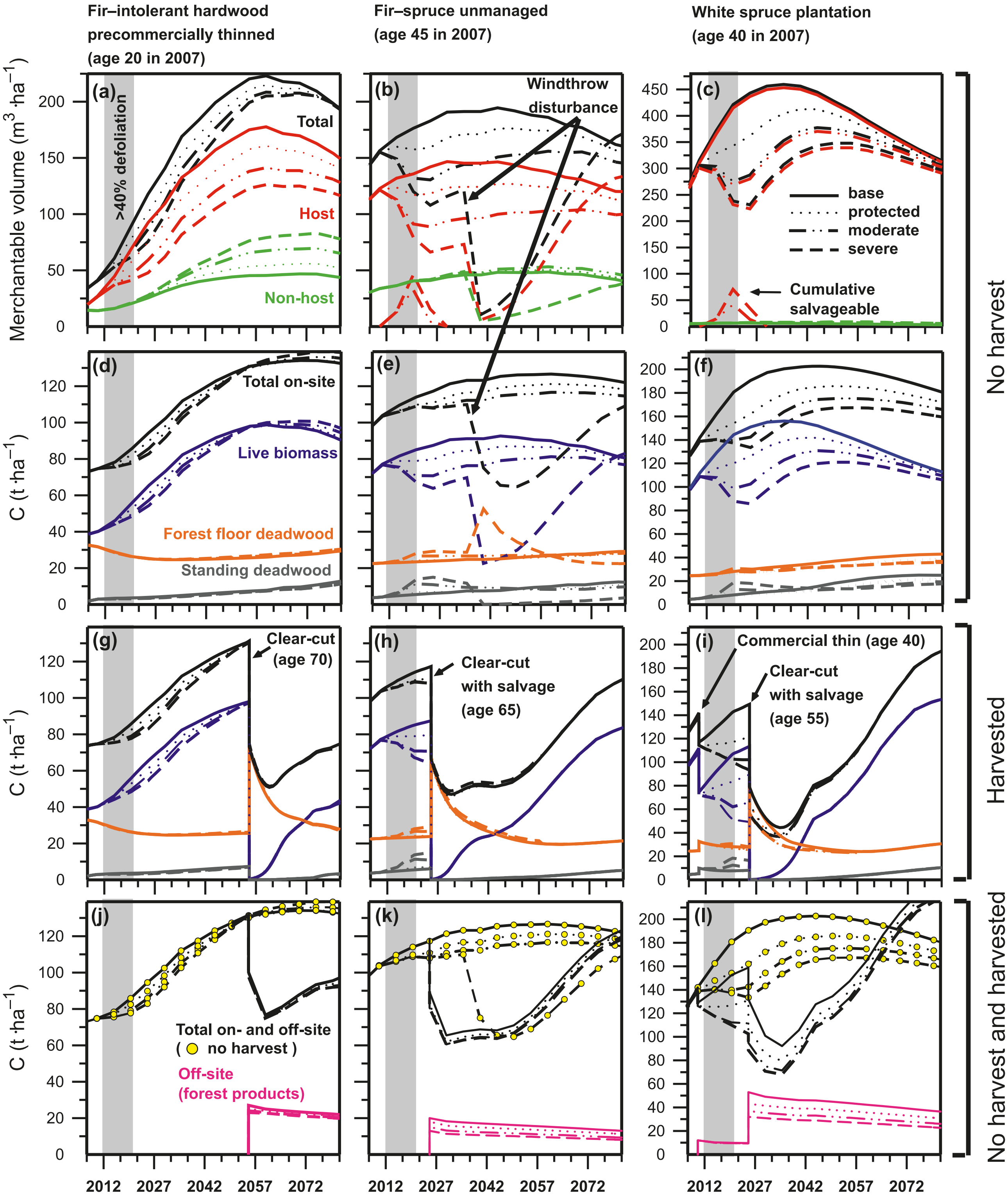
Stand dynamics during spruce budworm outbreaks

Stand volume and C dynamics, under moderate and severe SBW outbreak scenarios, with and without protection and harvesting, are shown for three example stand types in Fig. 2. Simulated host volume reductions from base (undefoliated) yield projections, 15 years following moderate and severe outbreaks, were 46% and 64% for mature (defined here as ≥40 years old) balsam fir stands, 34% and 51% for mature white spruce plantations, and 28% and 34% for immature (<40 years old) black spruce plantations, respectively (Figs. 2*a*–2*c*). Similar volume reductions under scenarios specifying application of foliage protection (Figs. 1*c* and 1*d*) ranged from 8%–20% (Figs. 2*a*–2*c*).

As a result of simulating lower white, red, and black spruce defoliation relative to balsam fir (Hennigar et al. 2008*b*), host volume reductions from base yields were 5%–10%, 10%–15%, and 15%–20% less for fir–spruce, white spruce, and black spruce stands, respectively, than were reported by Hennigar et al. (2007) for the same forest and outbreak scenarios. The 2007 inventory data showed that 29% of the host volume in black spruce plantation was balsam fir and was projected to increase to 45% by 2025 under base conditions, suggesting that these plantations could experience significant SBW impacts.

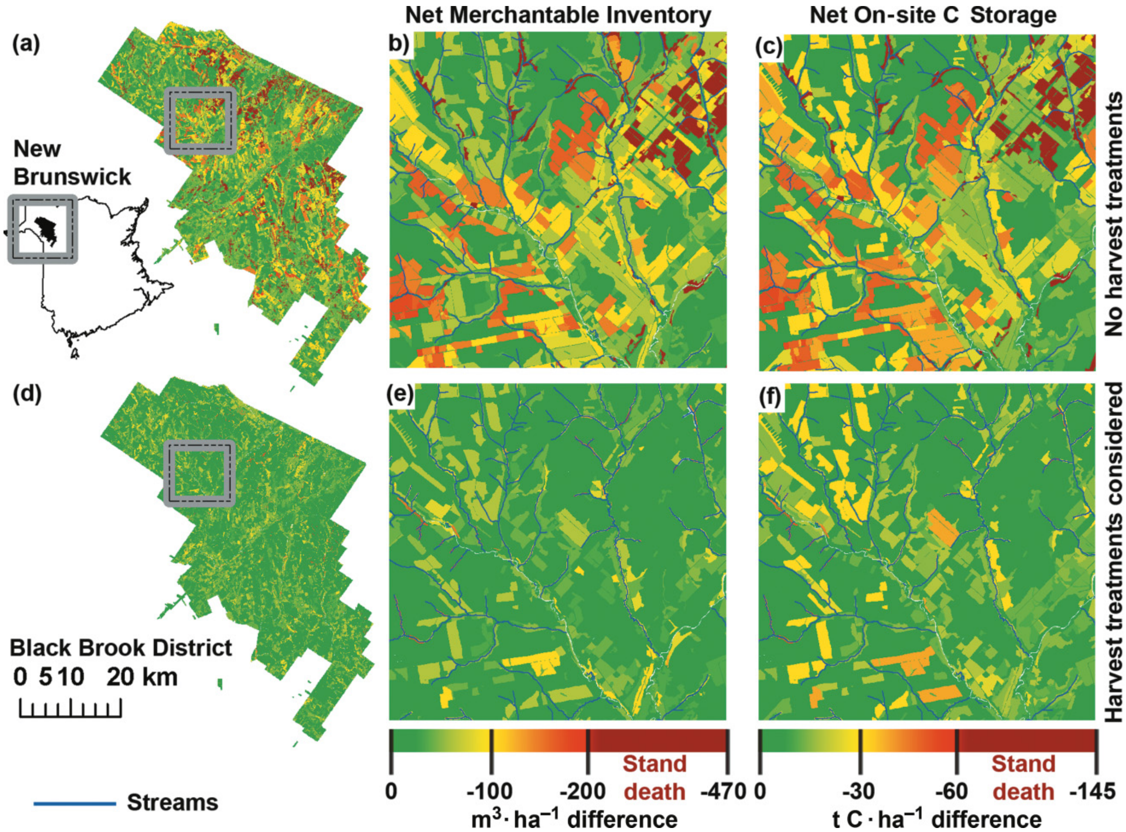
For stands with >75% host volume (96 300 ha; Table 1), the mean C sequestered in living biomass (stem wood, foli-

