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**Forests may need centuries to recover their original productivity after continuous intensive management: an example from Douglas-fir stands**

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**Abstract**

How long would it take for forests to recover their original productivity following continuous intensive management if they are left untouched? This issue was explored using the model FORECAST, calibrated and validated for coastal Douglas-fir stands on Vancouver Island (western Canada). Three types of forest management (production of timber, pulp, and biomass) were simulated, being different in utilization level and rotation length (stem-only and 75-year rotation for timber production, whole-tree and 30-year rotation for pulp/fibre, and whole-tree and 15-year rotations for biomass production). Management was simulated for 150 years, followed by several cycles of natural growth without management ending with a stand-replacing windstorm with a return time of 200 years. Productivity-related ecological variables in previously managed stands were compared to natural forests. Stands developed after management for timber would quickly reach values similar to non-managed forests for tree and understory total biomass, stored carbon, available nitrogen and soil organic matter (SOM). However, intensive management regimes designed for fibre and biomass production would cause a decrease in SOM and nutrient availability, increasing understory biomass. As a consequence, stands recovering from intensive management would need at least two stand-replacing events (400 years) to reach a productivity status similar to non-managed stands. Stands developed after management for biomass would take much longer, up to 600 or 800 years to recover similar values of SOM and understory biomass, respectively. Current fertilization prescriptions will likely be not enough to stop a quick drop in forest productivity associated with intensive management. Intensifying forests management to achieve short-term objectives could produce a reduction of stand productivity that would influence tree growth for very long time (up to several centuries), if such management is continuously implemented at the same stand. Some of these effects could be reduced if one rotation of intensive management (for pulp or bioenergy) is followed by a rotation of management for timber, or leaving the forest without management for an equivalent time.

**Keywords:** Ecosystem recovery; ecological model; FORECAST; long-term sustainability; site productivity; tree-understory interactions;



## 1. Introduction

Management effects on forest ecosystems are manifold and they can include changes in species composition, genetic diversity, regeneration patterns, stand structure, aboveground biomass pools, nutrient cycling, nutrient use efficiency, microclimate, forest fragmentation, gap formation, hydrology and more (Blanco et al., 2005; Chazdon, 2003; Foster et al., 2003; Franklin et al., 2002). Among the most important factors that can influence future forest growth are: nutrient removal, soil water retention, soil physical properties, and soil erosion (Ballard, 2000; Burger, 2009; Eisenbies, 2005; Powers, 2005; Smith, 1995). Anthropogenic changes in any of these factors are usually translated into long-term reductions of soil fertility, affecting forest productivity (Bi et al., 2007; Morris et al., 1997). As a consequence, forest management can be considered as an important disturbance that can greatly influence ecosystem processes. This influence can sometimes be seen decades or even centuries after human management (Foster, 2000; Foster et al., 1998).

The long-term effects of intensive management, however, are still cited as collections of separated observations rather than as a unified ecological theory (e.g. Burger, 2009; Eisenbies et al., 2009). One reason for this situation is that the effects of repeated rotations using whole-tree harvesting or other intensive management practices over long periods of time remain poorly documented. This issue, added to the long time needed to see the ecological effects of several consecutive rotations of intensive management, makes the assessment of long-term ecological sustainability of forest management especially difficult (Kimmins et al., 2010). This situation becomes more critical with the increasing interest world-wide in using short-rotation forestry to mitigate climate change, and to restore altered forest landscapes to more natural states (Lo et al., 2012). Timber or fibre products are seen as an option for sequestering atmospheric carbon, whereas energy from biomass could help to reduce the use of fossil fuels (Nichols et al., 2009; Stupak et al., 2007). In Canada, some policymakers have suggested the possibility of using not only traditional fast-growing species such as willow or poplar to produce bioenergy, but also other local species traditionally used for timber production such as Douglas-fir



(Hamilton, 2008; SFM, 2008). As a consequence, the potential impact of these new forms of management needs to be examined in the Pacific Northwest and elsewhere.

The common belief among forest managers is that fertilization or ash deposition can maintain forest productivity under intensive management (Burger, 2009; Lattimore et al., 2009; Stupak et al., 2007), and that forest stands can recover quickly after management if left alone. However, there are few long-term field studies that actually explore this possibility. Recent reviews have pointed out ecosystem forest models as a good alternative to costly and time-consuming field trials, because they can organize complex information and data at various scales, becoming decision-support tools helpful to set targets and guide the use of criteria and indicators of sustainable forest management (Eisenbies et al., 2009; Lattimore et al., 2009; Seely et al., 2010). In recent years several models have been used to study the long-term effects of intensive forest management, especially on tree biomass and soil organic matter (e.g. Corbeels et al., 2005; Palouse et al., 2008; Peng et al., 2002; Zhang et al., 2006). Among them, the FORECAST model stands out due to its hybrid approach (Kimmins et al., 2010).

The FORECAST model is an extension of the FORCYTE model (Kimmins and Scoullar, 1979), originally developed under commission from the Canadian Forest Service to estimate the potential of bioenergy from the forest. FORCYTE was one of the earliest “hybrid models”, developed to combine the reliability of empirical models with the flexibility of models simulating ecological processes (Landsberg, 2003; Kimmins et al., 2010). The original Chapman-Richards equation in FORCYTE used to simulate tree growth was later substituted by an algorithm that modifies the historical growth rates depending of resource availability. These and other improvements in the early model, together with the addition of multi-value capability, led to the development of the FORECAST model (Kimmins et al., 1999). Since then, FORECAST has been used as an evaluation tool of different natural and human-made disturbances (forest management, atmospheric pollution, agroforestry, etc.) in many types of world forest ecosystems. These include tropical (Bi et al., 2007; Blanco and González, 2010; Wei et al., 2012, Blanco et al., 2012), temperate (Wei et al., 2003, Welham et al., 2002, Seely et al., 2002, Blanco et al., 2007), and boreal forests (Seely et al., 2002, 2008, 2010; Welham et al., 2007). For each forest type, detailed evaluation



studies have been carried out that have shown the reliability of FORECAST when simulating long-term trends in forest productivity (Blanco et al., 2007; Wei et al., 2012; Seely et al., 2008). The model is specially designed to examine the impacts of different management strategies or natural disturbance regimes on long-term site productivity, N dynamics and C sequestration.

All these analyses mentioned above have studied only the period of active management and therefore they do not address two important questions: 1) what will happen to the ecosystem productivity, fertility, and biomass / carbon pools after intensive management has ended?; and 2) to which extent and how quickly will the forest recover their original productivity from human intervention if left alone? These questions are not trivial, especially in the current context of intensifying forest management in plantations and also managed forests. To fill this knowledge gap, in this paper I have used field data and the model FORECAST to model the growth of a Douglas-fir plantation in southern British Columbia (Canada) under different management regimes to explore the following hypothesis: 1) if common fertilization regimes can mitigate the long-term negative impact of intensive forestry on site productivity; and 2) if forests can recover their fertility and productivity pre-management conditions in a time scale similar to the time that they were under active management.

## 2. Material and methods

### 2.1. Study area

The study area is a Douglas-fir plantation located at Shawnigan Lake, on southern Vancouver Island, British Columbia, Canada, (48.38°N, 123.43°W). It is described in detail in the framework of a long-term study on thinning and fertilization established on 43 plots (0.08 ha in size each) by the Canadian Forest Service in 1975 (Crown and Brett, 1975), and therefore only a brief description is provided here. Mean annual precipitation is 1215 mm and mean annual temperature is 9.3 °C. Soils are coarse loamy orthic dystric brunisols, and the forest floor depth to the mineral soil averages 1.5 cm. At this site approximately 5000 stems ha<sup>-1</sup> were established through planting and natural regeneration in 1946 following a stand-replacing fire in an existing Douglas-fir stand. Salal (*Gaultheria shallon* Pursh), a



shrub that may reach a maximum height of 3 m, is the dominant understory species. Red alder (*Alnus rubra* Bong.), a common species in Douglas-fir forests, has not a significant presence in these stands. Site index (top height at stand age 80 years) for Douglas-fir at this site is estimated to be 25 m, although stands in the surrounding area have site indexes ranging from 17 to 26 m.

