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Soil Organic Matter: A Sustainability Indicator for Wildfire Control and Bioenergy Production in the Urban/Forest Interface

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Many rural communities in British Columbia (western Canada) are increasingly at risk from wildfire as temperatures rise and droughts become more frequent. In addition, these communities are also faced with rising fuel costs and a growing demand for heat as their populations increase. The fact that these communities are surrounded by forests presents an opportunity to combine community wildfire risk abatement with bioenergy development. We show how the ecological model FORECAST was linked with GIS and economic models to create a freely available online tool (FIRST Heat) to help other communities make their own screening-level ecological assessments of combining wildfire risk control with district heating systems. The tool incorporates an ecological sustainability index based on the relative change in soil organic matter (SOM) after 50 yr of management compared with initial levels. Two thresholds were defined: 10% SOM lost (warning level) and 20% SOM lost (critical level). The tool was able to adequately capture the influences of ecological zone, stand age, site quality, and intensity of forest management on SOM losses. Stands in the sub-boreal and arid interior were significantly more exposed to SOM losses than in other ecological zones, as well as soils in old-growth forests. Stands in poor sites were significantly more sensitive to forest management than young and fertile sites. All things considered, our results show the suitability of incorporating ecological models and SOM thresholds in user-friendly decision-support tools to successfully transfer scientific knowledge on forest soils to local stakeholders and decision makers.

Abbreviations: BEC, biogeoclimatic; ESSF, Engelmann Spruce–Subalpine Fir; FIRST Heat, Fire Interface Rural Screening Tool for Heating; ICH, Interior Cedar–Hemlock; IDF, Interior Douglas-fir; MS, Montane Spruce; SBS, Sub-Boreal Spruce; SOM, soil organic matter.

Since Canada's census of 1981, there has been a clear trend for rural suburban spaces to grow in population at a rate higher than city centers (Hirsch and Fuglem, 2006). In British Columbia (western Canada), the area of the forest–urban interface has been steadily increasing in the last few years, becoming vulnerable to damage and evacuation orders as wildfires strike. All forests in interior British Columbia are naturally adapted to fire as an ecological disturbance with a specific fire regime: a combination of fire frequency, intensity, and severity (Johnson et al., 2001); however, external factors are changing natural fire regimes. For example, excessive fire suppression or increased tree mortality by pest attacks cause an accumulation of fuel and therefore increases the fire occurrence and/or intensity (Keeley et al., 1999; Jolly et al., 2012). Fire frequency has also increased

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from direct anthropogenic sources (escaped fires, sparks, etc.) that occur when more people live in the urban–forest interface. Direct links between climate change and more fires have also been reported (Westerling et al., 2006), and an increase in fire frequency is predicted for western North America (Hirsch and Fuglem, 2006; Nitschke and Innes, 2013).

Due to this increasing area under wildfire risk, communities around British Columbia are implementing preventive forest management to reduce the wildfire risk. These activities are generating woody debris from stand density reduction. Until now, this biomass was removed from the site and then burned in piles so it would not fuel future wildfires. However, rural communities now incur fuel prices more expensive than in cities because fossil fuels have to be transported from the main population centers. Therefore, there are economic as well as environmental reasons to increase the use of woody debris as a source of energy generated from the management of the forest–urban interface.

The ecological context of the biomass residue is an important consideration, however, because removing harvesting residues could have negative impacts on fauna (Sullivan et al., 2011) and flora (Blanco, 2012). Long-term nutrient losses have been described as important for altering ecosystem productivity (Blanco et al., 2005, 2012; Wang et al., 2013). Maintaining ecosystem productivity is fundamental to the principles of sustainable forest management. However, there is still no effective method for measuring and monitoring the impacts of management activities on ecosystem productivity (Fox, 2000). Forest productivity, like soil or site quality, is a value-based concept related to the objectives of ecosystem management and hence will be management- and ecosystem-dependent (Schoenholtz et al., 2000).

Total soil organic matter (SOM) has been linked to ecosystem productivity, being recommended for monitoring programs (Peng et al., 2002, Seely et al., 2010). Soil organic C is commonly recognized as one of the key parameters of soil quality, yet quantitative assessments of its contribution to soil quality are often lacking. Through its role in aggregate stability, SOM influences the soil porosity and thus gas exchange reactions and water relationships. Soil organic matter is also a critical pool in the C cycle and a repository of nutrients, and through its influence on many fundamental biological and chemical processes it plays a pivotal role in nutrient release and availability (Henderson, 1995; Nambiar, 1996). The importance of SOM as a structural and functional component of soil productive capacity and in providing the critical linkage between management and productivity is widely recognized for forest soils (Henderson, 1995; Nambiar, 1996).

Sustainable forest management requires maintenance of the soil resource, including its biological, chemical, and physical properties and processes. Of the highest priority is to quantify the practical consequences of the changes in SOM that are important for sustainable forestry. Defining qualitative criteria for SOM is further hampered by the fact that the same C content in soil translates into different soil productive capacities (Schoenholtz et al., 2000) depending on soil types, climatic regions, land use, and species composition (Doran and Parkin, 1994).

If the anticipated economic rotation for a particular site is shorter than the associated ecological rotation, a decline in ecosystem productivity will result (Kimmins et al., 2010). However, because of the buffering capacity provided by the nutrient capital stored within SOM, the decline may be subtle and not easily detected after a single rotation. Evidence from ecosystems subject to long-term harvesting (plantations in the southern United States [Fox, 2000], New Zealand [Sims et al., 1988], Australia [Johnson, 1992], China [Bi et al., 2007], Spain [Blanco et al., 2005], and South Africa [Evans, 1996]) supports this contention but also indicates that the decline in productivity will be cumulative and increase nonlinearly across multiple rotations.

Therefore, the challenge for planners and local managers in rural British Columbia is how to keep their communities safe while reducing energy costs by using the forest biomass produced from sustainable forest management. Together, these issues make a complex picture as communities struggle to realize the potential of their forest–urban interface areas. A joint project involving the University of British Columbia, Community Energy Association, and Wood Waste to Rural Heat Project has developed a tool called the Fire Interface Rural Screening Tool for Heating (FIRST Heat). The tool uses SOM as an indicator of long-term ecological impacts and provides a "traffic light" index to help forest managers and community planners identify the situations in which biomass removal could produce undesirable long-term ecological consequences. In designing this tool, we have tried to address four of the 10 principles recently identified by Mead and Smith (2012) as basic to managing nutrient cycling in forest biomass production for energy (risk mapping, developing site-specific nutritional management plans, considering off-site impacts, and using decision-making aids). Our objective in this study was to analyze the sensitivity of the FIRST Heat tool to project SOM changes in different forest types and its capacity to account for differences among ecological zones, site qualities, and forest ages.

MATERIALS AND METHODS

The calculator (FIRST Heat) is a Microsoft Excel spreadsheet in which users can select different options from drop-down menus and input parameter values specific to their communities (or select among the default values). The tool is freely available at the Community Energy Association website (http://www.communityenergy.bc.ca/resources-introduction/first-heat). FIRST Heat combines estimation of future tree growth from an ecological model, engineering calculations describing district heating systems, and economic features estimating the associated financial costs and benefits. The tool's interface and its energy, financial, and engineering aspects were described by Blanco et al. (2013), whereas the tree biomass modeling was described in depth by Kimmins et al. (1999, 2010), and soil processes modeling is briefly described below.

