

# **Models for the Sustainable Management of Temperate Plantation Forests**

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Jean-Michel Carnus, Roderick Dewar, Denis Loustau,  
Margarida Tomé and Christophe Orazio (eds.)

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## Foreword

The workshop “Models for the Sustainable Management of Temperate Plantation Forests” was held at the INRA Centre de Recherches de Bordeaux-Aquitaine, 7–9 September, 2000. This event was organised by the Institut Européen de la Forêt Cultivée (IEFC) – an EFI Regional Project Centre – as a satellite activity of the 7<sup>th</sup> EFI annual conference in Lisbon, 2–4 September, 2000. This Proceedings is the first publication produced by a Regional Project Centre of EFI.

The aims of the workshop were: (i) to review the current use of models in understanding and quantifying the sustainable management of plantation forests; (ii) to encourage links between tree and soil modellers; (iii) to encourage links between researchers and end-users; and (iv) to identify priorities for future research and collaboration.

The programme of talks and discussions was attended by 54 participants from 12 countries, including 18 students from the Ecole Nationale d’Ingénieurs des Travaux Agricoles (ENITA) in Bordeaux. A visit was made to the experimental site of INRA’s Forest Research Station at Pierroton.

The present selection of papers from the workshop provides an overview of recent progress in developing and applying models to problems of sustainable forest management. The papers cover the full range from detailed, process-based ecosystem models to empirical growth and yield models and decision-support tools.

IEFC is organizing a follow-up meeting in Portugal (6–8 June, 2002). Further information on this meeting and other IEFC activities can be obtained from the IEFC web site [www.iefc.net](http://www.iefc.net).

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# Maximizing Wood Yield, Carbon Storage and Efficient Use of N

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## Abstract

It is argued that, in most circumstances, unmanaged forests are likely to store more carbon (in trees and soil) than plantations which are clearfelled to maximize volume yield (with carbon stored in trees, soil and wood products) assuming no change in species or site conditions. But is there a simple trade-off between carbon storage and volume yield? And which harvesting regime is most efficient in the use of nitrogen?

A mechanistic forest ecosystem simulator, which couples carbon, nitrogen and water (the Edinburgh Forest Model) was calibrated to mimic the growth of a pine plantation in a Scottish climate. It was then run to equilibrium with various harvesting regimes.

More carbon was stored in an unmanaged forest than in any regime in which wood was harvested ( $35.2 \text{ kgC m}^{-2}$ ). Plantations, clearfelled for maximum volume yield, gave moderate carbon storage ( $14.3 \text{ kgC m}^{-2}$ ) and timber yield ( $15.6 \text{ m}^3 \text{ ha}^{-1}\text{yr}^{-1}$ ). Annual removal of 10 or 20% of woody biomass per year gave both a high timber yield ( $25 \text{ m}^3 \text{ ha}^{-1}\text{yr}^{-1}$ ) and high carbon storage ( $20$  to  $24 \text{ kgC m}^{-2}$ ). The efficiency of the latter regimes could be attributed (in the model) to high light interception and net primary productivity, but less evapotranspiration and summer water stress than in the unmanaged forest, high litter input to the soil giving high soil carbon and  $\text{N}_2$  fixation, low maintenance respiration and low nitrate leaching owing to soil mineral pool depletion.

It was concluded that there is no simple inverse relationship between the amount of timber harvested from a forest and the amount of carbon stored. In theory, management regimes which maintain a continuous canopy cover and mimic, to some extent, regular natural forest disturbance, might achieve the best combination of high volume yield, carbon storage and low loss of nitrogen to the environment. There may, however, be biological, economic and practical constraints on realizing this theoretical ideal.