Fig. 2. Inventory of merchantable cubic volume (*a–c*) and on- and off-site carbon (C) dynamics (*d–l*) without (base) and with spruce budworm defoliation (for moderate and severe outbreak scenarios beginning in 2007 and foliage protection with biological insecticide described in Fig. 1) for three example stand types in the Black Brook District. Projected secondary wind disturbance impacts, 15-years postoutbreak (e.g., *b*, *e*, and *k*), were simulated to occur in stands with severe canopy disruption (≥40 years old at time of outbreak and ≥50% volume loss) from spruce budworm.



age, stumps, branches, bark, coarse and fine roots) from 2007 to 2026 was reduced by 0.95 and 1.42 t·ha^{-1}·year^{-1}, respectively, under moderate and severe SBW outbreak scenarios, but this declined to only 0.49 t·ha^{-1}·year^{-1} with foliage protection. The mean 2007–2026 living C reduction from a simulated severe SBW outbreak ranged from 0.39 to

Fig. 3. Black Brook District volume (*a*, *b*, *d*, *e*) and carbon (C) inventory (*c*, *f*) impacts per hectare in 2027 following a severe spruce budworm outbreak, without harvest activities (*a*–*c*) and with harvest and salvage activities (*d*–*f*) optimized to minimize base spruce–fir harvest losses.



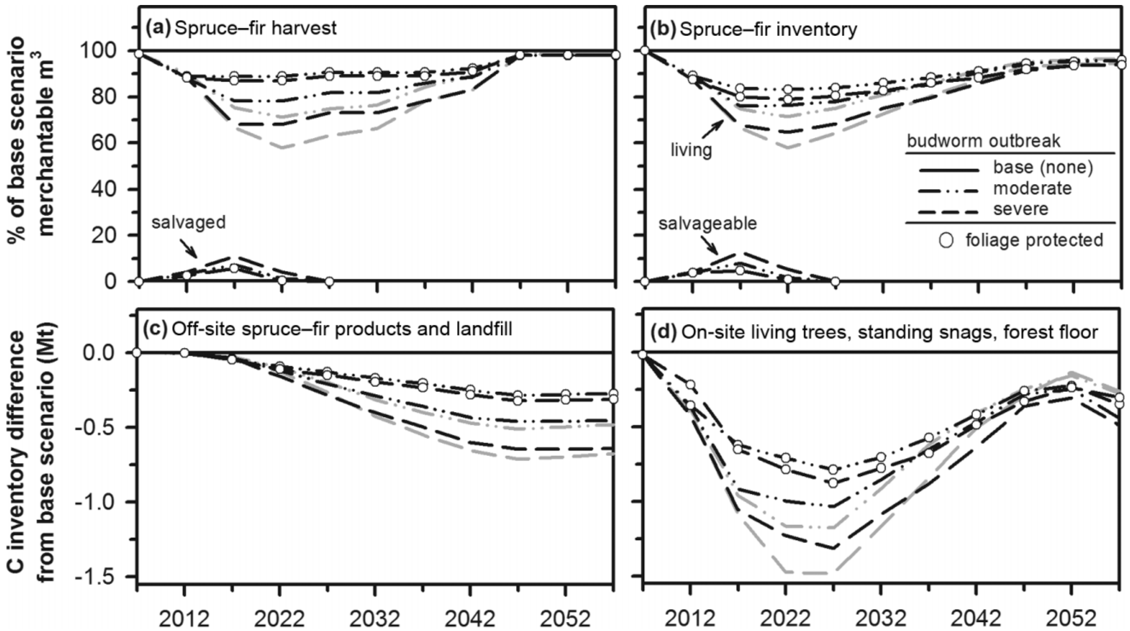
11.45 t·ha^{-1}·year^{-1} for spruce plantation, and averaged 1.1 t·ha^{-1}·year^{-1} for naturally regenerating stands. Non-host volume increases resulting from reduced stand competition from host SBW-caused mortality (Figs. 2*a*–2*c*) averaged 32% (range 6%–58%) and 48% (range 29%–67%) 15 years after moderate and severe outbreaks in immature host-dominated stands with 25%–50% shade-intolerant hardwood (e.g., Fig. 2*a*), which offset some C impacts from SBW (Fig. 2*d*). This effect was augmented by lower specific gravity of spruce (≈0.37) and fir (0.33) compared with shade-intolerant (poplar, 0.35; white birch, 0.48; red maple, 0.49) and shade-tolerant hardwood (≈0.55) species (Green et al. 1999). The 8% living biomass C reduction (Fig. 2*d*, defoliated versus undefoliated yields 15-years postoutbreak) was substantially less than the 24% total volume reduction (Fig. 2*a*). However, <2% of the susceptible area was projected to yield >10 m^3·ha^{-1} of additional non-host volume 15-years postoutbreak (e.g., Figs. 2*b* and 2*c*) as a result of host damage from SBW.

In mature unmanaged fir–spruce and white spruce plantation stand projections, 15%–25% of trees died from SBW defoliation in unprotected scenarios, resulting in large transfers of living biomass C to deadwood and forest floor C pools (Figs. 2*e* and 2*f*). SBW-caused mortality was nil with foliage protection, was generally low (<10% of base yield) for stands <40 years old, and was highest in mature stands at 10–90 m^3·ha^{-1} (salvageable volume, Figs. 2*b* and 2*c*). Direct host mortality from SBW subsided by 2022; however, growth loss continued until all foliage age classes recovered (2026).