Study Areas

Three pilot communities were selected based on their small population size, no existing district energy systems, and location.

The latter translates into difficult access to the natural gas grid, surrounded by forests prone to wildfires, and representing different biogeoclimatic (BEC) zones (Pojar et al., 1987) to provide a diversity of forest types in the study (Fig. 1).

Burns Lake (Northern British Columbia), in the interior plateau with its surrounding forests, is mostly within the Sub-Boreal Spruce (SBS) biogeoclimatic zone. In contrast to boreal, the sub-boreal climate is slightly less continental, thus slightly warmer in January and cooler in July. Sub-boreal winters are shorter and the vegetative season slightly longer. The mean annual temperature of the SBS ranges from 1.7 to 5°C. The average temperature is below 0°C for 4 to 5 mo of the year and above 10°C for 2 to 5 mo. Mean annual precipitation data from long-term stations ranges from 440 to 900 mm, of which 25 to 50% is snow. Mean annual precipitation can range from 415 to 1650 mm (Meidinger and Pojar, 1991). Upland soils are primarily from the Luvisolic, Podzolic, and Brunisolic soil orders (Boralfs, Udalfs, Spodosols, and Inceptisols in U.S. Soil Taxonomy; Soil Classification Working Group, 1998), being the most common soils found on the abundant morainal deposits. Imperfectly to poorly drained sites in the SBS typically have Gleysols or gleyed subgroups of Luvisols, Podzols, or Brunisols. These conditions create a landscape where hybrid Engelmann–white spruce (*Picea engelmannii* × *glauca*) and subalpine fir [*Abies lasiocarpa* (Hook.) Nutt.] are the dominant trees. Extensive stands of lodgepole pine (*Pinus contorta* Douglas ex Loudon) occur in the drier portions of the zone due to numerous past fires. Wetlands are abundant, dotting the landscape in poorly drained areas. Secondarily in importance in the area is the Engelmann Spruce–Subalpine Fir (ESSF) zone. The ESSF's climate is characterized by a long cold winter and a short cool summer, and only trees that survive long periods of frozen ground occur. Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and subalpine fir dominate wetter areas, with lodgepole pine as a pioneer after disturbance and mountain hemlock [*Tsuga mertensiana* (Bong.) Carrière] in higher snowfall areas.

Sicamous (Shuswap Valley) is mainly in the Interior Cedar–Hemlock (ICH) biogeoclimatic zone. A continental climate (cold winters, warm and dry summers) defines the area. The mean annual precipitation is 500 to 1200 mm, of which 25 to 50% falls as snow and the mean annual temperature is 2 to 8.7°C (Meidinger and Pojar, 1991). Soils are typically Podzols and Brunisols with Mor humus forms (Spodosols and Inceptisols in U.S. soil taxonomy). Podzolic soil development occurs in the wetter regions, whereas Brunisolic (Inceptisol) soil development is more common in the very dry to moist regions. Due to the steep topography, extensive wetlands are uncommon. Where wetlands occur, gleysols and organic soil development can occur (Aqu-suborders in the U.S. soil taxonomy). Western hemlock [*Tsuga heterophylla* (Raf.) Sarg.] and western red-cedar (*Thuja plicata* Donn ex D. Don) are characteristic species, but spruce (Engelmann–white hybrids) and subalpine fir are not unusual. Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco] and lodgepole pine are generally found on drier sites. Secondary in importance and located in the valley bottoms is the Interior Douglas-fir (IDF) zone, with warmer and drier summers. Douglas-fir is the dominant tree species, occurring in open savannah-like stands on drier sites. Wetter sites at higher elevations support dense stands of Douglas-fir. Common pioneer tree species after disturbance are lodgepole pine at upper elevations and ponderosa pine (*Pinus ponderosa* P. Lawson & C. Lawson) at lower elevations. At higher elevations there are some stands belonging to the ESSF zone.

Invermere (Kootenay mountains) is mostly in the Montane Spruce (MS) biogeoclimatic zone. The MS is typified by short, warm summers and long cold winters. Annual precipitation is between 380 and 900 mm, and the mean annual temperature is between 0.5 and 4.7°C (Meidinger and Pojar, 1991). Soils are typically Brunisols or Luvisols (Inceptisols and Boralfs/Udalfs in the U.S. soil taxonomy). In the wetter regions, podzolic soil development may occur. Engelmann and hybrid spruce and varying amounts of subalpine fir are characteristic tree species. However, due to past wildfires, successional forests of lodgepole pine, Douglas-fir, and trembling aspen (*Populus tremuloides* Michx.) are common. In the valley bottoms, there are stands in the IDF zone, whereas in the higher areas of the surrounding mountains there is some occurrence of stands belonging to the MS and ESSF zones.

Personal interviews and a review of official documents were performed on-site in each community to gather information on local wildfire protection plans, management recommendations, ecological surveys, and other related information. For each zone, GIS maps of the surrounding area in a 25-km radius were generated, using data from British Columbia's Vegetation Resources Inventory

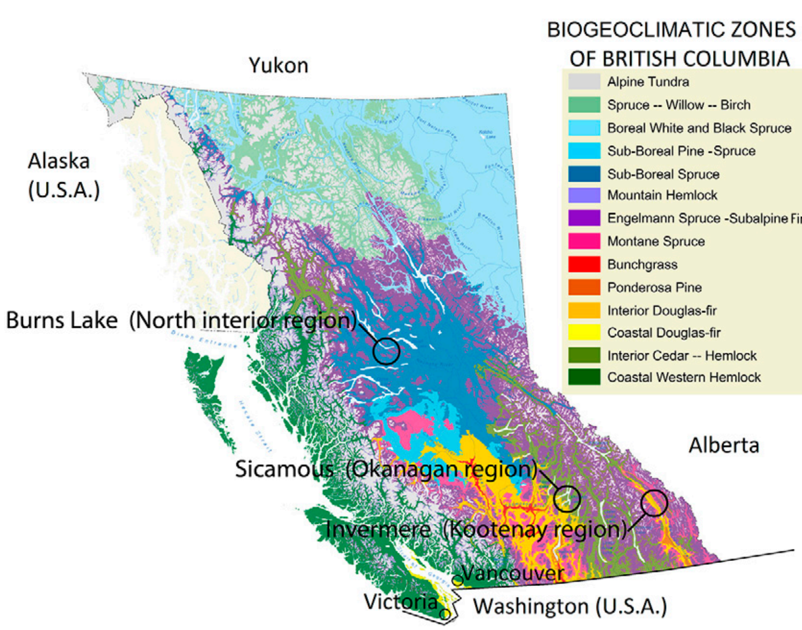

Fig. 1. British Columbia's biogeoclimatic zones (sensu Pojar et al., 1987). Also shown are the locations of the three communities used for the pilot projects: Burns Lake, Sicamous, and Invermere.

(http://www.for.gov.bc.ca/hts/vri/). Forest stands (GIS polygons) in each map were classified into different forest types depending on their biogeoclimatic zone, dominant tree species, density, and tree age (see list of stand types in Supplemental Table S1). Each forest type was individually simulated in the ecosystem model FORECAST (Kimmins et al., 1999) (see below).