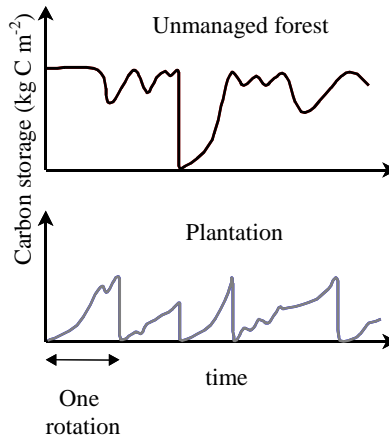
*Keywords: carbon, wood yield, nitrogen, sequestration, management, model, plantation*

## 1. Introduction

Three of the many objectives of forest management are: (i) to maximise wood yield; (ii) to maximise carbon storage in biomass, soils and wood products; and (iii) to minimise losses of nitrogen as nitrate to ground-waters and  $N_2O$  to the atmosphere. These are all aspects of C and N management and involve tradeoffs. The Edinburgh Forest Model, which simulates the flows of C, N and water in forest ecosystems, has enabled the tradeoffs to be explored (Thornley and Cannell 2000).

What, in theory, would be the best way to manage forests to optimise the three objectives stated above? The answer is not self evident, because of the many interactions and feedbacks between plant and soil processes in a forest ecosystem, involving light, nutrients and water. Different management regimes perturb the system in different ways. Also, the answer would be difficult to derive by experimentation, because it would take centuries before valid estimates of sustained yield and carbon storage could be made. Transient responses would depend on the initial conditions and could differ in sign as well as magnitude from the equilibrium response. A model that represents all the essential interacting processes offers a way forward.

In this paper, we first examine the evidence that unmanaged forests store more carbon and 'leak' less nitrogen than plantations. We then present model results for a range of theoretical harvesting regimes, which suggest that regular small harvests of wood may maximize both yields and carbon storage and give most control over nitrogen losses to the environment.



**Figure 1.** Notional changes in the amount of carbon stored in biomass in an unmanaged forest and four rotations of a plantation on the same site. Both are subject to periodic disturbance by fire or pest outbreaks. The plantation is assumed to be clear-felled at the time of maximum mean annual volume increment.

## 2. Unmanaged Forests Compared With Plantations

### 2.1 Carbon storage in biomass

At first glance, it might be supposed that more carbon may be removed from the atmosphere by replacing old, slowly growing forests with young, fast-growing plantations, and then



harvesting and storing the wood. But such reasoning ignores the fact that old undisturbed forests can contain huge amounts of living and dead biomass and plantations are often felled well before they reach maturity. Harmon et al. (1990) found that 450-year-old Douglas fir/western hemlock forests in the Pacific Northwest contain over twice as much carbon as 60-year-old Douglas fir plantations, and that this is also true for a landscape of old-growth forests in different stages of recovery from natural disturbance compared with a landscape of plantations of different ages.

If plantations are harvested at the time of maximum mean annual increment, it can be shown that, at equilibrium, the time-averaged carbon storage in the trees – from planting to clearfelling – is only about one-third of the maximum amount of carbon stored in the forest at maturity – with a variety of assumed sigmoidal growth curves (Cooper 1983; Dewar 1991). In other words, in theory, up to two-thirds of the carbon stored in trees in a mature, old-growth forest is eventually lost (i.e. transferred to the atmosphere) when it is replaced by plantations harvested for maximum volume yield.

In practice, the loss of carbon may be less than two-thirds. First, both the unmanaged and plantation forests will be subject to periodic disturbance, as illustrated in Figure 1. Disturbance may lower carbon storage proportionately more in unmanaged forests than in plantations. Secondly, plantation forests may be managed to maximize timber values rather than volumes. Greatest values may occur when forests are left to grow past the time of maximum mean annual volume increment, further towards maturity. Thirdly, old-growth forests may be replaced by faster-growing tree species, with site improvements, and so follow a steeper sigmoidal growth curve than the native species. And fourthly, we must include the carbon stored in the wood products derived from plantations, which may have a longer lifetime than dead trees in a natural forest.