For harvesting (clear-cut and salvage) scenarios, forest C inventory loss from SBW disturbance increased over the short term (*i*) by transferring C stored in merchantable volume (living or salvageable) off-site and (*ii*) by increasing forest floor temperature from increased light, which will increase the decay rate of remaining branches, foliage, submerchantable components, and roots (Kurz et al. 2009; Figs. 2*g*–2*i*). In the long term (>20 years), harvesting regenerated SBW-damaged stands, thereby increasing stand mean C sequestration rates and transfer of merchantable C, which would otherwise die and decay from continued defoliation, into wood products (Figs. 2*j*–2*l*). At harvest, about 35% ± 10% of biomass C was transferred to off-site storage, and 200 years later, 40%–50% of off-site C is projected to remain in landfills (Micales and Skog 1997; Hennigar et al. 2008*a*).

By 2027–2031, the impacts of SBW on spruce–fir volume and on-site C were proportionally similar with or without harvest disturbance simulated, as illustrated by similar color ranking of spruce–fir volume and on-site C impacts in Fig. 3*b* versus Fig. 3*c* and in Fig. 3*d* versus Fig. 3*e*. Spruce–fir volume impact was slightly higher than on-site C impact for mature stands (Fig. 3*b* versus Fig. 3*c*), because C was buffered by an increased transfer to snag and forest floor pools, and C impact was slightly higher than volume for young host plantations, where impact was primarily growth loss and non-host response was nil. The results indicate that prioritization for foliage protection by volume impact (MacLean et al. 2001) at the stand level would also minimize on-site C reduction.

Fig. 4. Differences in (*a*) spruce–fir harvest, (*b*) spruce–fir inventory, (*c*) off-site spruce–fir wood product carbon (C), and (*d*) on-site forest C storage compared with base levels (no defoliation), for moderate and severe budworm outbreak scenarios, with (◯) and without (no symbol) foliage protection applied to 40% of the susceptible landbase. Percentage of dead or dying trees salvaged (*a*) and salvageable (*b*) is shown. Grey lines indicate defoliated scenarios without harvest replanning and salvage.



Postoutbreak wind disturbance effects

Despite large potential stand impact effects of postoutbreak wind disturbance (e.g., Fig. 2*b*), the most highly vulnerable stands, projected to lose >170 m^3·ha^{-1} from SBW defoliation or wind by 2027–2031, were scheduled for harvest from 2007 to 2026 (e.g., Fig. 3, the areas in panel *e* that show less impact than those in panel *b* indicate the stand was clear-cut and regenerated). Thus, for scenarios explored, simulated secondary wind disturbance had little effect on spruce–fir harvest, as areas disturbed were few (0 ha for moderate and 13 900 ha for severe outbreaks) and harvested before secondary disturbance occurred (2027–2031; e.g., Figs. 2*k* and Figs. 3*d*–3*f*). In contrast, 28% of the 10 000 ha restricted from harvest (due to water course buffers and unique sites) was projected to die and regenerate without foliage protection, from the combined effects of a severe outbreak and subsequent wind disturbance by 2027–2031, resulting in on-site emissions of 3.75 t C·ha^{-1}·year^{-1}. Direct effects of defoliation caused on-site C of water course buffers and unique sites to be reduced by 11.3 t·ha^{-1} by 2031, and by 28.7 t·ha^{-1} when both direct SBW and indirect wind effects occurred. Although wind disturbance effects on stand on-site C are large in some stand types, forest-level effects averaged only 0.015 t·ha^{-1}·year^{-1} from 2007 to 2031, as the majority of these SBW–wind susceptible areas were harvested before secondary wind disturbance occurred.

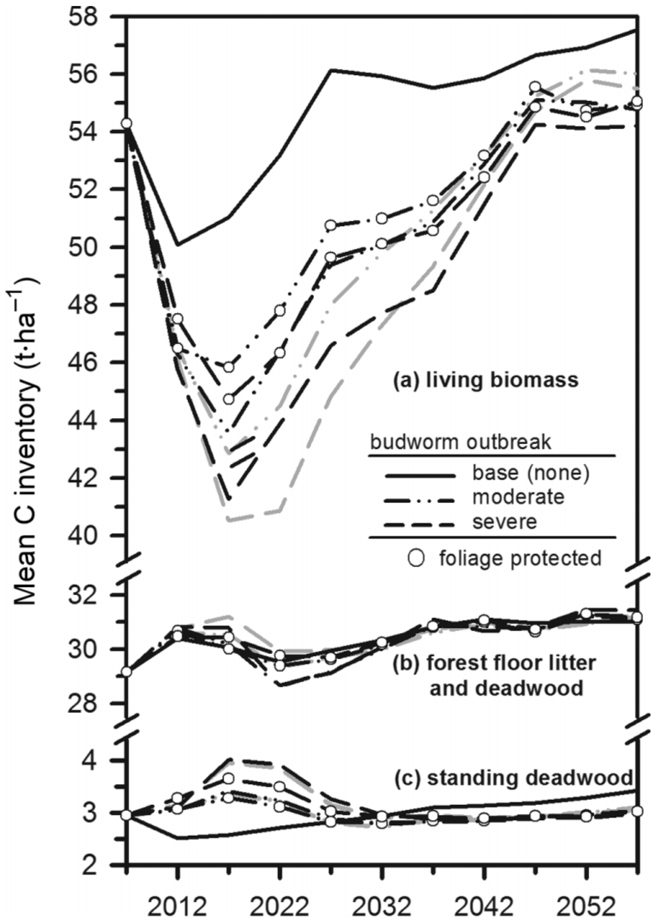
Forest dynamics under spruce budworm outbreak and management scenarios

Using the base harvest schedule and no foliage protection, maximum spruce–fir harvest reductions from base levels from 2022 to 2026 were 29% and 42%, respectively, for moderate and severe outbreak scenarios (Fig. 4*a*). Harvest reductions declined to 22% and 32% when reoptimized harvest scheduling with salvage (hereafter referred to as "replanning") was used to minimize harvest losses. Salvage from 2012 to 2022 contributed up to 16% of volume harvested during the peak of the outbreak (Figs. 4*a* and 4*b*). Spruce–fir harvest and inventory and total on-site C did not recover to base levels postoutbreak until about 2050 (within <5%) with or without replanning and or foliage protection (Figs. 4*a*, 4*b*, and 4*d*). By 2046, cumulative base harvest reduction from moderate and severe outbreaks amounted to 4.8 and 6.8 Mm^3, respectively, resulting in cumulative reductions of off-site C storage in products and landfills of 0.53 and 0.71 Mt (Fig. 4*c*). Off-site C reductions persisted to 2087 but were reduced to <10% with replanning (Fig. 4*c*).

The maximum on-site C storage reduction from base levels (12.4 Mt in 2027 for 139 500 ha), following moderate and severe SBW outbreaks, peaked in 2027 at 1.17 and 1.48 Mt, respectively, using the base harvest schedule, and declined to 1.03 and 1.31 Mt, respectively, with replanning (Fig. 4*d*). By 2017, the mean C stored in living biomass for the 139 500 ha of susceptible forest decreased to 40.5 t·ha^{-1} (–21%) for a severe outbreak scenario with no foliage protection or replanning, compared with base levels of 51 t·ha^{-1} (Fig. 5*a*). In comparison, mean standing deadwood and forest floor C increased to 3.9 (+50%) and 31.2 t·ha^{-1} (+4%), respectively, compared with base levels of 2.6 and 30 t·ha^{-1} (Figs. 5*b* and 5*c*). By 2032, standing deadwood resulting from SBW had mostly transferred to the forest floor (e.g., Fig. 2*e*), decayed, or was salvaged, and had returned to base C levels (Fig. 5*c*).