Forest Biomass Modeling
Wildfire Risk Management Scenarios

The information provided by the communities, combined with FireSmart guidelines (Partners in Protection, 2003) was used to design three different management regimes. They represent the trade-offs faced by many communities between carrying out intensive but costly forestry operations to maintain wildfire risk at a minimum and the cost of such operations. In all management regimes, broadleaf trees were left on site, without thinning or harvesting, to maintain their effect as wildfire barriers (Partners in Protection, 2003).

Intense management to keep wildfire risk at minimum: post-thinning conifer target density was 121 trees ha^{-1} (6 m between crowns) in Burns Lake and Sicamous and 65 trees ha^{-1} (11 m between crowns) in Invermere, with branches pruned up to 3 m; undergrowth control operations performed every 5 yr, removing all new trees, understory, and coarse woody debris with diameter >10 cm.

Moderate management: post-thinning target conifer density of 121 trees ha^{-1} (6 m between crowns), branches pruned up to 3 m; undergrowth control operations performed every 10 yr, removing all new trees, understory, and coarse woody debris with diameter >10 cm.

Minimum management to keep wildfire risk inside safety standards: post-thinning conifer target density of 286 trees ha^{-1} (3 m between crowns), branches pruned up to 2.5 m; undergrowth control operations performed every 10 yr, removing all new trees, understory, and coarse woody debris with diameter >10 cm.

The Ecological Forest Model FORECAST

Tree and understory growth was simulated with the ecosystem-level forest model FORECAST (Kimmins et al., 1999). FORECAST is a management-oriented, deterministic, stand-level forest growth and ecosystem dynamics simulator that operates at annual time steps. A detailed description was provided by Kimmins et al. (1999, 2010), and therefore only a summary is provided here. The model uses a mass balance approach to estimate how nutrients circulate in a forest ecosystem and how their availability limits tree growth together with available light in the canopy (Fig. 2).

Model Calibration: For each forest type, calibration data are assembled that describe the accumulation of biomass (above- and belowground components) in trees and minor vegetation for three chronosequences of stands, each one developed under homogeneous conditions, representing three different nutritional qualities. Tree biomass and stand self-thinning rate data are often generated from height, diameter at breast height, and stand density output of traditional growth and yield models in conjunction with species-specific component biomass allometric equations. To calibrate the nutritional aspects of the model, data describing the concentration of nutrients in the various biomass components are required. FORECAST also requires data on the degree of shading produced by different quantities of foliage and the photosynthetic response of foliage to different light levels. A comparable but simpler set of data for minor vegetation must be provided if the user wishes to represent this ecosystem component. Lastly, data describing the rates of decomposition of various litter types and SOM are required for the model to simulate nutrient cycling. The model is data intensive, but most of the data can be found in regular forest inventories performed by companies or forest services (tree size, age, mortality), and through simple chemical analysis (nutrient contents in plants and soil). Some more specific parameters such as decomposition or photosynthetic rates may need site-specific studies unless published studies of the target species in the ecological regions of interest are available.

The FORECAST data sets used for this project were based on existing calibration data sets assembled as part of previous research (Seely et al., 2010). In all cases calibration data (biomass accumulation rates, top height, diameter at breast height, and stand density) were derived from regional growth and yield tables in combination with species-specific allometric biomass equations. Other calibration data were derived from literature sources: N concentrations in biomass components (Kimmins et al., 1979; Peterson and Peterson, 1992; Wang et al., 1996), decomposition rates (Prescott et al., 2000a, 2000b; Trofymow et al., 2002), litterfall rates (Kimmins et al., 1979; Peterson, 1988; Li et al., 2003), light transmission (Messier et al., 1998; Leifers et al., 2002; Comeau and Heineman, 2003), and light-limited growth rates (Mailly and Kimmins, 1997; Leifers et al., 2002; Claveau et al., 2002). Calibration values for the most important parameters can be found in those studies and are not repeated here.

Model Initialization: The detailed representation of many different litter types and soil organic matter conditions makes it impractical to measure initial litter and soil conditions directly in the field; consequently, the model is used to generate starting conditions, simulating the historical fire regimes until the system reaches a steady state condition (for a broader discussion on this topic see, for example, Seely et al., 2002; Welham et al., 2002). Based on data on fire regimes for each BEC, the initial conditions were created for each ecosystem type using the parameters shown in Supplemental Table S2.

Simulation of Tree Growth: Projection of stand growth and ecosystem dynamics is based on a representation of the rates of key ecological processes regulating the availability of, and competition for, light and nutrient resources. The rates of these processes are calculated from a combination of historical bioassay data (biomass accumulation in component pools, stand density, etc.) and measures of certain ecosystem variables (e.g., decomposition rates, photosynthetic saturation curves, etc.) by relating