If all of the above four factors operate, then the difference in carbon storage between unmanaged and managed forests could be small. However, if we make the assumption that plantations are felled to maximize timber volumes, with no change in species or site conditions, it becomes difficult to construct scenarios in which equilibrium carbon storage is not substantially lowered by replacing unmanaged forests with plantations. Calculations made for European forests showed that plantations would have the same carbon storage capacity as unmanaged forests only if the average wood product lifetime was 2–3 times the rotation period – a scenario which seems unlikely without incurring carbon costs in wood preservation (Cannell et al. 1992).

## 2.2 Carbon storage in forest soils

In theory, it seems inevitable that equilibrium carbon storage in soils should be greater in unmanaged forests than in plantations at the same site which are clear-felled for maximum volume yield. The reason is that equilibrium soil carbon mass is proportional to total annual litter input. At equilibrium, total litter input to the soil is equal to net primary production. And net primary production is roughly proportional to intercepted solar radiation (see Waring et al. 1998). Because unmanaged forests maintain a continuous canopy they are likely, over time, to intercept more solar radiation than plantations, have a higher net productivity and greater litter input to the soil. Consequently, at equilibrium, unmanaged forests may be expected to have more soil carbon. Also, soil disturbance during site preparation is likely to deplete soil carbon in plantation forests.

However, over short timescales, in the first years or decades following conversion of unmanaged forests to plantations (well before equilibrium is reached) it may be difficult to detect a depletion in soil carbon. Thus, Johnson (1992) found no general trend towards lower

soil carbon following forest harvesting and site preparation in a review of thirteen studies. Also, when fertilizers are added, and fast-growing or nitrogen-fixing trees are planted, net primary production of plantations may exceed that of the native forest, even when averaged over a rotation, giving more soil carbon.

### **2.3 Carbon storage predictions of the Edinburgh Forest Model**

The Edinburgh Forest Model was parameterised to simulate the growth of a pine forest and was run to equilibrium in the climate of southern Scotland to simulate an unmanaged and plantation forest, with the same basic growth characteristics (see Thornley and Cannell 2000). The plantation was clearfelled every 60 years, at approximately the time of maximum mean annual volume increment, and yielded  $15.6 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$  of timber.

At equilibrium, the unmanaged forest contained  $35.3 \text{ kgC m}^{-2}$ , consisting of  $13.2 \text{ kgC m}^{-2}$  in the trees and  $22.1 \text{ kgC m}^{-2}$  in the soil. By contrast, the mean carbon content of the plantation was only  $14.2 \text{ kgC m}^{-2}$ , less than half of that in the unmanaged forest. The plantation contained an average of only  $3.8 \text{ kgC m}^{-2}$  in trees,  $6.4 \text{ kgC m}^{-2}$  in soil and  $4.0 \text{ kgC m}^{-2}$  in wood products, which were assumed to have a half-life of 20 years.

Thus, if the species and site conditions remain unchanged and the plantation is clearfelled to obtain maximum volume yield, the model suggests that total carbon storage, including wood products, will be decreased by transforming native forests into plantations.

### **2.4 Nitrogen leakage**

Nitrogen is input to forests from the atmosphere and by fixation and is lost by: (i) the removal of products; (ii) leaching of nitrate; and (iii) gaseous losses associated with nitrification and denitrification.