Applying balsam fir defoliation levels to all spruce species under a severe outbreak scenario, based upon the 1970s–1980s SBW outbreak on Cape Breton Island, Nova Scotia (MacLean et al. 2001, 2002), increased the maximum

Fig. 5. Mean on-site carbon (C) inventory for spruce-budworm-susceptible area (139 000 ha) from 2007 to 2057, divided into (*a*) living biomass, (*b*) forest floor litter and deadwood, and (*c*) standing deadwood pools. Scenarios include base (undefoliated) and moderate and severe budworm outbreaks, with (◯) and without (no symbol) foliage protection applied to 40% of the susceptible landbase. Grey lines indicate defoliated scenarios without harvest replanning and salvage.



harvest reduction in 2022–2026 from 42% to 56% and decreased the mean 2007–2026 on-site C sequestration of susceptible forest from –0.42 to –0.52 t·ha^{-1}·year^{-1}, versus the base rate of 0.07 t·ha^{-1}·year^{-1}. By 2046, following a severe outbreak scenario, spruce–fir base harvest reductions totaled 2.27 Mm^3 with fir defoliation levels applied to all hosts but were 1.25 Mm^3 less when reduced spruce defoliation (Hennigar et al. 2008*b*) was assumed in all outbreak years.

Effects of foliage protection on harvest and C sequestration

SBW outbreak scenarios that included harvest replanning and foliage protection applied to 40% of the susceptible area resulted in only a 13% reduction of the 2022–2026 spruce–fir base harvest (4.7 m^3·ha^{-1}·year^{-1}) versus 22% and 32% reductions for moderate and severe outbreak scenarios with replanning and no foliage protection (Fig. 4*a*). By combining replanning with foliage protection on 40% of the susceptible area, the 2007–2041 cumulative harvest reductions declined by 2.4 Mm^3 (50%) for moderate and 4.0 Mm^3 (60%) for severe outbreak scenarios, compared with the losses with no replanning or protection. While replanning reduced base off-site C storage loss to <10%, replanning with foliage protection applied to 40% of susceptible area during moderate and severe outbreak scenarios reduced 2041 base off-site C impacts by 0.23 Mt (48%) and 0.37 Mt (56%), respectively (Fig. 4*c*), and reduced maximum base on-site C impacts in 2022–2026 by 0.33 Mt (29%) and 0.58 Mt (41%), respectively (Fig. 4*d*). Living biomass C for susceptible forest was lowest at 50 t·ha^{-1} in 2012 for the base scenario and declined to 40.5 and 44.7 t·ha^{-1} in 2017 when a severe outbreak occurred without and with pest management (replanning and protection of 40% of susceptible area), respectively (Fig. 5*a*).

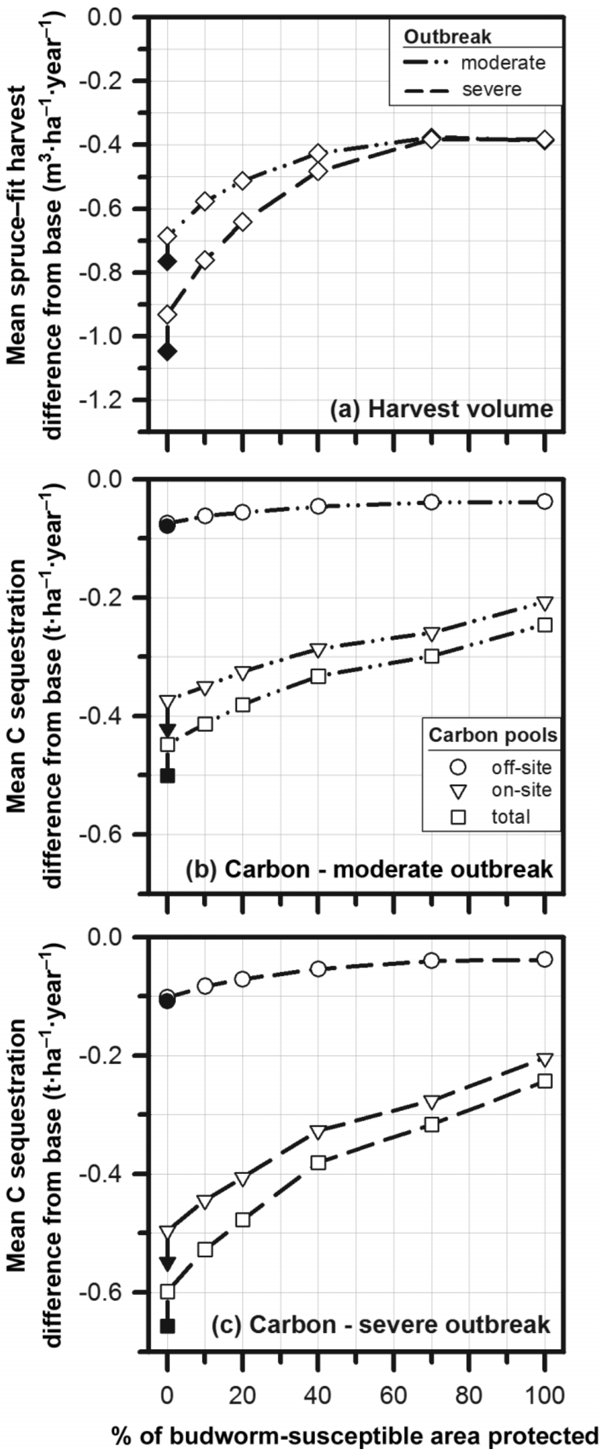
The marginal benefits of foliage protection on both spruce–fir harvest and off-site C declined progressively as the area protected increased (Fig. 6*a*). Since protection was scheduled optimally to minimize peak spruce–fir harvest losses, the stands protected first were those projected to yield the highest marginal harvest benefits (Erdle 1989; Hennigar et al. 2007). Of the total possible spruce–fir harvest and off-site C impact reductions achievable through replanning and protection of 100% of the area over the 2007–2026 impact period, reductions of 49% ± 5%, 66% ± 5%, and 88% ± 3% were achieved by replanning and protecting 10%, 20%, and 40% of susceptible area, respectively (Figs. 6*a*–6*c*).