## 2.2. The FORECAST model

FORECAST is a management-oriented, deterministic, stand-level forest growth and ecosystem dynamics simulator that operates at annual time steps. The model has been described in detail before (Kimmins et al., 1999, 2010) and therefore only a summary of the main driving function to calculate tree growth is provided here. The model uses a mass balance approach to estimate how nutrients circulate in the ecosystem, and how their availability limits tree growth together with available light in the canopy (see Supplementary Material Figure S.1 for a detailed diagram of nutrient fluxes simulated in the model). Detailed descriptions of decomposition, tree uptake and biogeochemical cycles can be found in Kimmins et al. (1999). FORECAST has three application stages: 1) assembling calibration data and generating historical rates of key ecosystem processes; 2) model initialization by establishing the ecosystem condition for the beginning of a simulation run; and 3) simulation of tree and plant growth.

### 2.2.1. Model calibration

Calibration data were assembled that describe the accumulation of biomass (above and below-ground components) in trees and understory vegetation for three chronosequences of stands, each one developed on homogeneous conditions, representing three different nutritional qualities (poor, medium and rich sites). Field data on individual tree height and diameter from control plots were used in combination with a series of published allometric equations developed for the region (Standish et al., 1985) to derive estimates of various biomass components. Data describing light and N requirements were derived from literature (Kimmins et al., 1999; Mitchell et al., 1996; Seely et al., 2002). Calibration values for other model parameters and their sources are described in the Supplementary Material (Tables S.1 and



S.2). An exhaustive description of the input data requirements can be found in Kimmins et al. (1999).

Projection of stand growth and ecosystem dynamics is based upon 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 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 (see Supplementary Material), 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. Therefore, for each plant species for which historical data are provided, the total net primary production (TNPP) that occurred for each annual time step ( $t$ ) is calculated with Equation (1).

$$\text{TNPP}_t = \Delta\text{biomass}_t + \text{litterfall}_t + \text{mortality}_t \quad (1)$$

where  $\Delta\text{biomass}_t$  = the sum of the change in mass of all the biomass components of the particular species in time step  $t$ ;  $\text{litterfall}_t$  = the sum of the mass of all ephemeral tissues that are lost in time step  $t$  (e.g., leaf, branch, bark and reproductive litterfall, and root death); and  $\text{mortality}_t$  = the mass of individual plants that die in time step  $t$ . Change in biomass ( $\Delta\text{biomass}_t$ ) 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 on 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, both parameters allow simulating the endemic, low level mortality events caused by pests and diseases typical of Douglas–fir forests (Cruickshank et al., 2009, 2011; Negrón et al., 2001). Blanco et al. (2007), in a detailed evaluation of the model performance for the Shawnigan Lake experimental forests



under different fertilization and thinning regimes, estimated that values of litterfall and mortality were inside the range of empirical values.

The model also estimates the shade-corrected foliage N content (SCFN), which represents the amount of fully illuminated foliage N that was required to produce the calculated historical TNPP. To estimate foliage shading, FORECAST simulates canopy foliage biomass as a “blanket” that covers the stand and that is divided in several layers of 0.25 m height, each of them increasingly darker from the top to the bottom of the canopy. The light absorbed by each layer is calculated based on the foliage biomass present in each time step and a user-defined empirical curve of foliage mass-proportion of full light. Once an estimation of self-shading has been completed for a particular time step using the method described above, FORECAST calculates a foliar N content adjusted for the effects of self-shading (Equations 2 and 3).

$$SCFN_t = \sum_{i=1}^n (FN_{t,i} \times PLSC_i) \quad (2)$$

$$FN_{t,i} = \text{foliage biomass}_{t,i} \times \text{foliar N concentration} \quad (3)$$

where  $FN_{t,i}$  = mass of foliage nitrogen in the  $i$ th quarter-meter height increment in the live canopy at time  $t$ ,  $PLSC_i$  = photosynthetic light saturation curve value for the associated light level in the  $i$ th quarter-meter height increment in the live canopy,  $n$  = number of quarter-meter height increments in the live canopy at time  $t$ . The mean photosynthetic rate of the foliage in canopy level  $i$  is calculated by combining simulated light intensities in canopy level  $i$  with input data that define photosynthetic light saturation curves for the foliage type in question. Finally, the driving function curve for potential growth of a given species in FORECAST is the shade-corrected foliar nitrogen efficiency (SCFNE) calculated for each annual time step ( $t$ ) with Equation 4:

$$SCFNE_t = TNPP_t / SCFN_t \quad (4)$$

When data describing the growth of a species on more than one site quality (i.e. nutrient availability) are provided, SCFNE function curves will be generated during the calibration stage for each site quality. To calculate the nutritional aspects of tree and plant growth, FORECAST requires data on





nutrient concentration in each different tree organ. Nutrient dynamics in this study were restricted to nitrogen (the most limiting nutrient at this site, McWilliams and Thérien, 1997), as only limitations for nitrogen and no other nutrients have been clearly and consistently demonstrated for Douglas-fir in southern Vancouver Island (Weetman et al. 1993, 1997; Jassal et al. 2010).

Kimmins et al. (2008) have shown how the combination of light and nutrient limitation is not enough to explain complex ecological patterns in models through models, and they recommended including understory vegetation also in the simulations. Therefore, a comparable but simpler (e.g. no data on bark, wood, mortality, etc.) set of data for understory vegetation must be provided to represent this ecosystem component. Lastly, data describing 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 (Kimmins et al., 1999). Snags and logs are tracked by placing them into different categories depending on their original sizes (with slower decomposition rates for snags and for stems with larger sizes).

### 2.2.2. *Model initialization*

To establish initial site conditions the model is run in set-up mode, forcing the model to match the observed site conditions (Blanco et al., 2007; Seely et al., 2002). Initial conditions were created by running the model for eight 200-year cycles ending with a stand-replacing windthrow. These runs simulated the pre-European settlement conditions in the forests. These runs allowed the model to accumulate soil organic matter until reaching a stable value, which was used as the starting conditions for the natural baseline (see section 2.3). For the management runs and additional run of 23 years (starting with a clear-cut and ending with a fire) simulated the first exploitation of the original forest. At the end of the set-up runs the model had accumulated an amount of soil organic matter (SOM) equivalent to the field measurements of humus mass in the soil profile (without including litter) at the time of stand establishment (Crown and Brett, 1975). Two different initial conditions were created, one for each



simulated site quality: poor and rich (site indexes 16 and 26 m), which cover the range of observed site qualities in the Shawnigan Lake area.

### 2.2.3. Simulation of tree and plant growth

During the simulation stage, for each annual time step, the annual potential growth (APG) of vegetation is driven by the photosynthetic production of the foliage biomass (Equation 5). The productive capacity of a given quantity of foliage biomass (photosynthetic rate) is assumed to be dependent on foliage nitrogen content corrected for shading created by the canopy of the simulated site ( $SCFN_t^*$ ).  $SCFN_t^*$  is different from the  $SCFN_t$  that was previously calculated during the internal calibration stage (section 2.2.1). During the simulation stage the canopy simulated corresponds to the site defined by the user for that particular scenario, which can be different from the empirical canopy data used during the calibration stage, and therefore  $SCFN_t^*$  is particular for each simulation.

$$APG_{(t+1)} = SCFN_t^* \times SCFNE_t \quad (5)$$

where:  $APG_{(t+1)}$  = annual potential growth for a given species in the next time step. During the simulation stage, the model interpolates between the different curves of SCFNE calculated before to find the site quality of the simulated site.

Nutrient uptake requirements to support APG are calculated based on rates of biomass growth and data on nutrient concentration in the different biomass components. Nutrient availability is calculated based on empirical data describing litter and humus decomposition rates, changes in chemistry as decomposition proceeds, and the size of nutrient pools in the mineral soil and humus (cation exchange capacity (CEC) and anion exchange capacity (AEC), respectively). If the availability of nutrients for each time step is less than required to support APG, vegetation growth is limited by nutrients and the realized annual growth is lower than APG.