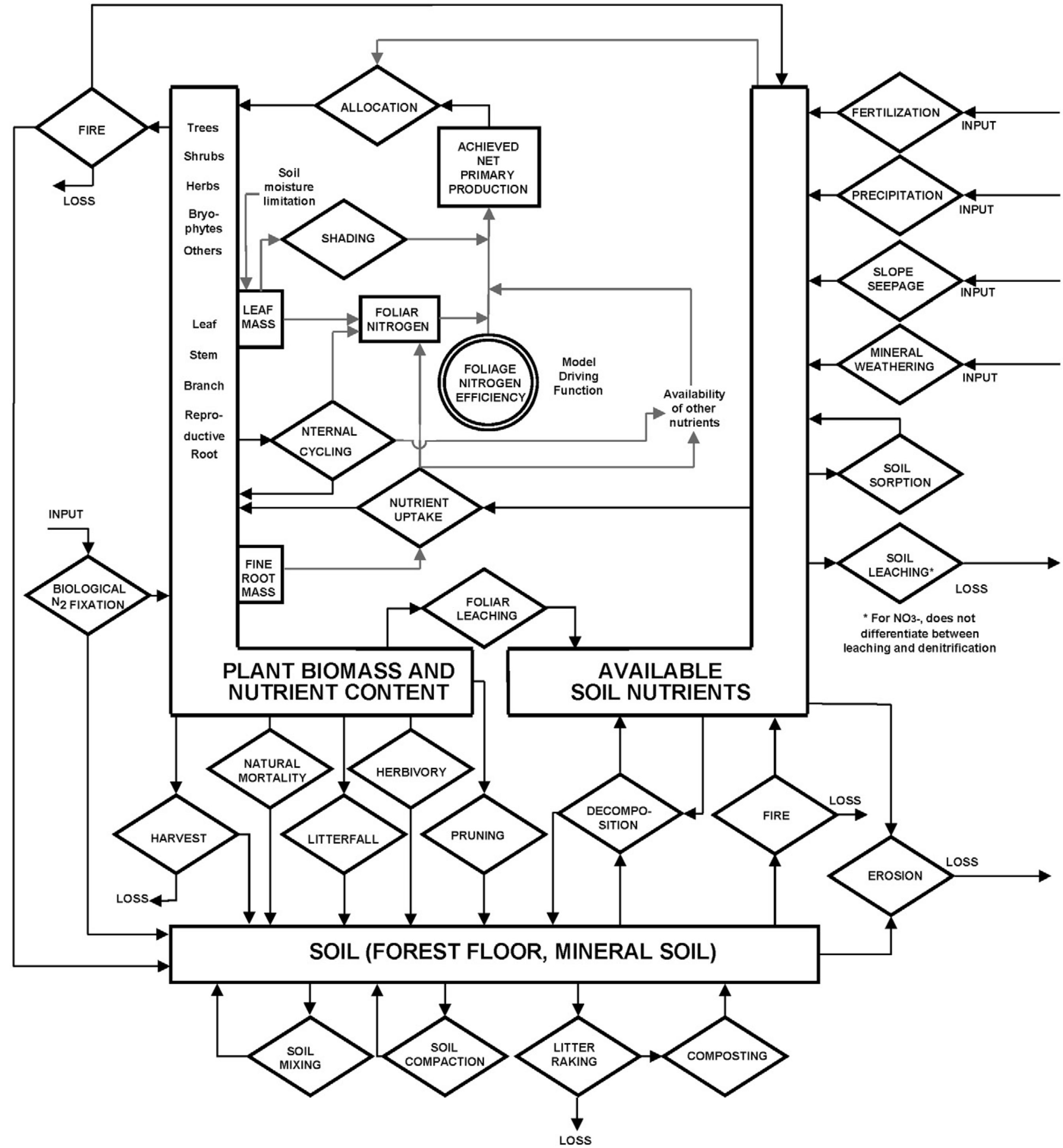

Fig. 2. FORECAST is the forest-growth engine of FIRST Heat. The model simulates tree growth based on potential biomass generated by the foliage through photosynthesis (central circle). Photosynthesis depends on the efficiency of leaf N, which is limited by the availability of light and nutrients. Nutrients in the ecosystem can be in different compartments (rectangles) and transfer between them following natural or anthropogenic pathways (diamonds) (modified from Kimmins et al., 1999).

biologically active components (foliage and small roots) with calculations of nutrient uptake, capture of light, and net primary production. With the calibration data obtained from different sources, the model calculates the annual rates of different ecological processes (tree growth, litterfall production, mortality, etc.) based on the historical data on tree growth and density provided by the user. The change in biomass in each time step is derived from a series of age–biomass curves created with empirical data. Litterfall is calculated using user-defined values based on empirical litterfall rates. Mortality is derived from a series of age–stand density curves created with empirical data (for a detailed description of mortality simulation in FORECAST, see Kimmins et al., 1999). Mortality is calibrated through two different parameters: curves of historical stand density for different ages and the proportion of mortality that is due to non-interspecific competition factors. Together, these two parameters allow simulation of the endemic, low-level mortality events caused by pests and diseases.

To estimate photosynthesis, FORECAST simulates canopy foliage biomass as a "blanket" that covers the stand and that is divided into several layers of 0.25-m height, each of them increasingly darker from the top to the bottom of the canopy. To calculate the nutritional aspects of tree and plant growth, FORECAST requires data on the nutrient concentration in each different tree organ. Nutrient dynamics in this study were restricted to N, the most limiting nutrient in temperate and boreal forests (Seely et al., 2010). The effects of soil moisture limitation were assumed to be implicitly simulated by calibrating the model for different site qualities that are limited by water and nutrient availability and by setting a maximum foliar area for each site quality (Kimmins 1993). However, this approach may be limited in the most arid sites, where an explicit simulation of water limitation may be necessary (Kimmins et al., 2010). Data describing the decomposition rates for various litter and humus types are required to simulate nutrient cycling. Decomposition rates are defined by the user (using values from empirical studies) and are affected by site quality, which in turn is defined depending on nutrient and water availability. Snags and logs are tracked by placing them into different biomass pool categories depending on their original sizes (with slower decomposition rates for snags and for stems with larger sizes).

Nitrogen cycling in FORECAST is based on a mass balance approach where N can exist in three distinct pools: (i) the plant biomass pool; (ii) the available soil nutrient pool; and (iii) the SOM–forest floor pool. Inputs and outputs of N to the ecosystem are simulated in a four-stage process for each annual time step. The available N pool is calculated by consecutively simulating the different inputs and outputs of the biogeochemical cycle: deposition, fertilization, seepage, leaching, mineralization, and immobilization (Fig. 3). A detailed description of the simulation of each of these fluxes in FORECAST can be found in Blanco et al. (2012).

Scenario Analysis: Three different forest age scenarios were simulated for each forest type: young forest (average stand age younger than 80 yr), mature forest (average stand age between 80 and 150 yr), and old growth (average stand age >150 yr). For each forest type, management operations were simulated to start at the average stand age for each age scenario and lasted for 50 yr. The resulting 50-yr trends of tree, understory, and forest soil biomasses for each simulation were linked to each polygon type in the GIS map (for a detailed description of the biomass production estimations, see Blanco et al., 2013) In addition, for each stand type, the percentage losses of total SOM after 50 yr were calculated. To simplify the output information for FIRST Heat users, an ecological sustainability index was created. This index indicates the proportion of stand types under wildfire abatement risk management that may have significant SOM losses. The index can have one of the following values: (i) Low ("green light")—most of the stand types will have no appreciable reductions in SOM after 50 yr; (ii) Moderate ("yellow light")—more than 40% of stand types will lose 10% or more of their initial total SOM after 50 yr; and (iii) Investigate ("red light")—some forest types may be heavily affected by forest management. For the last situation, >40% of stand types will lose 20% or more of the original SOM after 50 yr, and detailed stand-specific ecological assessments are recommended.

Although for many interior British Columbia communities, the mountain pine beetle epidemic has been a catalyst for considering biomass heating projects, and although infested wood can be used in these systems, the epidemic is already declining (Chen, 2014) and the extraordinary salvage logging levels will probably last for only a few more years (Burton, 2010). Because the FIRST Heat Tool is a long-term (50-yr) planning tool that extends past these short-term pulses of biomass, much shorter than the forest growth simulations with FORECAST, we did