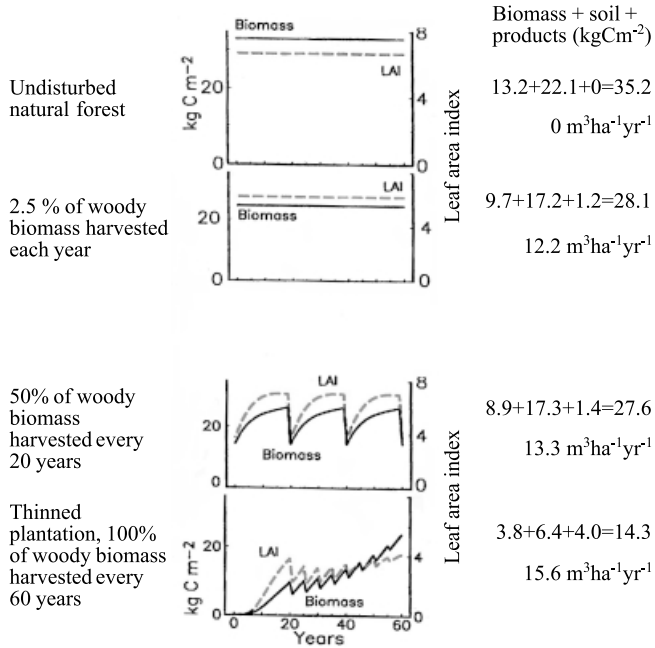
Plantations are prone to leaching nitrogen after site preparation and clearfelling, when soil mineral nitrate pools are not depleted by tree growth and there may be increased drainage of water through the soil (e.g. Fenn et al. 1998). However, plantations have the advantage that much of the nitrogen is removed in products and so is under human control. Unmanaged forests, subject to the same inputs, are likely to leach less nitrogen, but at equilibrium must lose more nitrogen to the environment, primarily as gases.

### **2.5 Conclusion**

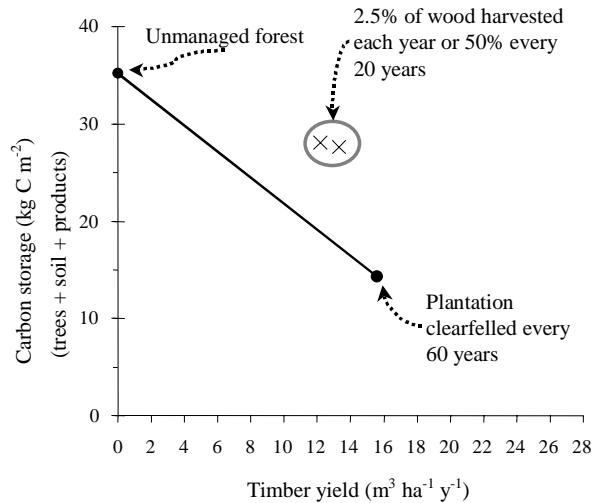
The comparison between unmanaged forests and plantations leads to the conclusion that timber harvesting carries the risk of lessening carbon storage, even including wood products. Harvesting gives some control over nitrogen losses, but risks enhanced nitrate leaching.

At first sight, it may be supposed that the more timber that is harvested from a forest the less carbon is stored. But, if timber were removed by regularly thinning, without clearfelling, would it be theoretically possible to obtain a high sustained yield of timber, more control over nitrogen losses and a large store of carbon? Are there tradeoffs between these objectives or is there an optimum management regime?

Below, we use the Edinburgh Forest Model to estimate sustained timber volume yield, carbon storage and nitrogen budgets in forests subjected to different harvesting regimes. The model is parameterised, as above, to simulate a pine forest in the climate of Scotland, but the principles elucidated may apply to other species in other environments.



**Figure 2.** Simulated effects of four forest management regimes on forest biomass (continuous lines), leaf area index (dashed lines) and long-term equilibrium values for carbon storage in biomass, soil and wood products (the three numbers on the right, respectively; thus 13.2+22.1+0 means biomass+soil+products) and mean wood volume yield. The four regimes are: undisturbed forest; removing 2.5% of woody biomass each year; removing 50% of woody biomass every 20 years; and thinned plantation. The simulations were made using the Edinburgh Forest Model with parameters for a pine forest in the climate of southern Scotland.



**Figure 3.** Theoretical relationship between long-term, equilibrium carbon storage (in trees plus soil and wood products) and mean wood volume yield. Values are taken from the simulations shown in Figure 2. Harvesting 2.5% woody biomass per year or 50% every 20 years, enables more carbon to be stored than expected based on a linear interpolation between unmanaged and plantation forests.