However, the relationship of on-site C to area protected differed from that for harvest level. With replanning, increasing the susceptible area that was protected from 0% to 100% resulted in linear increases to the mean 2007–2026 on-site C sequestration rates of 0.18 t·ha^{-1}·year^{-1} for moderate outbreak scenarios and 0.29 t·ha^{-1}·year^{-1} for severe (Figs. 6*b* and 6*c*). On-site C sequestration rates increased with the area protected until 2021, remained constant from 2022 to 2026, and then declined thereafter, as this reserve of inventory was harvested to reduce maximum harvest losses from 2027 to 2041. Because virtually all protected stands were scheduled for harvest before 2041, the long-term benefits of foliage protection to reduce mean on-site C sequestration impacts were limited to restricted harvest zones, which were not harvested and were increasingly selected for protection as the extent of the area protected increased from 70% to 100% (Figs. 6*b* and 6*c*). With replanning, increasing the area protected from 0% to 100% increased the total (on-site plus off-site) mean 2007–2026 C sequestration from 0.41 to 0.51 t·ha^{-1}·year^{-1} for moderate outbreak scenarios and from 0.34 to 0.51 t·ha^{-1}·year^{-1} for severe. Of the total 2007–2026 mean C sequestration impact reduction possible from replanning with foliage protection applied to 40% and 100% of susceptible area, 20% and 14%, respectively, was attributed to off-site C dynamics.

Discussion

For the Black Brook District, assuming no salvage or foliage protection, the mean on-site C reduction from 2007 to 2027 for moderate and severe SBW outbreak scenarios was projected to be 0.42 and 0.53 t C·ha^{-1}·year^{-1} (1.17 and 1.48 Mt), respectively. In comparison, British Columbia's

Fig. 6. Mean annual spruce–fir harvest (*a*) and carbon (C) sequestration per hectare for (*a*, *b*) on-site, off-site, and total (on site + off site) pools for susceptible area (139 000 ha) from 2007 to 2027, expressed as difference from base levels (no defoliation; 0.0 line) for moderate (*a*, *b*) and severe (*a*, *c*) outbreak scenarios beginning in 2007 and with 0%–100% of susceptible area protected with biological insecticide. Solid symbols indicate no salvage or harvest plan amendments during an outbreak; open symbols indicate the opposite.



recent mountain pine beetle epidemic has been estimated to cause an impact of 0.36 t C·ha^{-1}·year^{-1} (270 Mt) across 37 Mha of forest from 2000 to 2020 (Kurz et al. 2008*b*), or 0.75 t·ha^{-1}·year^{-1} if only the susceptible pine area is considered (17.9 Mha; E. Neilson, Canadian Forest Service, personal communication, 2009). At its peak in 1975, the SBW outbreak extending across eastern Canada resulted in 50 Mha of moderate–severe defoliation (MacLean 2004). Our results and the historical periodicity and extent of past outbreaks quantitatively reinforce the magnitude of the effects a widespread regional SBW outbreak could have on Canada's forest C balance, as suggested by others (Fleming 2000; Kurz et al. 2008*a*). Even with harvest replanning and foliage protection applied, on-site C reductions from SBW were estimated at 0.2 t·ha^{-1}·year^{-1}.

Relative to New Brunswick's public forests (50% of total forest area; 2.99 Mha), the Black Brook District (210 000 ha) has proportionally similar (±1%) balsam fir and spruce volume composition at 18% and 36%, respectively (Erdle and Ward 2008), but has a higher proportion of white and Norway spruce species (50% compared with 11%). Extrapolating our on-site C results to New Brunswick's public forest area, we estimate mean C reductions of 20–25 Mt by the end of a moderate–severe SBW outbreak. These reductions in C sequestration plus increased emissions from decomposition of killed trees on only ∼3 Mha of public forests would translate to an increase in New Brunswick's total annual greenhouse gas emissions from all sources (2005 base year; Environment Canada 2008) of up to 20%, on average, over the next 20 years, and up to 40% (40–50 Mt) if private lands were included and assumed to have similar percentage host abundance and SBW vulnerability (6 Mha total forest area; MacLean et al. 2002). However, since New Brunswick's forests contain less white and Norway spruce relative to red and black spruce than does the Black Brook District, and given that all stands in New Brunswick may not experience severe defoliation in the next outbreak (e.g., Gray 2008), these extrapolated provincial C impact estimates may be high.

Using CBM-CFS3, Dymond et al. (2010) simulated moderate to severe SBW defoliation from 2011 to 2024 across 5.8 Mha of susceptible area for eastern Quebec and projected on-site base C reductions of 30 Mt. Lower relative C impacts simulated in Quebec compared with Black Brook may be attributed to 15% less balsam fir (most susceptible) and relatively more black spruce (least susceptible) in host stands and to a less severe defoliation outbreak simulated (Quebec — Gray et al. 2000).

The magnitudes of on-site versus off-site C inventory reductions from base scenario values in 2041 were generally similar, at 0.24–0.69 Mt for off-site and 0.40–0.63 Mt for on-site (Figs. 4*c* and 4*d*). Since forests regenerate following periodic SBW disturbance, we expect that if successive outbreaks were simulated, the projected base off-site C reductions would accrue and exceed base on-site C stock impacts during the second outbreak period. This premise assumes that global consumption of wood commodities is actually reduced following large timber supply deficits (production accounting approach) from SBW outbreak, and no compensatory harvest increase on other landbases occurs to fill consumer demand. In reality, reduced timber supply in one area must cause (*i*) an increase in harvest from other areas to satisfy mill and (or) consumer demand, and (or) (*ii*) a reduction in the consumption of wood products in place of less economical or efficient substitutes (steel, concrete), assuming that demand for commodities remains strong (e.g., Mogus et al. 2006). The first case would cause increased on-site C harvest from other forests (“leakage”; Murray et al. 2004) but reduced global off-site C impacts. For the second case, off-site C impacts would accrue, as modeled here, and may be exacerbated further by up to 0.25–1.0 t $C{\cdot}m^{-3}$ of lumber replaced by more fossil-fuel-demanding building alternatives such as steel or concrete (Petersen and Solberg 2003; Perez-Garcia et al. 2005). Additional indirect C emissions from product substitution would amount to $\approx$0.5–2.0 Mt C, if assumed for projected spruce–fir lumber production loss during a severe outbreak scenario for the Black Brook District. Thus, accounting for leakage or product substitution would increase the overall SBW C impacts reported here.

Harvest replanning with 40% of the susceptible area (139 500 ha) protected with insecticide during a severe outbreak reduced maximum base harvest impacts about 20%–30% more than base on-site C impacts, because spruce–fir inventory was always scheduled for harvest, if available, to minimize harvest impacts from SBW. Averaged from 2007 to 2041, base on-site C impacts were reduced by 26%–36% during moderate–severe outbreaks with replanning and foliage protection applied to 40% of the susceptible area. On the other hand, salvage and foliage protection reduced Black Brook maximum base harvest impacts up to 73%, which would buffer increased demand for wood products, and hence, on-site C loss from other areas.