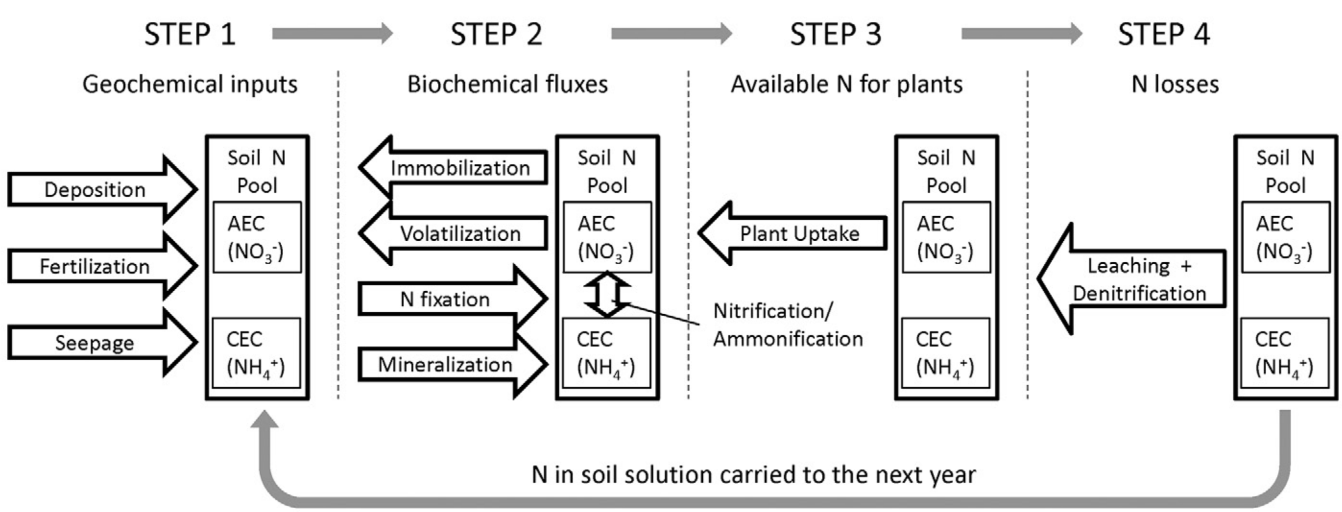

Fig. 3. Estimation of available N in FORECAST in each annual time step. Step 1: geochemical inputs are calculated, with all the forms of N lumped together. Step 2: biochemical fluxes. Step 3: Plants take up the available N. Step 4: Soil N remaining for next time step is calculated by subtracting the remaining N from the soil cation exchange capacity (CEC, for NH_4) or anion exchange capacity (AEC, for NO_3). The N excess was assumed to be lost via leaching. (For complete descriptions of the simulation of these processes, see Kimmins et al., 1999).

not account for mountain pine beetle dynamics and their influence on wood supply, although background mortality levels induced by pests or other factors were included in the simulation. Finally, because the model has been evaluated before for an examination of soil productivity in British Columbia forests (e.g., B. Seely, unpublished data, 2005; Seely et al., 2010; Blanco, 2012; among others), we did not repeat validation tests.

Statistical Analysis

An initial one-way ANOVA was used to analyze differences among management intensities (factor) in relative changes in SOM after 50 yr. Then, one-way ANOVAs (with the data split by management intensity) were used to test significant differences among the levels of each of the four factors that defined forest sites: management type, ecological (BEC) zone, site quality, and age class (each factor tested separately). Data were logit-transformed before analyses to achieve a normal distribution for percentage values. Tukey's honestly significant differences were used to detect significant differences between groups for each factor. Decision-tree analysis was used to study the capability of the FORECAST model to distinguish among different situations and to compare with the ad hoc threshold of 10% loss of original SOM. Partitioning results were cross-validated using a k-fold cross-validation procedure ($k = 10$). This method divides the original data into k subsets. In turn, each of the k sets is used to validate the model fit on the rest of the data, fitting a total of k models. The model giving the best validation statistic is chosen as the final model. All statistical analyses were performed using JMP Version 5.0.1 (SAS Institute).

RESULTS

The model was able to simulate the evolution of SOM with time in a large variety of stand types (952 in total). Managing the urban–forest interface caused significant SOM losses compared with the no-management situation in all kinds of forest stands (Table 1). Intense management caused significantly higher SOM losses than moderate and minimum management in all stand types except in medium and rich sites, where there was no differences among management intensities (Table 1). For a given ecological zone, stand productivity (determined by site quality and forest age) had a clear influence on the resilience of the ecosystem to changes in SOM (Fig. 4).

In the more productive stands (medium site quality and young stand age), the influence of any type of management was small and the temporal pattern of SOM was very similar to the unmanaged stands: an initial slight increase as the young stand accumulated litter, followed by a slight drop as the forest soil stabilized. By the end of the 50 yr, every management type caused no more than an additional 3% SOM loss compared with the no-management scenario and an average of ∼8% loss of the initial SOM. In less productive stands (poor site quality, mature stands), however, both no-management and minimum management allowed a small buildup of SOM of 6.2%and 2.7%, respectively, whereas both moderate and intense management, which removed twice as many trees from the stands as the minimum management, produced a drop in SOM since the beginning of the simulation, amounting to 10.7 and 11.3% losses under moderate and intense management, respectively.

When SOM changes were summarized for all the ecological zones, it could be seen how the SBS zone was more sensitive to SOM losses in all type of scenarios, with SOM losses that could reach 23.1% in 50 yr under intense management (Fig. 5). The IDF zone was the second most sensitive ecological zone to SOM losses. It is also noticeable how stands in the two most extreme zones (SBS and IDF, boreal and semiarid, respectively) were prone to accumulate SOM if no wildfire risk prevention mea-

Table 1. Accumulated change in SOM after 50 yr for different management regimes in different stand types. Negative values indicate net increases in SOM. For all the rows, the treatment effect was significant at $p < 0.0001$.

Stand type	F	n	Management regime: Intense	Moderate	Minimum	No management
			% of initial value			
BEC zone†						
SBS	461.490	196	−25.03 ± 0.62 a‡	−18.53 ± 0.49 b	−14.31 ± 0.54 c	4.95 ± 0.73 d
ESSF	89.357	253	−4.66 ± 0.58 a	−4.33 ± 0.66 b	−4.66 ± 0.58 ab	7.33 ± 0.62 c
MS	47.740	84	−5.04 ± 0.59 ab	−4.39 ± 0.55 b	−6.65 ± 0.35 a	10.70 ± 0.44 c
ICH	43.492	307	−7.32 ± 0.64 a	−8.30 ± 0.51 a	−3.39 ± 0.73 b	0.60 ± 0.55 c
IDF	50.852	112	−11.60 ± 0.89 a	−7.53 ± 1.14 b	−10.75 ± 0.50 ab	2.16 ± 0.89 c
Site quality						
Poor	175.845	673	−12.89 ± 0.74 a	−9.80 ± 0.57 b	−7.51 ± 0.54 c	4.68 ± 0.43 d
Medium	36.003	195	−7.79 ± 0.72 a	−6.70 ± 0.77 a	−6.96 ± 0.67 a	1.05 ± 0.56 b
Rich	84.384	84	−6.14 ± 1.43 a	−7.07 ± 0.96 a	−4.47 ± 0.96 a	−0.04 ± 0.85 b
Stand age						
Young	62.575	285	−9.87 ± 0.98 a	−7.35 ± 0.74 ab	−6.24 ± 0.70 b	4.75 ± 0.83 c
Mature	81.984	384	−11.17 ± 0.98 a	−8.63 ± 0.76 ab	−7.07 ± 0.67 b	3.87 ± 0.40 c
Old growth	62.970	283	−12.71 ± 1.04 a	−10.88 ± 0.75 ab	−8.10 ± 0.78 b	1.81 ± 0.60 c

† BEC, biogeoclimatic zone site classification for British Columbia (Pojar et al., 1987); SBS, Sub-Boreal Spruce; ESSF, Engelmann Spruce–Subalpine Fir; MS, Montane Spruce; ICH, Interior Cedar–Hemlock; IDF, Interior Douglas-fir.

‡ Means (± standard errors) followed by different letters in the same row are significantly different among management types with Tukey's honestly significant difference.