Nitrogen cycling in FORECAST is based on a mass balance approach (Figure 2) where N can exist in three distinct pools: 1) the plant biomass pool; 2) the available soil nutrient pool, and 3) the soil organic matter/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 in FORECAST can be assimilated to represent the interchangeable N present in the soil during one year as  $\text{NH}_4^+$ ,  $\text{NO}_3^-$  or labile organic N fractions with turnover rates shorter than one year. N deposition and N fixed by bryophytes and other microorganisms are simulated as constant annual N fluxes that directly reach the soil solution and are incorporated into the available N pool. Available nitrogen pool is calculated by simulating consecutively the different inputs and outputs of the biogeochemical cycle: deposition, fertilization, seepage, leaching, mineralization, immobilization (Figure 2). A detailed description of the simulation of each of these fluxes in FORECAST can be found in Kimmins et al. (1999) and Blanco et al. (2012). The definition of site fertility based on N availability assumes that soil moisture is not limiting in these sites (McWilliams and Thérien, 1997). However, soil moisture is still implicitly affecting the simulation by the use of the parameter “maximum foliage per tree” which is directly correlated with soil moisture availability (Kimmins et al., 1999).

Carbon and nitrogen cycles are linked through the use of the foliar nitrogen efficiency as the driving function of the model (amount of biomass generated in a year per kg of foliar N). Therefore, a limitation in N uptake will result in a reduction of foliar N, reducing biomass produced by the trees. Nutrient uptake demands on sites of different N fertility are based on observed biomass accumulation rates and tissue nutrient concentrations on these sites, allowing for internal cycling of nutrients.

### **2.3. Management simulation, model validation, and sensitivity analysis**

Three different management plans were simulated based on the data from the field trials and common management activities in the region aimed to generate three different types of forest products: 1) timber (producing the maximum number of trees suitable for sawmills); 2) pulp (production of stem and branch biomass with high content in cellulose and hemicelluloses for paper and chemical products); and 3) biomass (maximizing the mass for burning after being transformed into pellets, oil, charcoal, ethanol or other product to produce energy to heat water or steam to be transformed into electricity) (Crown and Brett, 1975; SFM, 2008). These management prescriptions are an illustration of the range of possible forest practices that are currently being implemented or under discussion in British Columbia.



Management details for each scenario are described in Table 1.

All the management simulations in each site started with the same initial conditions and ended at year 150. Salal was simulated to colonize the stands at year 3 (Stanek et al., 1979). To simulate forest growth after human disturbance, ecosystem conditions at the end of each 150-year run were recorded. They became the starting conditions for the post-management runs, which were simulated as several cycles of natural regeneration. These cycles were 200 years of tree growth without any human intervention, ending with a stand-replacing windstorm (the average return time of stand-replacing storm events on Vancouver Island). An additional baseline run was created for each site by simulating non-managed stands starting with the pre-management conditions.

Validation of model projections was carried out by comparing model output for the timber scenario at the rich site with data for the trial plots in Shawnigan Lake. Published field data were used on merchantable volume, dominant height and DBH, salal biomass and SOM mass, combined with field data collected by the Canadian Forest Service until 2007 (Crown and Brett, 1975; McWilliams and Thérien, 1997; Messier and Mitchell, 1994; Preston and Newman, 1995; Stanek et al., 1979). Unfortunately, no field data suitable for validation could be found for the poor site. Model performance was assessed with several measures of goodness-of-fit: coefficient of regression, modelling efficiency and Theil's inequality coefficient (Theil, 1966; Vanclary and Skovsgaard, 1997). In addition, model bias was characterized with the average bias and the mean absolute deviation.

Kimmins et al. (1999) identified a list of parameters for which the model was the most sensitive. Among them, two parameters stand out due to the practical difficulty of obtaining data to estimate them and the uncertainty usually associated to their estimated values: wood decomposition rate and fine root turnover (Kimmins et al., 2004). To study the sensitivity of the model to changes in those parameters, the calibration values were modified in +20%, +10%, -10%, -20%, and the changes in the two main target variables (total tree biomass and soil organic matter) were recorded.



### 3. Results

#### 3.1. Changes in ecosystem during management

The model slightly over-predicted aboveground biomass (Table 2), but its bias decreased with stand age (Figure 3). Volume was under-predicted, but after 40 years the difference was only 10% of the observed values. Dominant height and diameter also produced biases lower than 5% of the recorded values after 40 years. All variables had low Theil's coefficients and high values of modelling efficiency and regression coefficients (Table 2). Predicted salal biomass and SOM mass (corresponding to the humus fraction, without accounting for litter) were very similar to field data, but the number of ages with data for comparison was scarce (Figure 3). Sensitivity analysis showed that relative changes in tree biomass and SOM were always smaller than changes in the calibration parameters. The modification in a tree physiological parameter (fine root turnover) affected more tree biomass than SOM, whereas changes in a soil parameter (wood decomposition rate) affected more SOM than tree biomass (Table 3). The recovery patterns were not significantly modified by changes in the parameter values at any site (see Figures S2 to S5 in the Supplementary Material).

For the bioenergy scenario, tree biomass at both sites was kept low due to continuous harvesting and biomass removals, and it showed a decreasing trend over time (Figures 3 and 4). The pulp scenario also showed a decreasing trend after the first rotation. On the other hand, tree biomass in the timber scenario increased after one rotation. The highest harvested biomass and volume were found when managing for timber, with management for bioenergy being the least productive at both site qualities (Table 4).

A clear reduction of SOM mass was projected for both bioenergy and pulp scenarios, being especially severe in the case of managing for bioenergy at the rich site, which lost about 40% of the initial SOM mass after 150 years of management. On the other hand, when managing for timber, SOM initially decreased but then recovered, reaching stabilization after ~125 years (Figures 3 and 4).

Salal biomass expanded quickly at the beginning of each rotation at both sites, when the trees were small and did not create much shade (Figures 3 and 4). In the pulp and timber scenarios it reached a



peak around year 25 and then it quickly decreased as available light was also quickly reduced after canopy closure. In the case of management for timber, salal remained with a minimum biomass until the end of the rotation as the understory light level was enough to support this residual biomass even after canopy closure. Finally, ecosystem carbon for the bioenergy and pulp scenarios also followed a decreasing pattern at both sites, with sharp decreases following the extraction of biomass after the end of each rotation. These post-harvest reductions in ecosystem C were much gentler in the timber scenario as the biomass left on site (branches, leaves, snags and logs from dead trees) also contributed to ecosystem C (Figures 3 and 4).

### 3.2. Changes in the ecosystem after the end of active management

After the end of active management for bioenergy and pulp, tree biomass at both sites was lower than in the non-managed forest, but the biomass production was increased after the first stand-replacing windstorm. However, the post-timber scenario was virtually identical to the baseline right after the end of active management. At both sites there were similar levels of available N in the baseline and post-timber scenarios. However, after managing for pulp and for bioenergy the ecosystem needed one cycle of natural disturbance to recover similar N levels. Accumulated N losses were much higher for the bioenergy scenario than for the others, with management for timber remaining neutral (rich site) or even positive (poor site) (Table 5). A similar situation could be observed for ecosystem C. For the post-bioenergy scenario at the rich site, 3 to 4 disturbance cycles were needed by all the variables to reach values similar to the non-managed forest (Table 6).

At both sites, the levels of SOM mass increased in all the scenarios, but only in the post-timber scenario the levels were similar to the non-managed forest after one cycle of disturbance. The stand-replacing windthrow generated a clear signal in the non-management scenario, with one peak in SOM ~20 years after the storm corresponding to the highly decomposed branches and roots that became humus and therefore were included in the SOM pool. A second, bigger peak ~80 years after the storms corresponded to the residual stem biomass than was incorporated into the SOM after being decomposed.