### 3. Theoretical Effects of Repeatedly Harvesting Small Amounts of Wood

#### 3.1 Removing 2.5% of woody biomass every year or 50% every 20 years

The model was run to equilibrium when removing 2.5% of the total above and belowground woody biomass each year, or 50% every 20 years, the numerical equivalent. These harvestings might consist of thinnings or prunings. In this study no account is taken of the quality or value of the harvested wood and it is assumed that there are no biological constraints to regrowth or regeneration other than light, water and nutrient supplies..

Removing just 2.5% of woody biomass each year yielded  $12.2 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ , 78% of that yielded by the plantation forest, while storing a total of  $28.1 \text{ kgC m}^{-2}$  in biomass, soil and products, twice that stored in the plantation forest (Figure 2). These numbers may seem remarkably high. They are theoretically possible because the forest maintains a continuously high leaf area index (~6) intercepting most of the radiation and maintaining a continuously high net primary productivity. Removing 50% of the woody biomass every 20 years gave similarly high productivities, wood yields and carbon storage (Figure 2). Thus, thinning did not have to be done every year to achieve a combination of high yield and carbon storage. Within limits, it was the average removal rate that was important, giving flexibility to adopt various thinning regimes.

The straight line in Figure 3 shows the simple trade-off between carbon storage and volume yield predicted by linearly interpolating between unmanaged forests with plantations, clearfelled for maximum volume yield. Harvesting 2.5% of woody biomass each year, or 50% every 20 years, gave points above that line – storing more carbon than expected for the volume yield, or a greater yield for the amount of carbon stored. Regular harvesting may therefore be a better means of achieving these dual objectives.

#### 3.2 Removing 10% or 20% of woody biomass every year

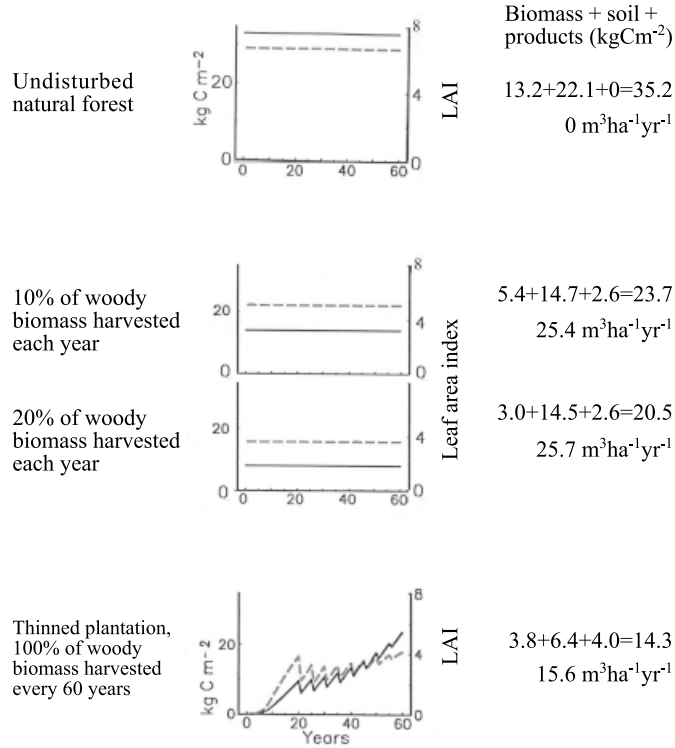
More surprisingly, thinning or pruning the forest to remove 10% or 20% of the woody biomass each year yielded a remarkable  $25.4$  and  $25.7 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ , respectively, about 60% more than that yielded by the plantation forest (Figure 4) while storing a total of  $23.7$  and  $20.5 \text{ kgC m}^{-2}$ , respectively, significantly more than stored in the plantation forest. Similarly high yields and carbon storage were predicted by varying the harvesting interval, within limits, but removing the same percentage of woody biomass (e.g. removing 30% every 3 years rather than 10% every year).

Figure 5 shows that these harvesting regimes gave points well off the linear interpolation, calling into question the notion that any simple tradeoff between carbon storage and volume yield really exists. Regularly thinning forests seem to result in stands that are potentially considerably more efficient at generating, storing and yielding carbon than conventionally clearfelled plantations.