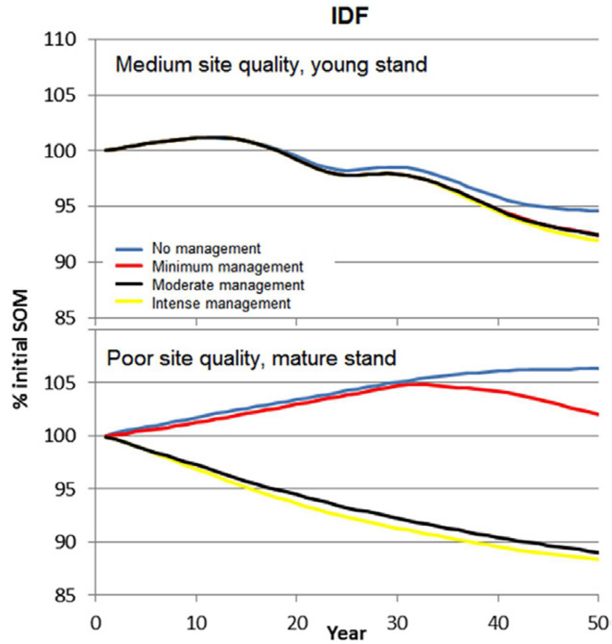

Fig. 4. Relative change in soil organic matter content in two mixed conifer forests in the Interior Douglas-fir (IDF) ecological zone consisting of Douglas-fir, western red-cedar, and western hemlock) at Sicamous—a young forest in a medium-quality site (top) and a mature forest in a poor-quality site (bottom)—under different management control plans for three different levels of wildfire risk corresponding to maximum, intermediate, and minimum interventions as recommended by FireSmart. Stand types correspond to Fd Cw Hw medium and Hw Cw Fd poor in Supplemental Table S1.

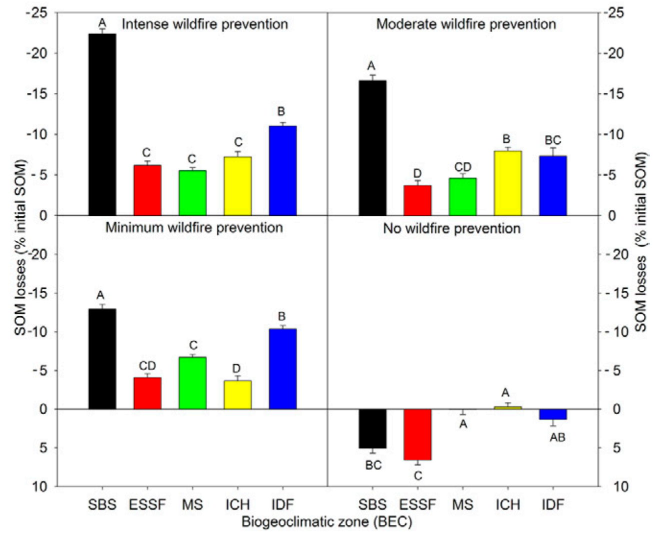

Fig. 5. Percentage of initial soil organic matter (SOM) lost after 50 yr in forests of the Sub-Boreal Spruce (SBS), Engelmann Spruce–Subalpine Fir (ESSF), Montane Spruce (MS), Interior Cedar–Hemlock (ICH), and Interior Douglas-fir (IDF) biogeoclimatic zones (Pojar et al., 1987) under four different types of management for wildfire risk control.

sures were taken. When estimates of SOM losses were grouped by stand age, there was a consistent trend to increased losses with increasing stand age, but the differences among age classes only became significant between young and old-growth stands when moderate wildfire prevention was applied (Fig. 6). On average, all age classes also showed a tendency for small buildups of SOM after 50 yr when no management was used, but this trend was significantly weaker in the old-growth forests than the other two age classes (Fig. 6). Poor sites were more sensitive than medium and rich sites, although these differences became significant only under moderate and intensive wildfire risk management (Fig. 7). On average, poor sites lost 11.8% of their initial SOM levels after 50 yr if intensive management was applied. On the other hand, poor sites were also significantly more prone to accumulate SOM under no-management conditions, whereas the medium and rich sites kept virtually stable amounts of SOM.

The significant differences among groups for the four stand-defining variables allowed a partitioning study in which a decision tree was used to separate situations with significant differences in SOM losses in a hierarchical way. Obviously, the first partition separated unmanaged stands from those under management (Fig. 8). After management, the second most important variable was the ecological zone, first separating the SBS zone from the rest. Under all scenarios, SOM losses in SBS stands were above the 10% threshold. On the other hand, no scenarios in the ESSF, MS, and ICH ecological zones produced SOM losses above 10% of the initial levels after 50 yr, but in the IDF zone the losses surpassed the 10% threshold depending first on the type of management and second on the stand age. Moderate management scenarios did not cause losses above 10% of the initial SOM, but for the other management types, all stands with age >80 yr lost >10% of the initial SOM. Splitting the young stands further, minimum management did not cause losses above the threshold, but with intense management there were average losses of 12.03% in medium sites (Fig. 8). The decision tree had $r^2 = 0.633$, and it showed a partition with values for cross-validation (k-fold = 10) of crossvalSSE = 28978.89, which was the average of error the 10 folded cross-validations. This error was very similar to the error of the full sample SSE of 28547.97, indicating the robustness and acceptable capacity of the model to predict and correctly split future samples.

DISCUSSION

Although there is wide consensus among the scientific community on the utility of ecological models (Messier et al., 2003), it has been more difficult to generate tools that are scientifically sound but practical for managers (although some examples already exist, see Kimmins et al., 2010). In this work we present a novel approach: linking an ecological model, a GIS database, and a user-friendly spreadsheet to create a communication tool easily used by managers. FIRST Heat represents a proof of concept, and we do not pretend our simulations to be an accurate representation of specific stands or forests. Rather, we intended to test the capability of the ecological engine of the tool (the FORECAST model) to differentiate among different stand types, environmental conditions, and management regimes.

The guidelines of Partners in Protection (2003) for wildfire safety in the urban–forest interface define an initial intense operation to reduce conifer stand density, followed by a continuous maintenance management. To avoid problems of excessive windthrow losses among the remaining trees following a sudden reduction in stand density, thinning operations were designed in two steps, where 50% of the harvestable trees were removed per step and each step was separated by 10 yr. As a consequence, during the first 10 yr of management, a large amount of biomass was generated (Blanco et al., 2013). Although our previous research has shown that implementing different levels of stand density control may not produce large differences in biomass production during the regrowth period (Blanco et al., 2013), this biomass production could have important ecological consequences.

Capability of FIRST Heat to Discriminate among Factors Influencing Soil Organic Matter Losses

The moderately high r^2 of the regression tree indicated an acceptable result when splitting the different samples into groups. The cross-validation procedure also indicated the capability of the selected tree to acceptably place future samples into the correct categories, with an estimated error similar to that in the set of simulations. In the intermediate and intense management scenarios, long-term biomass production was predicted to decline because biomass removal depleted nutrients from the sites, thereby reducing site quality and tree productivity. This phenomenon was more pronounced in areas with slower growth rates: the sub-boreal stands in the SBS zone and the semiarid stands in the IDF zone. However, even the most productive sites such as those in Sicamous could be affected. Interestingly, our projections showed that in the absence of management, both areas would tend to accumulate SOM.