Our modeling approach can aid in determining whether a forest-pest management C offset project is “additional”[2]. In the case of the Black Brook District, management places strong emphasis on timber production with a substantial component of intensive silviculture. The revenue generated from harvest and the original cost of silviculture (e.g., planting) far outweigh the foliage protection investment cost (\$50–\$100$\cdot ha^{-1}\cdot treatment^{-1}$). For this case, pest management reflects an activity to reduce risk within a larger project portfolio (silviculture investment) to increase growing stock for harvest. If foliage protection is not viable on economic or social grounds based on ratios of cost (\$, public relations) to benefit (\$, timber supply, old-growth conservation), cost could potentially be subsidized through tradable C credits, if the project activity can demonstrate long-term C sequestration benefits. In our analyses, marginal harvest benefits of foliage protection to reduce impacts from SBW declined to zero when >70% of the susceptible area was protected, yet reduction of forest C impacts continued to accrue as unharvested or restricted harvest zones were included as the area protected increased from 70% to 100% (Fig. 6). For this simplified cost–benefit case, assuming little or no harvest (i.e., economic) incentive to protect >70% of the susceptible area, C offsets generated from protecting the remaining 30% of the susceptible landbase should be considered additional. In reality, many more economic and social costs and benefits would need to be considered within this model framework to identify whether the pest management activity was additional or not.

Given the complex nature of SBW population dynamics (Royama et al. 2005; Eveleigh et al. 2007) combined with potential climate change effects (Gray 2008), future outbreak defoliation severity, duration, and spatial dynamics remain highly uncertain, and hence, projections of future SBW C impacts are also uncertain. Gray (2008) concluded that climate change is predicted to result in SBW outbreaks that are an average of approximately 6 years longer with an average of 15% greater defoliation. Use of the SBW DSS has always been proposed as a scenario planning “what if” approach; essentially “if this defoliation scenario unfolds, here are the projected effects on timber supply or harvest or C” (MacLean et al. 2001; Hennigar et al. 2007). During an actual SBW outbreak, empirical defoliation data for the past 5 years would be estimated from ground and aerial surveys, updated each year, and combined with sampled SBW population levels to develop reliable short-term impact assessments and benefits (e.g., t C, m^3, \$) of pest management activities. Since the last outbreak, precise aerial insecticide application using computer-controlled “boom-on/boom-off” technology has advanced techniques to precisely target specific areas for protection (D. Davies, Forest Protection Limited, personal communication, 2008). Advanced aerial application technology combined with an extensive Black Brook District road network would alleviate operational constraints and facilitate optimal implementation of forest harvest replanning and optimized protection.

Despite these aerial application advancements, reducing defoliation to 40% when projected defoliation is expected to be >70% may be unrealistic with either single or double applications of *Bacillus thuringiensis* based on efficacy models by Fleming and van Frankenhuyzen (1992) and Régnière and Cooke (1998). For this reason, generalized equations should be distilled from these existing *Bacillus thuringiensis* efficacy models to estimate cumulative defoliation for alternative SBW population forecasts and treatment options (applications per year, insecticide type). These population scenarios and aerial application options could then be represented in the forest model to better quantify cost–benefits (\$, m^3, C) of alternative protection programs with respect to landowner objectives.

As the severity of stand impacts caused by SBW increases, stand development will slow and delay economically operable conditions for harvest. Indirect SBW-caused harvest losses from postponed stand operability have been simulated to increase total harvest losses by as much as 40% in New Brunswick (MacLean et al. 2002). The period of stand operability delays from SBW defoliation, as well as changes to pulp:log ratios, are lacking in the stand impact matrix. However, development of such operability and product impact multipliers is straightforward, and linking these measures to forest estate models can be streamlined with collaboration between forest and pest model designers.

[2] Additional or additionality refers to the extent to which greenhouse gas mitigation is above and beyond what would have occurred in the absence of a given C-offset project (e.g., foliage protection to reduce defoliation-caused tree mortality and resulting C emissions from decay on-site).

We compared the SBW outbreak scenarios used in this paper with those used by others, in terms of outbreak duration and defoliation intensity. The respective values for outbreak duration, number of years of moderate–severe (≥30%) defoliation, and summed annual defoliation (≥30%) were as follows: 8 years, 8 years, and 620% defoliation (equivalent to removal of 6.2 age classes of foliage) for our moderate scenario; 10 years, 10 years, and 820% for our severe scenario; 14 years, 8 years, and 470% for actual Black Brook District defoliation data (Porter et al. 2004); and for scenarios representing five clusters of outbreak severities from Gray and MacKinnon (2006): 7 years, 6 years, and 505% defoliation for mild–moderate class outbreaks; 15 years, 11 years, and 727% for moderate; 18 years, 17 years, and 995% for moderate–severe; 25 years, 19 years, and 1268% for severe; and 28 years, 24 years, and 1661% for very severe scenarios. Thus our moderate scenario, which resulted in maximum spruce–fir harvest reductions of 29% from base levels from 2022 to 2026 and maximum on-site C storage reduction of 1.17 Mt (9%), was generally similar to the Porter et al. (2004) scenario and the Gray and MacKinnon (2006) moderate outbreak class. Gray and MacKinnon's moderate–severe outbreak class was similar to our severe scenario, which resulted in maximum spruce–fir harvest reductions of 42% from 2022 to 2026 and maximum on-site C storage reduction of 1.48 Mt (12%). Gray and MacKinnon's severe and very severe outbreak classes were of longer duration (18–28 years) and higher summed annual defoliation than the scenarios that we modeled, and would result in greater on-site C reductions,

Our results show that a severe SBW outbreak, simulated on the Black Brook District, was projected to cause total harvested volume, off-site C, and on-site C in 2027 to be reduced by 1.34 Mm^3, 0.26 Mt, and 1.48 Mt, respectively; and that these losses could be reduced by 73%, 56%, and 41%, respectively, with harvest replanning and salvage plus foliage protection applied to 40% of the susceptible area. Under current C accounting rules, potential future losses from natural disturbance will provide little incentive for management change (Böttcher et al. 2008). Our modeling approach permits projection of C dynamics with or without SBW disturbance and management, thereby allowing the effects of management on C storage to be evaluated independently of natural disturbance C impacts. Such approaches are increasingly desired by government and industry to (*i*) estimate potential fluctuation of forest values (e.g., harvest, C, habitat, profit) for plausible natural disturbance scenarios, (*ii*) identify cost-effective forest management designs to minimize such fluctuations, and (*iii*) assess whether a forest-C-offset project (e.g., foliage protection program) will lead to additional and long-term C storage in forests or forest products.

Acknowledgements

This project was funded or supported by the Canada Sustainable Forest Management Network Centre of Excellence, Canadian Forest Service Atlantic Forestry Centre, Atlantic Innovation Fund, Remsoft Inc., and J.D. Irving, Limited. Chris Hennigar was supported by a Natural Sciences and Engineering Research Council of Canada Industrial Postgraduate Scholarship.

References

Albani, M., Moorcroft, P.R., Ellison, A.M., Orwig, D.A., and Foster, D.R. 2010. Predicting the impact of hemlock woolly adelgid on carbon dynamics on eastern United States forests. Can. J. For. Res. **40**(1): 119–133. doi:10.1139/X09-167.