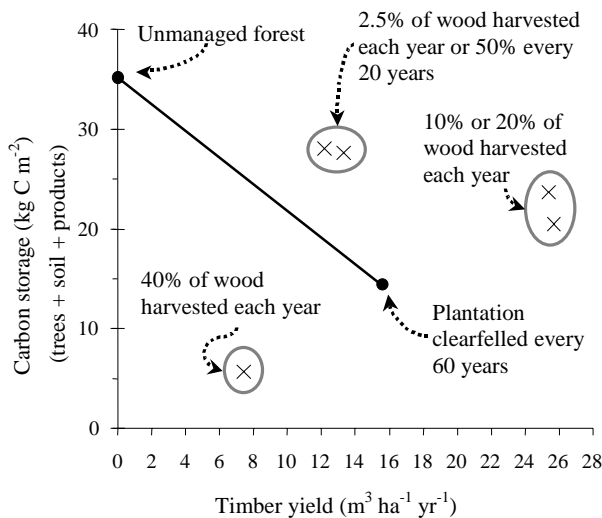
However, there is clearly a limit to the severity of annual thinning, beyond which the forest has such a low leaf area index that it is no longer productive. This limit was surpassed with the removal of 40% woody biomass every year, which resulted in very low leaf areas, yield and carbon storage (Figure 5).

#### 3.3 The nitrogen budget of regularly thinned stands

As expected, in the unmanaged forest, leaching losses were small, because soil water drainage was small, and the N balance was maintained by gaseous loss. In the plantation forest, almost



**Figure 4.** Simulated effects of four forest management regimes on forest biomass (continuous lines), leaf area index (dashed lines) and long-term equilibrium values for carbon storage in biomass, soil and wood products (the three numbers on the right, respectively) and mean wood volume yield. The four regimes are undisturbed forest, removing 10% of woody biomass each year, removing 20% of woody biomass each year, and thinned plantation. The simulations were made using the Edinburgh Forest Model with parameters for a pine forest in the climate of southern Scotland.



**Figure 5.** Theoretical relationship between long-term, equilibrium carbon storage (in trees plus soil and wood products) and mean wood volume yield, for a range of forest management regimes.

half of the N output was in harvested products, but leaching losses were also relatively high, owing to mineralization of litter and high drainage during the years after clearfelling.

In the 10 and 20% annually thinned natural forests, about 47% of the nitrogen output was in harvested products, while leaching losses were no greater than in the undisturbed forest, owing to low drainage and continued depletion of the soil mineral pools by root uptake (driven by tree growth). Thus, these thinned natural forests avoided the N loss to groundwaters that occurred in plantations and yet captured (in wood products) a substantial fraction of the N that was otherwise lost as gases in undisturbed forests (Thornley and Cannell 2000).

Thus, the regularly thinned forests used nitrogen efficiently as well as excelling in carbon storage and volume yield.

### **3.4 Reasons for the high performance for regularly thinned stands**

As mentioned, the high performance of the 10 and 20% thinned natural forests could be traced to high canopy photosynthesis and net primary productivity. This resulted from a combination of moderately high light interception (with sustained leaf area indices about 4), and also lower evapotranspiration and less water stress on summer days than in the undisturbed forest, plus a lower respiratory load (net/gross photosynthesis was about 0.65 compared with 0.60 in plantations and undisturbed forests; Thornley and Cannell 2000).

The biomass in these forests was obviously decreased by thinning, compared to the undisturbed forest, but even with 20% of the biomass removed each year, there was 3.0 kgC m<sup>-2</sup> in biomass, about 80% of that in the plantation forest averaged over a rotation. More importantly, high sustained net primary production maintained a high input of litter to the soil, generating over twice as much soil carbon in forests subject to 10 and 20% annual thinning as in the plantation forest.