The reason for this greater sensitivity to management is probably the relatively lower SOM in IDF and SBS zones than in the other three ecological zones (ESSF, MS, and ICH). These relatively low levels of SOM are probably related to frequent and intense stand-replacing fires during the past several hundred years in pine forests (Wong et al., 2004). Natural disturbances and stand history have a significant and persistent effect on SOM (Robertson et al., 1997). Such fires commonly result in the consumption of significant quantities of biomass and litter (thus reducing future sources of SOM) and an associated loss of soil nutrients via volatilization and leaching (Boerner, 1982; González-Pérez et al., 2004; Johnson et al., 2007). The model seems to adequately capture the importance of the ecological zone (and therefore stand composition) as the main factor causing differences among sites in SOM losses under management for wildfire control.

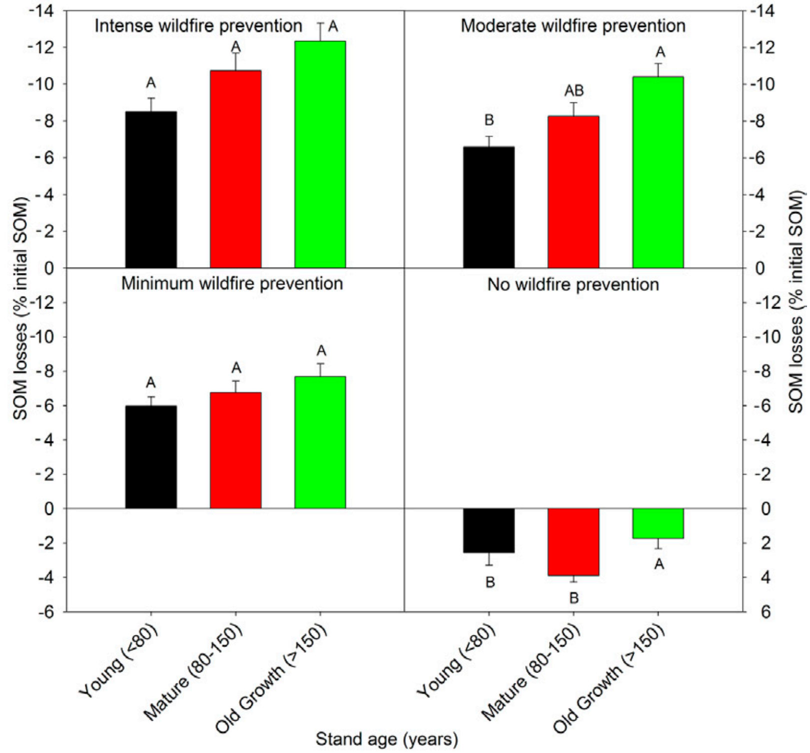

Fig. 6. Percentage of initial soil organic matter (SOM) lost after 50 yr in stands of three age classes under four different types of management for wildfire risk control.

Moreover, the generally low levels of soil productivity in the two ecological zones most sensitive to SOM loses (the sub-boreal SBS and the arid IDF, with mean site index values <16) is

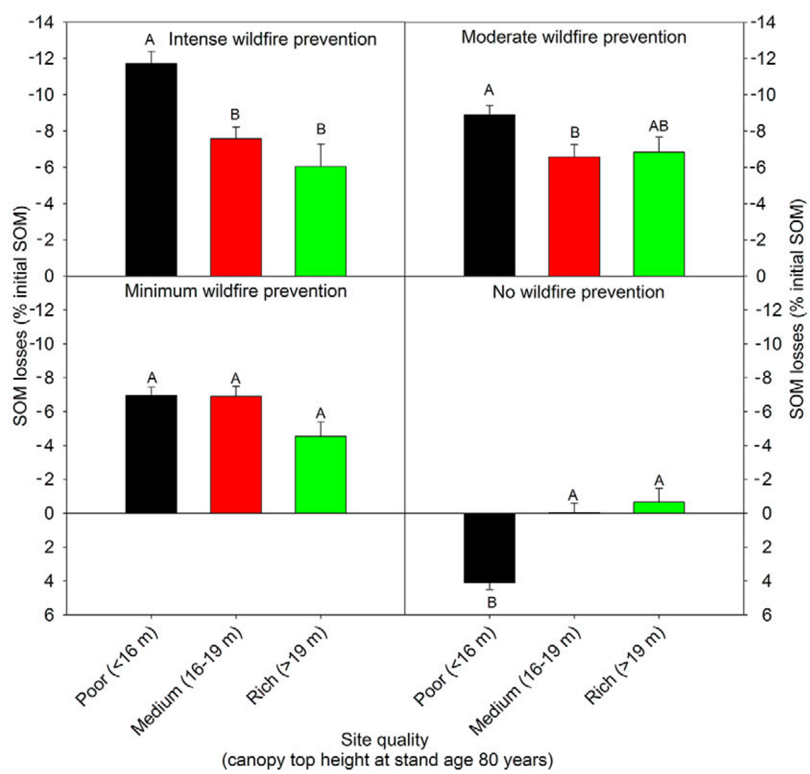

Fig. 7. Percentage of initial soil organic matter (SOM) lost after 50 yr in stands of three site qualities (measured as site index) under four different types of management for wildfire risk control.

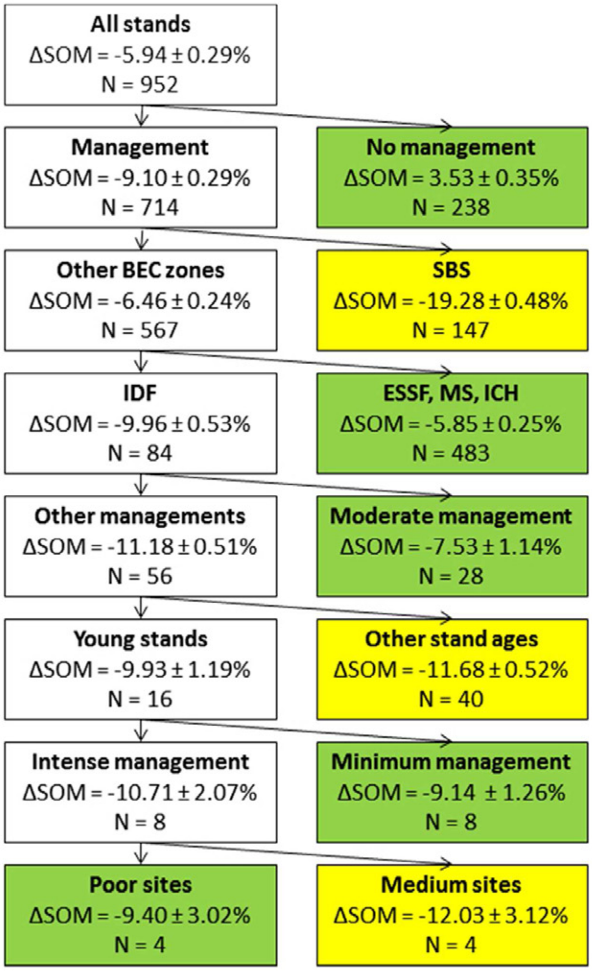

Fig. 8. Decision tree defining the hierarchy and grouping of the variables defining soil organic matter (SOM) losses in the Sub-Boreal Spruce (SBS), Engelmann Spruce–Subalpine Fir (ESSF), Montane Spruce (MS), Interior Cedar–Hemlock (ICH), and Interior Douglas-fir (IDF) biogeoclimatic (BEC) zones. White boxes indicate groups that contain stand types with losses above and below the 10% threshold and that can be further split. Green boxes are terminal nodes with all the stand types below the 10% threshold, and yellow boxes are terminal nodes with all the stand types above the 10% threshold (negative values denote SOM losses).

partly a reflection of low soil N levels. The relationship between forest productivity and N availability has been described several times in forest soils (Ågren, 1985; Reich et al., 1997; Perry, 1998; among others). The relationship is based on the fact that SOM is an important source of nutrients that can be released at relatively constant levels for long periods of time. The SOM also improves the soil structure and water holding capacity. Others have also found SOM to be a good indicator of site productivity for similar reasons (Doran and Parkin, 1994; Prescott et al., 2000a, 2000b; Seely et al., 2010).