SOM was later reduced as the mass lost by SOM decomposing was higher than the litterfall inputs. At both sites, similar signals were seen in the post-timber scenario and after one disturbance cycle in the post-bioenergy and post-pulp scenarios (Figures 5 and 6).

Maximum understory biomass increased in consecutive disturbance cycles at both sites (Figures 5 and 6). At the rich site, the post-timber scenario was very similar to the baseline right after the end of the active management. However, understory biomass in the post-bioenergy scenario was more different from the baseline in the second disturbance cycle than in the first one (Figure 6). At the poor site, only the post-timber scenario had understory biomass similar to the baseline during the 400 years simulated, whereas the other two scenarios kept much lower understory biomasses (Figure 6). The slower recovery in the post-bioenergy scenario means that understory biomass may need 4 disturbance cycles (800 years) at both sites to reach values similar to non-managed forest (Table 6).

## 4. Discussion

### 4.1. Model evaluation and limitations

The  $R^2$  coefficient indicated acceptable agreement between observed and predicted values. Linear regression can be considered as a "hypothetical re-calibration" (Mayer and Butler, 1993) in which the model minimizes the differences with the observed data. However, it has been argued that it is not the most reliable measure of model performance (Power, 1983), because the  $R^2$  coefficient is not related to the "perfect fit" line (the line in which observed equals predicted). As a consequence, this coefficient is more about model's capacity to get a calibration data set to reduce differences between observed and predicted values rather than a measure of the "perfection" of model's predictions. Modelling efficiency has been proposed as an important overall measure of fit by Mayer and Butler (1993) and was also recommended by Power (1993) and Smith et al. (1997), because modelling efficiency is a dimensionless statistic which directly relates model predictions to observed data. Modelling efficiency also showed values close to 1, indicating good model performance. Similarly, Theil's coefficient was always lower than 1, indicating that the model was a better predictor than a general mean value.



FORECAST has also shown good agreement with field data in other ecosystem types, such as semitropical, tropical and boreal forests (Bi et al., 2007; Blanco and González, 2010; Seely et al., 2008). In addition, field data from natural forests indicated that the model output for the post-management period was also in the natural range for these forests. For example, Zobel et al. (1976) reported a range of 734-1773 Mg ha<sup>-1</sup> of total tree biomass for mature Douglas-fir forests. Model projections were well inside this range, with an average of 915 and 856 Mg ha<sup>-1</sup> after 400 years of no management at the rich and poor sites, respectively. Model projections of SOM were similar to estimations by Harmon et al. (2004) (186 Mg ha<sup>-1</sup>). For ecosystem C the average model predictions at year 400 were very similar to field records of 619 Mg ha<sup>-1</sup> (Harmon et al., 2004) or 635 Mg ha<sup>-1</sup> (Sollins et al., 1980) (696 and 618 Mg ha<sup>-1</sup>, rich and poor site respectively). In addition, the model was only moderately sensitive to two of the main tree and soil parameters, showing its capability to reduce error propagation through the simulation while simulating the long-time effects of changing those parameters (Kimmins et al. 2010). Such model performance provides confidence on the temporal patterns and relative differences among scenarios and sites, but the exact values predicted for each variable should be taken cautiously given the long time scales involved in this work.

Natural disturbances other than windthrow (i.e. pests and diseases infestations) are also present in Douglas-fir forests. However, they commonly cause patchy and limited mortality at stand level, and rarely cause the replacement of the whole stand (Cruickshank et al., 2009, 2011; Negrón et al., 2001). These and other causes of low-level mortality events are already included in the simulations. Therefore, the effects of non-stand-replacing disturbances such as wildfires, insects, diseases or climate change were not included to avoid increasing complexity unnecessarily (Kimmins et al., 2008). However, these disturbances could alter some of the post-management change rates, although the relative differences among treatments would likely be similar. Finally, the weakest component of the model calibration was the understory vegetation, which traditionally has received less attention than the trees. Only scarce and disperse documentation is available for understory biomass in mature and old-growth forest at these sites, which also have a high variability of understory cover (Messier and Mitchell, 1994). Therefore,





calibrating the understory component comprised a large degree of uncertainty. However, the estimated maximum salal total biomass (9.1 and 13.3 Mg ha<sup>-1</sup> in the first and second disturbance cycles) was in the same range as values previously reported for the same region (6.2 to 12.1 Mg ha<sup>-1</sup>; Messier and Mitchell, 1994).

#### 4.2. Reduction of forest productivity under intensive management

Shorter and intensive rotations depleted nutrients and SOM rapidly, as young trees had proportionally more crown biomass and are richer in nutrients than mature trees. SOM reduction was especially severe in the case of management for bioenergy, which in turn reduced the amount of nutrients being mineralized, consequently limiting tree growth. The decline in forest productivity increased through time in a non-linear pattern, with productivity losses bigger from the first to the second rotations than at any other time. This phenomenon has been observed in other conifer plantations around the world (Bi et al., 2007; Blanco et al., 2005; Fox, 2000; Johnson, 1992). Only the timber scenario returned to pre-harvest SOM levels within a few decades.

It seems unlikely that fertilization alone could prevent the long-term effects of intensive forestry on nutrient availability, at least at the simulated fertilization rates. Although in the bioenergy scenario a total of 1000 kg N ha<sup>-1</sup> were applied during 150 years, these N additions did not prevent the long-term decrease in forest productivity. SOM and CEC were reduced over time, causing a reduction in the amount of fertilizer that could be stored in the soil. Therefore, fertilization efficiency would also have been reduced and fertilization losses increased. This result supports Eisenbies et al. (2009) suggestion that to keep stable tree growth rates, fertilizer doses should have increased in every rotation, especially at the rich site, by 45-60% when managed for bioenergy. Assuming that only massive N applications can change long-term productivity and site quality seems more adequate (Weetman, 1984). However, increasing the frequency or intensity of N fertilization may have other environmental consequences, such as increasing N leaching to watercourses and groundwater (Wei et al., 2012). 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 non-managed forests. As a consequence, salal would grow more, competing more intensively with trees and therefore reinforcing nutrient limitation. The spread of salal in Douglas-fir forests caused by intense management has been reported before (Thysell and Carey, 2000).

#### **4.3. Tree growth after the end of forest management**

Tree biomass after management for bioenergy (and also for pulp in the poor site) reached pre-management values after 400 years of ecosystem recovery. However, understory biomass may need up to 800 years or more to reach pre-management values. Duffy and Meier (1992) also reported differences in species abundance and composition between managed and non-managed forests persisting after 87 years after the end of active management. Intensive management could not only reduce tree productivity temporarily, but they have also the potential to push the ecosystem across a threshold to reach a new state with lower understory biomass that could last up to 800 years before reverting to pre-management conditions. This phenomenon could be missed if ecosystem recovery is assessed only with indicators related to tree growth. Therefore, results presented here support the simultaneous use of ecological indicators for several species (Lattimore et al., 2009).

SOM is the key component of forest ecosystems, supporting site fertility and productivity by releasing nutrients, providing soil structure, and increasing water holding capacity (Kimmins, 2004). Sharp declines in SOM after harvesting followed by recovery have been described in chronosequence studies (Federer, 1984; Martin et al., 2002). In the simulations presented here, SOM and available N followed the pattern previously described for soils in the same region: first a small peak after harvesting, then decrease for about 15 years and then slow increase (Martin et al., 2002). The small peak about 20-30 years after stand initiation was a consequence of the humification of the massive input of woody debris left on the stand after the disturbance (Kimmins, 2004; Martin et al., 2002).

Soil nutrient and carbon stocks could also be used as measures of recovery of ecosystem structure and function (Silver et al., 1996). During the post-management period, SOM accumulated faster in the most degraded conditions due to a positive feedback: soil nutrient capital increased, more N was released,



nutrient limitation was reduced, and trees grew more and produced more leaf litter with high N content, which decomposed to become SOM. SOM eventually accumulated to a point when the amount of SOM mineralized each year equalled the amount of litterfall produced, reaching equilibrium.