## **4. Discussion**

### **4.1 Concluded principle**

The salient conclusions from this study are that (i) unmanaged forests will normally store more carbon than plantations clearfelled to maximize volume yield, including the wood product store, but (ii) there is no simple inverse relationship between the amount of timber harvested from a forest and the amount of carbon stored in the ecosystem and wood products. The method of harvesting is all important. In particular, regular removal of timber from a forest (annually or every few years) in a way that maintains a continuous canopy can, in theory, give substantially higher sustained yields *and* levels of carbon storage than periodical clearfelling, as in conventional plantations. Runs of the model in other temperate climates and with different calibrations and nitrogen deposition levels suggested that, qualitatively, this conclusion might apply in a wide range of forest types.

*It should be stressed, however, that we claim only to have elucidated a principle, not to have made quantitative predictions, much less to have considered the practicalities, costs and other implications of different harvesting regimes. We are aware that, in some situations, there may be biological constraints that may limit forest recovery from thinning, which are not included in the Edinburgh Forest Model, and the assumption that plantations are managed for maximum volume yield does not always apply.*

## 4.2 Essential features of unmanaged, plantation and regularly thinned forests

In the model, differences between unmanaged, plantation, and regularly thinned forests in average leaf area and biomass affected evapotranspiration and light interception, which in turn affected the carbon and nitrogen dynamics.

The *unmanaged forest* maintained a high leaf area index and biomass, giving high light interception, but also high evapotranspiration and summer water stress, so that net primary production was only moderately high. However, leaching losses were small (because of low drainage) and, because no biomass was removed, soil organic matter levels and  $N_2$  fixation rates were moderately high. The net result was high carbon sequestration in biomass and soils, but, of course, no yield and a high loss of nitrogen as gases.

The *plantation forest* had a low leaf area index and biomass, averaged over a rotation, resulting in relatively low light interception, low evapotranspiration, high drainage and nitrate leaching, leading to low net primary production, despite relatively little summer water stress. Low net primary production, combined with the removal of wood, resulted in low soil organic matter levels and  $N_2$  fixation rates. The outcome was a combination of moderate carbon sequestration, moderate wood yield, and considerable nitrogen loss to groundwaters.

The *regularly thinned forests*, which were managed to remove about 10% of the biomass each year (or 50% every 5 years) maintained a moderately high, continuous leaf area and biomass, giving moderate light interception and evapotranspiration, resulting in relatively little summer water stress. Consequently, net primary production was high, sustaining both a high yield and litter input to the soil, resulting in moderately high soil organic matter levels and high  $N_2$  fixation rates. The outcome was moderately high carbon sequestration and high wood yield.

## 4.3 Tradeoffs between multiple objectives

The study suggested that, if the objective were simply to maximize timber volume yield (regardless of cost or value) the order of optimal management system would be regularly thinned forest > plantation > undisturbed forest, whereas if the objective were to maximize carbon storage, the order would be undisturbed forest > regularly thinned forest > plantation (Figure 5). If the objective were to minimize uncontrolled N emissions to the environment, the order would be the same as that to maximize timber volume yield.

In the simulations, more carbon was stored in the undisturbed forest than in any management regime in which wood was harvested, including wood products, supporting previous analyses which compared plantations with undisturbed forests (see Cannell 1995) and suggesting that any biomass removal from a forest will lower carbon storage, without unrealistic assumptions on the rates of decay of harvested wood.

Plantations offered the worst combination of yield and carbon storage and regularly thinned forests the best, provided thinning removed not less than about 5% and not more than about 25% of the woody biomass each year. There seemed to be some scope for flexibility in thinning frequency, in that, for instance, harvesting 5% of the woody biomass per year gave similar results to harvesting 25% every 5 years.

## 4.4 Efficiency of regularly thinned forests

Why were regularly thinned forests so efficient in the model? The main features were as follows.