Secondary to the ecological zone, stand age also showed up as a significant factor determining the effects of forest management on SOM. The simulations for many stands showed SOM following a pattern previously described for forest soils: first a small peak after harvesting, then a decrease for about 15 yr, and then a slow increase (Martin et al., 2001) (Fig. 4). In productive stands, the small peak about 15 to 20 yr after stand initiation was a consequence of the humification of the massive input of woody debris left on site after the disturbance caused by management. Sharp declines in SOM after harvesting followed by recovery have been described in chronosequence studies (Federer, 1984; Martin et al., 2001). However, although the model showed considerable fluctuation of litter mass pools following the initial thinning in most stands (data not shown), in the short term, soil C was largely unaffected by management activities in most of the simulated scenarios, except the most sensitive ones. Hence the decision tree showed how, with a threshold of 10% reduction after 50 yr, the FIRST Heat tool was able to discriminate 191 out of 952 scenarios (20.1%) as potentially problematic from an ecological sustainability point of view. These results indicate the capability of the decision-support tool to match the observed scarce effect of forest management on total SOM at half-century time scales (Johnson and Curtis, 2001). However, the tool was sensitive enough to simulate how the long-term effect of management activities on soil C is dependent, at least in part, on the time the forest has had to accumulate SOM.

For example, after stand-replacing fires, forest stands usually have relatively low amounts of SOM and often show a gradual increase in SOM as the forests develop (Johnson, 1992; Huntington, 1995; Blanco, 2012). Old-growth forests, in contrast, typically contain significant quantities of SOM. These forests are more susceptible to losses in organic C following harvest and conversion to managed stands (Harmon et al., 1990; Johnson, 1992; Schulze et al., 2000). In addition, old-growth forests usually had lower productivities than young or mature stands, which translates into lower litter production rates. Litter pools represent a small fraction of the total ecosystem C, but rates of litter production can have a significant impact on C stored in the SOM because litter is the main SOM input in forest soils (Alban, 1982: Seely et al., 2002) and the main source of mineralized N (Blanco et al., 2011). If there is less litter entering the forest soil, and also in more open stands where litter decomposition can be reduced compared with closed-canopy stands, nutrients could become increasingly limiting (Blanco et al., 2005). Limitation in nutrient availability produces lower tree growth and therefore keeps the canopy more open, with levels of available light for the understory higher than in the unmanaged forests. As a consequence, the understory would grow more, competing more intensively with trees and therefore reinforcing nutrient limitations (Bi et al., 2007; Kimmins et al., 2008; Blanco, 2012).

The last factor to be discriminated by the decision tree was site quality (measured as site index or dominant tree height at a given age). However, most of the influence on resource availability is already accounted for in the previous two factors in the decision tree (ecological zone and forest age). The contribution of the soil to the site index is also the result of interactions with other site factors, tree breeding, and silvicultural practices that manipulate soil functions (Schoenholtz et al., 2000). However, from our results it seems clear that poor sites should be more carefully managed to avoid overexploiting their productive capacity because they will need longer times to recover from human activities (Blanco, 2012). Site quality, however, is not a fixed stand attribute, and forest management can improve or reduce it (Kimmins et al., 2008). Modeling combined with risk mapping could be an important tool to design site-specific management plans for biomass production that could avoid nutrient losses from the stands in the long term (Mead and Smith, 2012). It is becoming clear that modeling forest ecosystems under changing conditions (e.g., new management practices, forest decline, climate change, invasive species, etc.) has to be done with models able to simulate changes in site quality, and traditional statistical growth and yield models are unlikely of fulfill these requirements (DeAngelis and Mooij, 2003; Bi et al., 2007: Seely et al., 2010; Kimmins et al., 2010).

The Use of Models to Estimate Potential Changes in Soil Organic Matter

Detecting changes in soil C brought about by changes in land management requires precise field measurements, but a number of soil characteristics make this challenging (Conant et al., 2003). Having reliable monitoring techniques and, more importantly, tracking the consequences of soil disturbance for forest growth and hydrology are paramount to improving understanding and prediction of the practical consequences of forest practices (Curran et al., 2005).The modeling results presented here are similar to the results from previous studies in British Columbia (Seely et al., 2002, 2010; Blanco, 2012), providing support for the use of total site SOM (in particular the relative loss of SOM on a given site) as an effective measure of the maintenance of long-term ecosystem productivity. Seely et al. (2010) indicated that monitoring plans for forest management sustainability should at least be able to detect losses of 20% of SOM at monitoring intervals of 10 yr. Also, 20% has been determined as a key threshold for loss of ecosystem productivity (DiStefano, 2001; Yanai et al., 2003).

Given the inherent uncertainties associated with a modeling exercise of the characteristics presented here, we think that using a threshold of 10% SOM losses in 50 yr is an adequate level to "turn on the yellow light" in our decision-support tool, advising for more accurate studies and still being on the safe side. In addition, in the most severe cases of potential SOM losses (most of them in the sub-boreal SBS zone), the model predicted an average of 23% loss, in the "red light" zone that indicates additional ecological assessments are needed before implementing any forest management for biomass. Therefore, we consider that given the capability of the tool to segregate between different site conditions, risks for nutrient losses, and added off-site costs, FIRST Heat would be a valuable tool to follow the 10 principles of nutrient management in forest biomass production regimes (Mead and Smith, 2012).

CONCLUSIONS

Our analysis has shown the feasibility of linking ecological science with financial, social, and energy models to ensure that ecological sustainability is incorporated into the decision process for local forest management plans in rural communities. It is possible to satisfy multiobjective management goals by linking a reduction in wildfire risk and energy production in an ecologically sustainable way, provided the ecological conditions underlying forest productivity and health are understood. Therefore, the use of the "traffic light" approach provides a good example of the practical use of these thresholds of SOM change to define sustainable forest management plans that are easy to communicate to stakeholders and local decision makers. Although the primary variable for making a decision would be wildfire safety for the communities, especially when managing the stands right beside human populations and infrastructures, management practices must be carefully designed so as not to remove more organic matter (e.g., slash, forest floor material, coarse woody debris) from the site than is necessary (Ballard, 2000; Prescott et al., 2000a). Having a preliminary assessment of the stand types more at risk of losing SOM under different management plans is indeed useful information on which to base more informed decisions.

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