Post-timber stands had similar patterns of C storage to the non-managed forest. However, the post-pulp and post-bioenergy stands stored substantially less carbon after 200 years. Previous observations have described greater C storage in natural forest relative to those managed for timber production (Harmon et al., 1990; Kurz et al., 1992; Seely et al., 2002). This phenomenon could last beyond the phase of active management into the period of forest recovery. This may have important implications for climate-change related policies, although a detailed C balance would be needed (accounting for fuel used in forest operations, export and residence time of forest products, etc.). If intensive management is used in these Douglas-fir plantations to produce bioenergy as a substitute of fossil fuels (as has been encouraged in this region by the B.C. Ministry of Forests; Hamilton, 2008), the amount of carbon released to the atmosphere from fossil fuels could be decreased in the short term, but on the other hand the amount of carbon released from forests soils could increase in the long term. As a consequence, C storage in these sites could be reduced for a very long time. This trade-off should be accounted for when dealing with the issue of increasing bioenergy production from forests to reduce C emissions.

## 5. Conclusions

Stands managed for timber are the only ones that would become similar to the non-management scenarios in a few decades after the end of active management. Therefore, fast recovery of forest fertility from human management seems to be possible, but low-impact management techniques should be used that keep adequate levels of nutrients, SOM, and other ecological structures at the end of the human intervention. In any site quality, avoiding the continuous repetition of the most intensive management regimes in the same stands could reduce the risk of overexploitation. Otherwise, several centuries may be needed before forests can reach a condition similar to non-managed forests. To do so, different



management intensities should be alternated in the same stand. Periods of intensive extraction (i.e. management for bioenergy) could be followed by management for timber, allowing the forest to recover some of its fertility attributes, or even by avoiding managing the forest for a time similar to one timber rotation. Therefore, intensive management schemes can produce a flow of forest products in the short-term, but to keep this management sustainable in the long term different management intensities should be implemented in a mosaic at landscape level. Natural disturbances have an important role in restoring the forest to a more natural state, with dramatic improvements in the bioenergy and pulp scenarios, especially at the poor site.

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## 7. Supplementary material

A detailed model diagram, values used to calibrate the model FORECAST for Douglas-fir plantations in Shawnigan Lake (Vancouver Island, BC, Canada), and results of the sensitivity analyses are shown in the Supplementary Material, available online.

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**Table 1.** Definition of simulation runs for three different types of management. Both pre- and commercial thinnings were thinning from below (smaller trees removed first). Stem-only harvesting only extracted the stems with diameter 12.5 cm or higher. Whole-tree harvesting extracted all the aboveground tree biomass. Slash/CWD harvesting extracted branches, logs, snags and other major dead biomass components. Understory harvesting extracts all understory aboveground biomass. Total simulation time was 150 years for all scenarios.

Management regime	Production of Timber	Production of Pulp/Fiber	Production of Biomass
Objective	Maximizing production of stems larger than 12.5 cm at DBH <sup>a</sup>	Producing biomass of high fiber content	Maximizing flow of forest biomass
Initial stand density <sup>b</sup>	5000 trees ha <sup>-1</sup>	5000 trees ha <sup>-1</sup>	5000 trees ha <sup>-1</sup>
Pre-commercial thinning	40% trees removed at year 15	40% trees removed at year 15	-
Fertilization	100 kg ha <sup>-1</sup> N at year 16	100 kg ha <sup>-1</sup> N at year 16	100 kg ha <sup>-1</sup> N at year 5
Commercial thinning	30% trees removed at years 25 and 50	-	-
Rotation length	75 years	30 years	15 years
Fraction harvested	Stems	Whole trees	Whole trees + slash/CWD + understory
Number of rotations	2	5	10

<sup>a</sup> DBH: diameter at breast height (1.30 m).

<sup>b</sup> Historical plantation density reported for Shawnigan Lake plots (Crown and Brett, 1975).



**Table 2.** Comparisons of model predictions versus field estimations. Historical field data reported by McWilliams and Thérien (1997). Estimations of total biomass calculated from field measurements by the Canadian Forest Service combined with allometric equations by Barclay et al. (1986). A more detailed validation of FORECAST for these sites can be found in Blanco et al. (2007).

<b>Measurement of model performance</b>	<b>Merchantable volume</b>	<b>Total biomass</b>	<b>Top height</b>	<b>Dominant DBH</b>
Average bias	-16.50 m <sup>3</sup> ha <sup>-1</sup>	13.16 Mg ha <sup>-1</sup>	-1.05 m	1.12 cm
Mean absolute deviation	21.25 m <sup>3</sup> ha <sup>-1</sup>	13.16 Mg ha <sup>-1</sup>	1.18 m	1.12 cm
Adjusted R <sup>2</sup>	0.98	0.95	0.98	0.94
Theil's inequality coefficient	0.13	0.14	0.06	0.08
Modelling efficiency	0.96	0.87	0.92	0.91

**Table 3.** Sensitivity analysis, measured as relative change in the maximum value of target variables to changes in two main parameter values: wood decomposition rates and fine root mortality rate. Parameter calibration values were modified by +20%, +10%, -10%, and -20% (see Figures S.2 to S.5, supplementary material)

Site	Management type	Wood decomposition rate				Fine roots mortality rate			
		+20%	+10%	-10%	-20%	+20%	+10%	-10%	-20%
TOTAL TREE BIOMASS									
Rich site	Baseline	4.9	2.4	-2.8	-4.4	-1.5	-0.6	1.1	2.1
	Bioenergy	4.2	1.2	-0.7	-2.5	-5.6	-2.6	4.6	8.9
	Pulp	4.9	2.3	-3.0	-6.8	-0.5	-0.3	0.5	0.7
	Timber	5.2	3.0	-3.6	-6.9	-0.9	-0.6	0.7	1.5
Poor site	Baseline	10.9	8.8	-7.3	-7.8	-0.7	-0.3	0.5	1.0
	Bioenergy	1.9	0.0	-0.9	-1.5	-5.2	-1.8	2.2	5.1
	Pulp	3.3	2.7	-0.9	-2.2	-1.8	1.1	4.7	6.4
	Timber	1.9	0.0	-13.2	-14.1	-8.5	-8.7	-8.4	-8.5
SOIL ORGANIC MATTER									
Rich site	Baseline	-7.4	-3.7	4.5	8.2	1.8	0.8	0.2	-1.0
	Bioenergy	-3.2	-2.8	1.8	4.2	0.0	0.1	0.9	1.6
	Pulp	-5.9	-3.4	2.8	5.3	-0.6	-0.4	2.5	4.5
	Timber	-7.3	-4.1	4.3	-1.7	1.6	1.1	-0.8	-1.7
Poor site	Baseline	-8.88	-4.70	3.04	7.96	0.24	0.07	-0.32	-0.44
	Bioenergy	-5.3	-2.5	2.9	5.0	-1.5	-1.3	0.7	0.7
	Pulp	-5.9	-3.4	2.8	5.3	-0.6	-0.4	2.5	4.5
	Timber	-4.7	0.0	8.1	13.9	5.0	5.1	4.1	3.6

**Table 4.** Accumulated values of total aboveground biomass and merchantable volumes harvested after 150 years of management in two sites with different quality at Shawnigan Lake.

Site quality	Management regime	Total merchantable volume harvested m <sup>3</sup> ha <sup>-1</sup>	Total biomass harvested <sup>a</sup> Mg ha <sup>-1</sup>
Rich	Bioenergy	0	502.7
	Pulp	170	679.2
	Timber	1021	736.3
Poor	Bioenergy	0	278.4
	Pulp	42	429.9
	Timber	870	510.9

<sup>a</sup> See Table 1 for a description of the biomass fractions harvested in each scenario.