Amos-Binks, L.J., MacLean, D.A., Wilson, J.S., and Wagner, R.G. 2010. Temporal changes in species composition of mixedwood stands in northwest New Brunswick: 1946–2008. Can. J. For. Res. **40**(1): 1–12. doi:10.1139/X09-162.

Baskerville, G.L. 1975. Spruce budworm: super silviculturalist. For. Chon. **51**: 138–140.

Baskerville, G.L. 1995. The forestry problem: adaptive lurches of renewal. *In* Barriers and bridges to the renewal of ecosystems and institutions. *Edited by* L.H Gunderson, C.S. Holling, and S.S. Light. Columbia University Press, New York. pp. 39–102.

Blais, J.R. 1957. Some relationships of the spruce budworm, *Choristoneura fumiferana* (Clem.), to black spruce, *Picea mariana* (Moench) Voss. For. Chron. **33**: 364–372.

Böttcher, H., Kurz, W.A., and Freibauer, A. 2008. Accounting of forest carbon sinks and sources under a future climate protocol — factoring out past disturbances and management effects on age-class structure. Environ. Sci. Policy, **11**(8): 669–686. doi:10.1016/j.envsci.2008.08.005.

Carter, N.E., and Lavigne, D.R. 1994. Protection spraying against spruce budworm in New Brunswick 1993. New Brunswick Department of Natural Resources and Energy, Fredericton, N.B.

Clark, K.L., Skowronski, N., and Hom, J. 2010. Invasive insects impact forest carbon dynamics. Glob. Change Biol. **16**(1): 88–101. doi:10.1111/j.1365-2486.2009.01983.x.

Dymond, C.C., Neilson, E.T., Stinson, G., Porter, K., MacLean, D.A., Gray, D.R., Campagna, M., and Kurz, W. 2010. Future spruce budworm outbreak forecast to reduce C-sink of eastern Canadian forests. Ecosystems. In press.

Environment Canada. 2008. National inventory report 1990–2006: greenhouse gas sources and sinks in Canada. Environment Canada [online]. Available from http://www.ec.gc.ca/pdb/ghg/inventory_report/2006_report/tdm-toc_eng.cfm [accessed 18 January 2009].

Erdle, T.A. 1989. Concept and practice of integrated harvest and protection design in the management of eastern spruce fir forests. Ph.D. thesis, University of New Brunswick, Fredericton, N.B., Canada.

Erdle, T.A., and MacLean, D.A. 1999. Stand growth model calibration for use in forest pest impact assessment. For. Chron. **75**: 141–152.

Erdle, T.A., and Ward, C. 2008. Management alternatives for New Brunswick's public forest [online]. Department of Natural Resources, Fredericton, N.B., Canada. Available from http://www.gnb.ca/0079/ErdleReport-e.asp [accessed 8 March 2009].

Eveleigh, E.S., McCann, K.S., McCarthy, P.C., Pollock, S.J., Lucarotti, C.J., Morin, B., McDougall, G.A., Strongman, D.B., Huber, J.T., Umbanhowar, J., and Faria, L.D.B. 2007. Fluctuations in density of an outbreak species drive diversity cascades in food webs. Proc. Natl. Acad. Sci. U.S.A. **104**(43): 16976–16981. doi:10.1073/pnas.0704301104. PMID:17940003.

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

Fleming, R.A., and van Frankenhuyzen, K. 1992. Forecasting the efficacy of operational *Bacillus thuringiensis* Berliner applications against spruce budworm, *Choristoneura fumiferana* Clemens (Lepidoptera: Tortricidae), using dose ingestion data: initial models. Can. Entomol. **124**(6): 1101–1113. doi:10.4039/Ent1241101-6.

Gray, D.R. 2008. The relationship between climate and outbreak characteristics of the spruce budworm in eastern Canada. Clim. Change, **87**(3–4): 361–383. doi:10.1007/s10584-007-9317-5.

Gray, D.R., and MacKinnon, W.M. 2006. Outbreak patterns of the spruce budworm and their impacts in Canada. For. Chron. **82**: 550–561.

Gray, D.R., Régnière, J., and Boulet, B. 2000. Analysis and use of historical patterns of spruce budworm defoliation to forecast outbreak patterns in Quebec. For. Ecol. Manage. **127**(1–3): 217–231. doi:10.1016/S0378-1127(99)00134-6.

Green, D.W., Winandy, J.E., and Kretschmann, D.E. 1999. Mechanical properties of wood. Wood handbook: wood as an engineering material. USDA Forest Service, Forest Products Laboratory, Madison, Wisc. Gen. Tech. Rep. 113. pp. 4.1–4.45.

Hennigar, C.R., MacLean, D.A., Porter, K.B., and Quiring, D.T. 2007. Optimized harvest planning under alternative foliage-protection scenarios reduce volume losses to spruce budworm. Can. J. For. Res. **37**: 1755–1769. doi:10.1139/X07-001.

Hennigar, C.R., MacLean, D.A., and Amos-Binks, L.J. 2008*a*. A novel approach to optimize management strategies for carbon stored in both forests and wood products. For. Ecol. Manage. **256**(4): 786–797. doi:10.1016/j.foreco.2008.05.037.

Hennigar, C.R., MacLean, D.A., Quiring, D.T., and Kershaw, J.A., Jr. 2008*b*. Differences in spruce budworm defoliation among balsam fir and white, red, and black spruce. For. Sci. **54**: 158–166.

J.D. Irving, Limited. 2002. 2002–2026 forest management plan and 80-year habitat and wood supply analysis. J.D. Irving, Limited, Saint John, N.B., Canada.

Kettela, E.G. 1995. Insect control in New Brunswick, 1974–1989. *In* Forest insect pests in Canada. *Edited by* J.A. Armstrong and W.G.H. Ives. Natural Resources Canada, Canadian Forest Service, Ottawa, Ont. pp. 655–665

Kurz, W.A., and Apps, M.J. 1995. An analysis of future carbon budgets of Canadian boreal forests. Water Air Soil Pollut. **82**(1–2): 321–331. doi:10.1007/BF01182844.

Kurz, W.A., and Apps, M.J. 1999. A 70-year retrospective analysis of carbon fluxes in the Canadian forest sector. Ecol. Appl. **9**(2): 526–547. doi:10.1890/1051-0761(1999)009[0526:AYRAOC]2.0.CO;2.

Kurz, W.A., and Beukema, S.J. 2001. Refinement of the analysis of carbon stock changes in the managed forest in Canada (1980–2032). *Prepared by* ESSA Technologies Ltd., Vancouver, B.C., for the Canadian Forest Service, Natural Resources Canada, Ottawa, Ont., Canada.

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

Kurz, W.A., Webb, T.M., Apps, M.A., Banfield, E., and Metsaranta, J. 2003. Accounting for the influence of large scale natural disturbances in Canadian forest ecosystems. Science Focus Workshop draft report. Canadian Forest Service, Pacific Forest Research Center, Victoria, B.C., Canada.