**Table 5.** Net effects in ecosystem N after 150 years under different management regimes.

Site quality Management	Rich site			Poor site		
	Bioenergy	Pulp	Timber	Bioenergy	Pulp	Timber
<b>Nitrogen imports (kg ha<sup>-1</sup>)</b>						
Fertilization	1000	500	200	1000	500	200
Atmospheric deposition	375	375	375	375	375	375
Non-symbiotic fixation	150	150	150	150	150	150
(1) TOTAL	1525	1025	725	1525	1025	725
<b>Nitrogen exports (kg ha<sup>-1</sup>)</b>						
Harvested biomass	1299	1553	453	661	853	387
Slash/CWD removal	697	0	0	562	0	0
Leaching	832	261	273	1067	391	224
(2) TOTAL	2828	1814	726	2290	1244	611
<b>Nitrogen balance (kg ha<sup>-1</sup>)</b>						
(1 – 2)	-1303	-789	-1	-765	-219	114

**Table 6.** Number of natural disturbance events needed to recover pre-management values for different variables. Each disturbance event accounts for a 200-year period ending in stand-replacing windthrow.

Site quality Previous management	Rich site			Poor site		
	Bioenergy	Pulp	Timber	Bioenergy	Pulp	Timber
Aboveground biomass	3	2	1	3	2	1
Merchantable volume	3	2	1	2	2	1
Ecosystem Carbon	3	2	1	3	2	1
Understory biomass	4	2	1	4	4	1
Available N	4	3	1	3	2	1
Soil Organic matter mass	3	3	1	3	3	1



**Figure 1.** Schematic representation of key ecosystem processes and interactions (black dotted lines), and mass flows between ecosystem pools (black solid lines). Light and nutrient limitations were simulated explicitly, whereas soil moisture limitation was simulated implicitly (see text).

**Figure 2.** Estimation of available N in FORECAST in each annual time step. Step 1: geochemical inputs were calculated, with all the forms of N lumped together. Step 2: biochemical fluxes. Step 3: Plants uptake the available N. Step 4: Soil N remaining for next time step is calculated by subtracting the remaining N from the soil CEC (for ammonium) or AEC (for nitrate). The N excess was assumed to be lost via leaching (for a complete description of the simulation of these processes see Kimmins et al., 1999).

**Figure 3.** Simulated temporal evolution of several stand-level variables of a Douglas-fir plantation in southern Vancouver Island (British Columbia, Canada) established in a rich site (site index 26 m), under three different management regimes (described in detail in Table 1), for a total of 150 years of simulation. Historical field data (black dots) of biomass and merchantable volume were provided by McWilliams and Theri n (1997), understory biomass by Stanek et al. (1979); and SOM biomass were calculated from soil analysis by Crown and Brett (1975) and Preston et al. (1995).

**Figure 4.** Simulated temporal evolution of several stand-level variables of a Douglas-fir plantation in southern Vancouver Island (British Columbia, Canada) placed in a poor site (site index 16 m), under three different management regimes (described in detail in Table 1), for a total of 150 years of simulation.

**Figure 5.** Simulated temporal evolution of several stand-level variables of a Douglas-fir natural forest regenerated after ending three different management regimes in southern Vancouver Island (British Columbia, Canada) placed in a rich site (site index 26 m), for a total of 400 years of simulation.

**Figure 6.** Simulated temporal evolution of several stand-level variables of a Douglas-fir natural forest regenerated after ending three different management regimes in southern Vancouver Island (British Columbia, Canada) placed in a rich site (site index 16 m), for a total of 400 years of simulation.



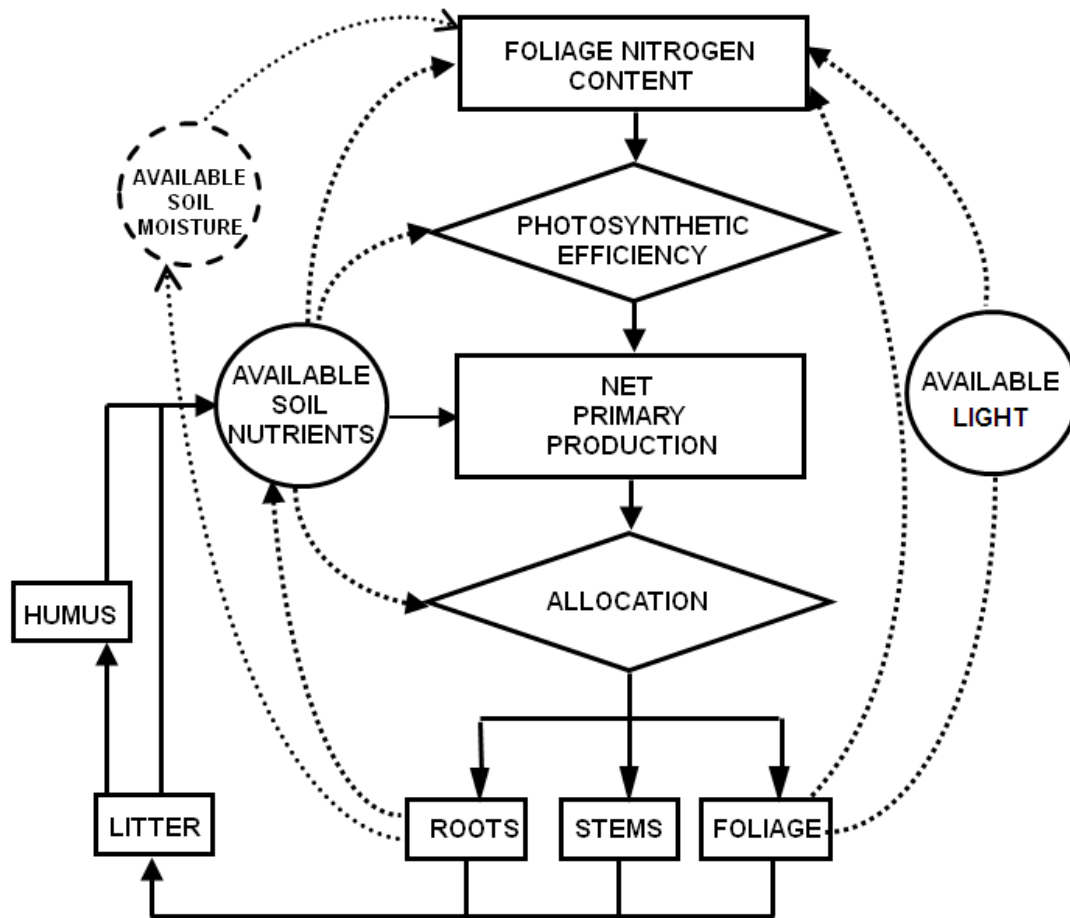


Figure 1.

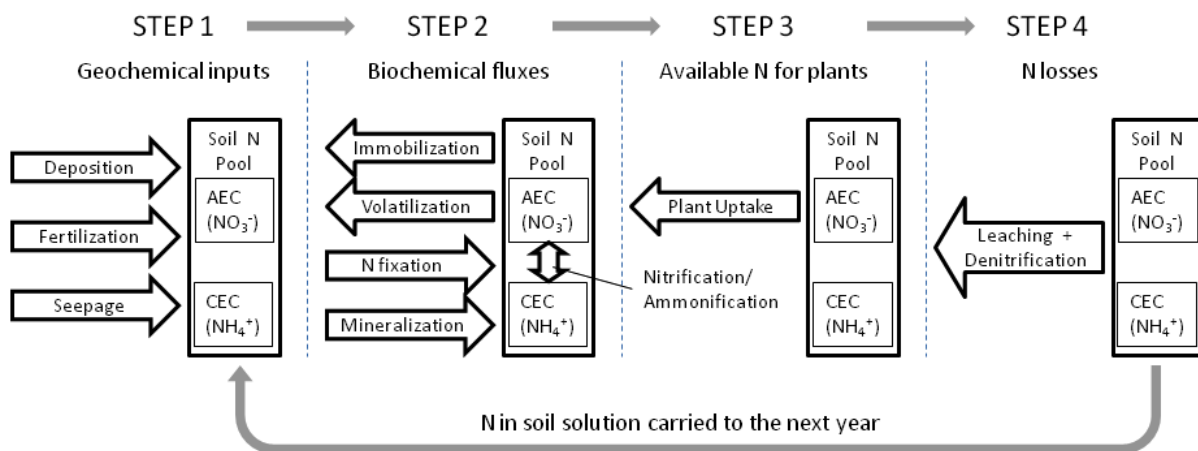


Figure 2.

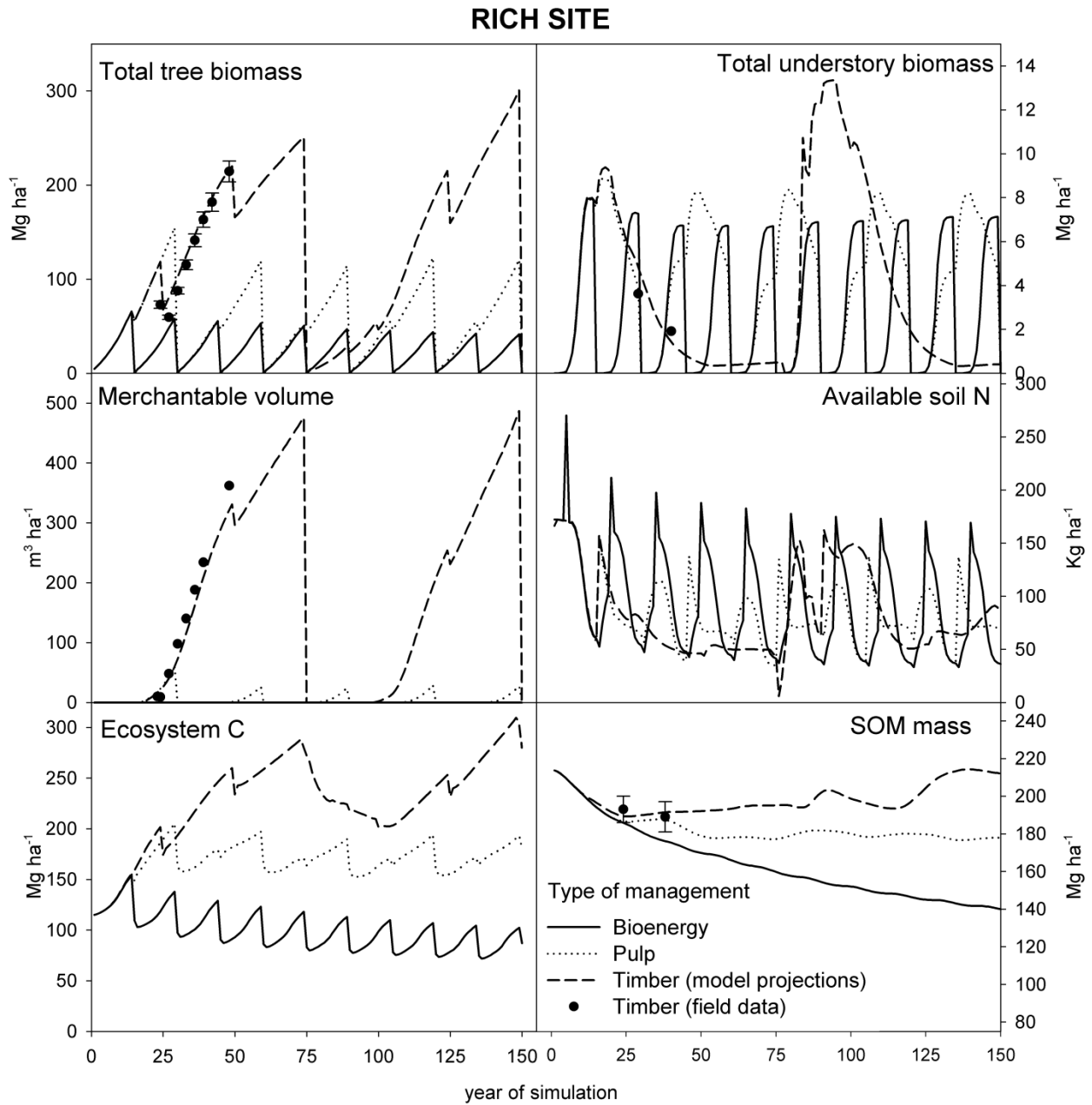


Figure 3.



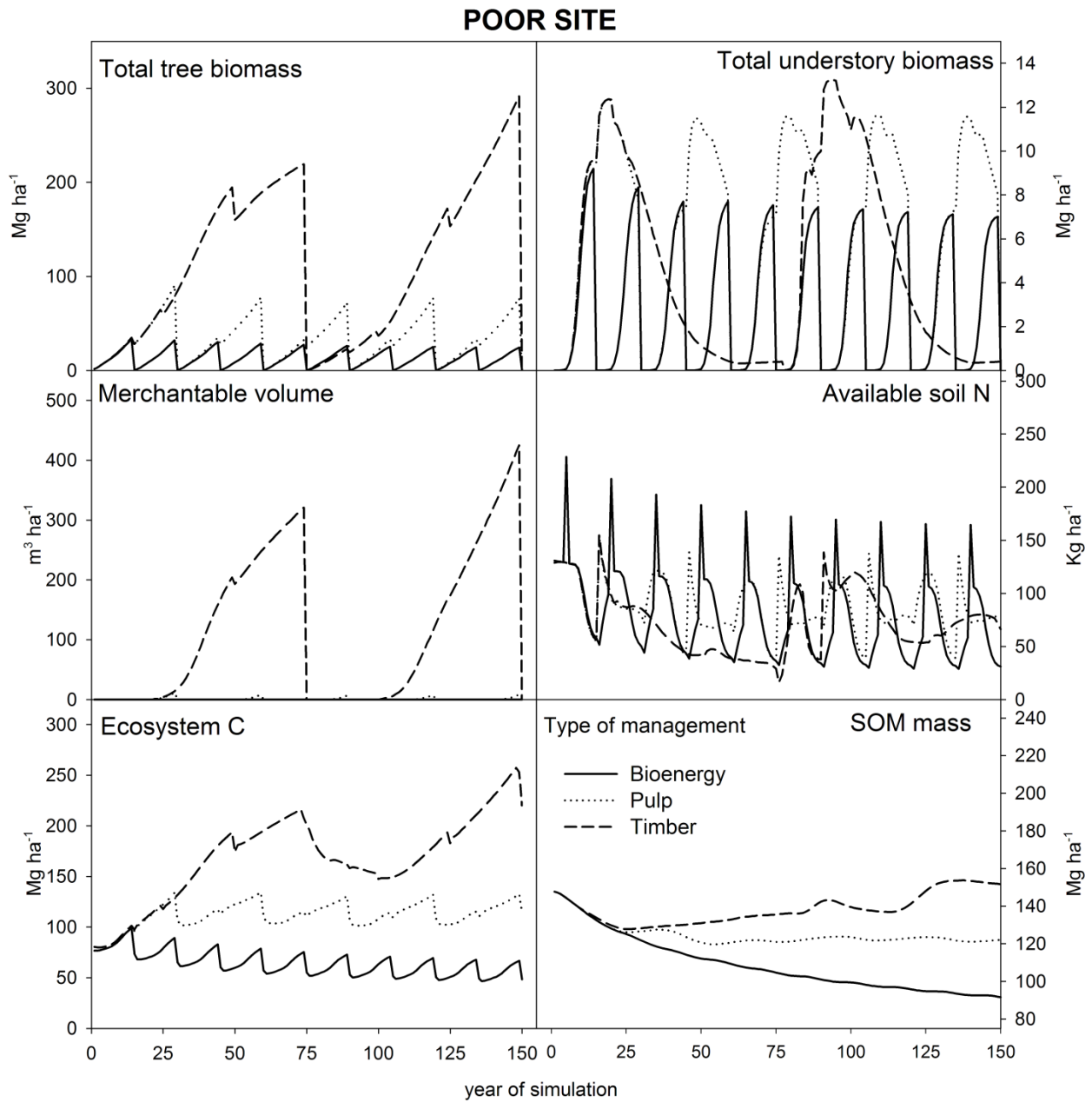


Figure 4.