Kurz, W.A., Stinson, G., Rampley, G.J., Dymond, C.C., and Neilson, E.T. 2008*a*. Risk of natural disturbances makes future contribution of Canada's forests to the global carbon cycle highly uncertain. Proc. Natl. Acad. Sci. U.S.A. **105**(5): 1551–1555. doi:10.1073/pnas.0708133105. PMID:18230736.

Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A.L., Ebata, T., and Safranyik, L. 2008*b*. Mountain pine beetle and forest carbon feedback to climate change. Nature, **452**(7190): 987–990. doi:10.1038/nature06777. PMID:18432244.

Kurz, W.A., Dymond, C.C., White, T.M., Stinson, G., Shaw, C.H., Rampley, G.J., Smyth, C., Simpson, B.N., Neilson, E.T., Trofymow, J.A., Metsaranta, J., and Apps, M.J. 2009. CBM-CFS3: a model of carbon-dynamics in forestry and land-use change implementing IPCC standards. Ecol. Model. **220**(4): 480–504. doi:10.1016/j.ecolmodel.2008.10.018.

MacLean, D.A. 1980. Vulnerability of fir–spruce stands during uncontrolled spruce budworm outbreaks: a review and discussion. For. Chron. **56**: 213–221.

MacLean, D.A. 1996. The role of a stand dynamics model in the spruce budworm decision support system. Can. J. For. Res. **26**(10): 1731–1741. doi:10.1139/x26-197.

MacLean, D.A. 2004. Predicting forest insect disturbance regimes for use in emulating natural disturbance. *In* Emulating natural forest landscape disturbances: concepts and applications. *Edited by* A.H. Perera, L.J. Buse, and M.G. Weber. Columbia University Press, New York. pp. 69–82.

MacLean, D.A., Erdle, T.A., MacKinnon, W.E., Porter, K.B., Beaton, K.P., Cormier, G., Morehouse, S., and Budd, M. 2001. The spruce budworm decision support system: forest protection planning to sustain long-term wood supply. Can. J. For. Res. **31**(10): 1742–1757. doi:10.1139/cjfr-31-10-1742.

MacLean, D.A., Beaton, K.P., Porter, K.B., MacKinnon, W.E., and Budd, M.G. 2002. Potential wood supply losses to spruce budworm in New Brunswick estimated using the spruce budworm decision support system. For. Chron. **78**: 739–750.

Micales, J.A., and Skog, K.E. 1997. The decomposition of forest products in landfills. Int. Biodeterior. Biodegrad. **39**(2–3): 145–158. doi:10.1016/S0964-8305(97)83389-6.

Mogus, A., Stennes, B., and van Kooten, G.C. 2006. Canada–U.S. softwood lumber trade revisited: substitution bias in the context of a spatial price equilibrium framework. For. Sci. **52**: 411–421.

Murray, B.C., McCarl, B.A., and Lee, H. 2004. Estimating leakage from forest carbon sequestration programs. Land Econ. **80**(1): 109–124. doi:10.2307/3147147.

Nealis, V.G., and Régnière, J. 2004. Insect–host relationships influencing disturbance by the spruce budworm in a boreal mixedwood forest. Can. J. For. Res. **34**(9): 1870–1882. doi:10.1139/x04-061.

Neilson, E.T., MacLean, D.A., Meng, F.-R., Hennigar, C.R., and Arp, P.A. 2008. Optimal on- and off-site forest carbon sequestration under existing timber supply constraints in northern New Brunswick. Can. J. For. Res. **38**(11): 2784–2796. doi:10.1139/X08-120.

New Brunswick Department of Natural Resources. 2005. Objectives and standards for the New Brunswick Crown forest [online]. Forest Management Branch, Department of Natural Resources, Fredericton, N.B., Canada. Available from http://www.gnb.ca/0078/publications/technical-vision-EN.pdf [accessed 4 April 2007].

Ostaff, D.P., and MacLean, D.A. 1989. Spruce budworm populations, defoliation, and changes in stand condition during an uncontrolled spruce budworm outbreak on Cape Breton Island, Nova Scotia. Can. J. For. Res. **19**(9): 1077–1086. doi:10.1139/x89-164.

Ostaff, D.P., and MacLean, D.A. 1995. Patterns of balsam fir foliar production and growth in relation to defoliation by spruce budworm. Can. J. For. Res. **25**(7): 1128–1136. doi:10.1139/x95-125.

Perez-Garcia, J., Lippke, B., Comnick, J., and Manriquez, C. 2005. An assessment of carbon pools, storage, and wood products market substitution using life-cycle analysis results. Wood Fiber Sci. **37**: 140–148.

Petersen, A.K., and Solberg, B. 2003. Substitution between floor constructions in wood and natural stone: comparison of energy consumption, greenhouse gas emissions, and costs over the life cycle. Can. J. For. Res. **33**(6): 1061–1075. doi:10.1139/x03-020.

Porter, K.B., Hemens, B., and MacLean, D.A. 2004. Using insect-caused disturbance patterns in northern New Brunswick to inform forest management. Chap. 16. *In* Emulating natural forest landscape disturbances: concepts and applications. *Edited by* A.H. Perera, L.J. Buse, and M.G. Weber. Columbia University Press, New York. pp. 135–145.

Raske, A.G. 1980. Damage caused by the spruce budworm. *In* Review of the spruce budworm outbreak in Newfoundland — its control and forest management implications. *Edited by* J. Hudak and A.G. Raske. Inf. Report N-X-205. Canadian Forest Service, Environment Canada, St. John's, Newfoundland. pp. 28–30.

Régnière, J., and Cooke, B. 1998. Validation of a process-oriented model of *Bacillus thuringiensis* Variety *kurstaki* efficacy against spruce budworm (Lepidoptera: Tortricidae). Environ. Entomol. **27**: 801–811.

Royama, T. 1984. Population dynamics of the spruce budworm *Choristoneura fumiferana*. Ecol. Monogr. **54**(4): 429–462. doi:10.2307/1942595.

Royama, T., MacKinnon, W.E., Kettela, E.G., Carter, N.E., and Hartling, L.K. 2005. Analysis of spruce budworm outbreak cycles in New Brunswick, Canada, since 1952. Ecology, **86**(5): 1212–1224. doi:10.1890/03-4077.

Schoemaker, P.J.H. 1995. Scenario planning: a tool for strategic thinking. Sloan Manage. Rev. **36**: 25–40.

Sewell, C.D., and Maranda, J. 1979. Problems in salvaging timber affected by the spruce budworm. FERIC, Vancouver, B.C., Canada. Tech. Rep. No. TR-25.

Sterner, T.E., and Davidson, A.G. 1982. Forest insect and disease conditions in Canada 1981. Canadian Forest Service, Ottawa, Ont., Canada.

Taylor, S.L., and MacLean, D.A. 2007. Spatiotemporal patterns of mortality in declining balsam fir and spruce stands. For. Ecol. Manage. **253**(1-3): 188–201. doi:10.1016/j.foreco.2007.07.016.

Taylor, S.L., and MacLean, D.A. 2009. Legacy of insect defoliators: increased wind-related mortality two decades after a spruce budworm outbreak. For. Sci. **55**: 256–267.