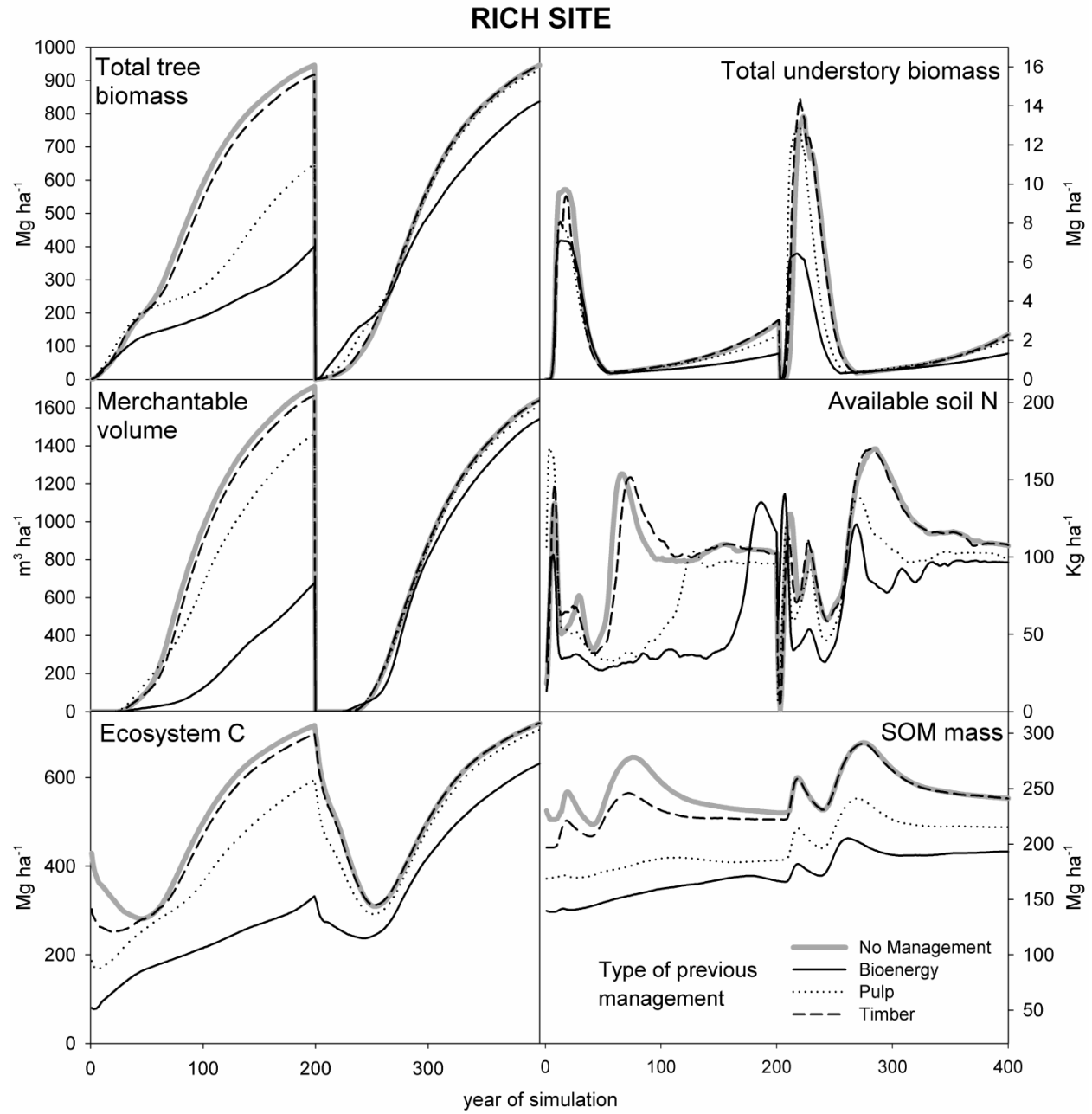


Figure 5.



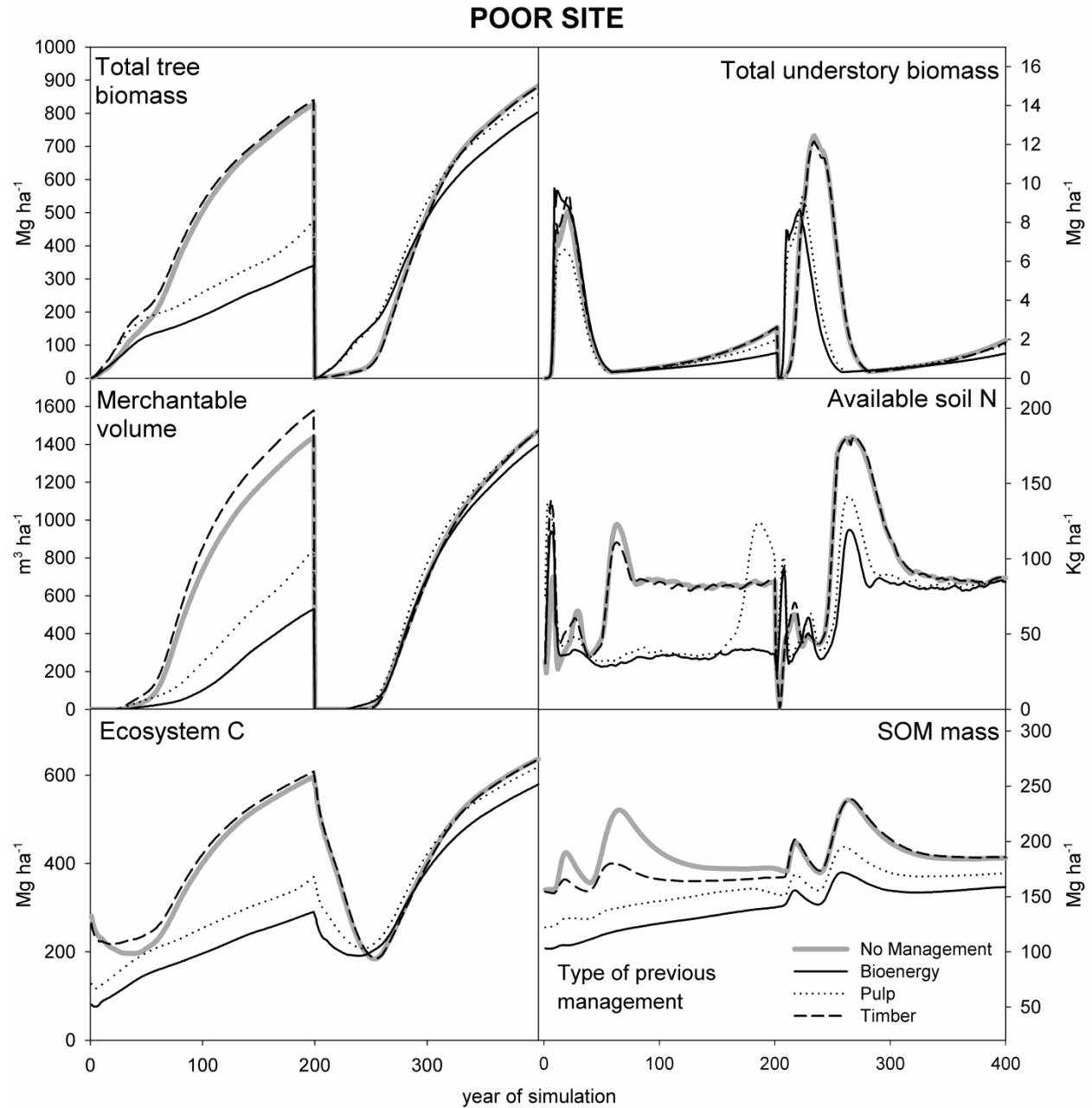


Figure 6.

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