- The continuous canopy and moderately high leaf area index (about 4) gave high light interception and net primary productivity; there was no period of slow recovery, which occurs after clearfelling.
- However, the leaf area index was less than in an undisturbed forest and so evapotranspiration was less, with less risk of water stress on dry summer days, also enhancing net primary productivity.
- High net primary productivity ensured high litter input to the soil and a large equilibrium soil carbon store, also favouring non-symbiotic N<sub>2</sub> fixation.
- Regular thinning meant that the forest had a lower biomass than an undisturbed forest and was continually growing, resulting in less maintenance respiration.
- Continuous growth also meant that the soil mineral nitrate pool remained depleted and nitrogen losses by leaching were reduced.

#### 4.5 General remarks

The conclusion that regular thinning is better than clearfelling is in keeping with much of the current discussion regarding forest management to sustain multiple functions, including the maintenance of biodiversity, maintaining soil fertility, preventing erosion and so on (Gale and Cordray 1991; Wiersum 1995). It is increasingly recognized that the many demands made on forests may best be satisfied by maintaining an 'intact forest ecosystem' or 'ecological integrity' (Armstrong 1999). This analysis offers some scientific basis for those concepts with regard to carbon dynamics in the soil-plant system, as affected by nitrogen dynamics and the water balance.

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# Stand Thinning Effects on C Fluxes in 20<sup>th</sup> and 21<sup>st</sup> Century Scots Pine Forests: a Sensitivity Analysis Using ASPECTS

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## Abstract

Stand thinning effects on present and future productivity and carbon sequestration in a Scots pine forest were examined using the ASPECTS model. Downscaled GCM scenarios were generated from the Canadian Global Coupled Model (CGCM1) and two years of on-site measurements at the Brasschaat experimental forest. Simulated stand thinning had major effects on predicted forest carbon fluxes, due to reduced photosynthesis and autotrophic respiration, and increased heterotrophic respiration. This result implies that the history of stand thinning needs to be considered for accurate simulations of the net ecosystem exchange (NEE) of carbon between forest ecosystems and the atmosphere. ASPECTS predicted that the productivity of Belgian Scots pine forests will increase under 21<sup>st</sup> century climate, while NEE will become more variable due to (i) the effect of thinning an increased woody biomass, and (ii) increased variability of the 21<sup>st</sup> century climate.

*Keywords: thinning, ASPECTS, modelling, NEE, climate change*

## 1. Introduction

Concerns about global environmental change have led to numerous research programmes to assess the potential mitigating effects of temperate forests on the increase in atmospheric CO<sub>2</sub> concentration. The net CO<sub>2</sub> flux between the atmosphere and temperate forest canopies has been measured by eddy-covariance techniques in a variety of European and North American forests (Greco and Baldocchi 1996; Valentini et al. 1996; Baldocchi 1997; Valentini et al. 2000). Some of these monitoring programmes have now accumulated 4 years of continuous

NEE data. While this data represents a tremendous amount of information, it is of short duration relative to the life span of a forest. Measured CO<sub>2</sub> fluxes in forest ecosystems result from the combined effects of short-term and long-term processes (Rasse et al. 2001a). For example, autotrophic respiration is a function of: (1) temperature, which displays substantial sub-hourly variations; (2) tissue N concentration, which potentially varies in the course of the growing season; and (3) total biomass and the fraction of sapwood, which depend on stand age and long-term forest growth.

Until now, modelling studies of forest NEE, using eddy-covariance measurements for calibration and validation, have mostly focused on the effects of the daily and seasonal cycles of climatic variables such as temperature, radiation, and water vapour pressure deficit (VPD) (Baldocchi and Harley 1995; Baldocchi 1997; Law et al. 2000). Nevertheless, models are also needed for long-term simulations of forest growth, in order to predict the carbon stored in forest ecosystems over the 21<sup>st</sup> century (Grant and Nalder 2000). These longer-term growth processes, in addition to plot management history, potentially affect current NEE measurements, although they have received little attention until now (Rasse et al. 2001b). Stand thinning generates large amounts of fresh litter from leaves, branches, fine roots and coarse roots. The decomposition of this litter will affect the soil CO<sub>2</sub> efflux for many years following thinning, especially for woody branch and coarse root tissues.

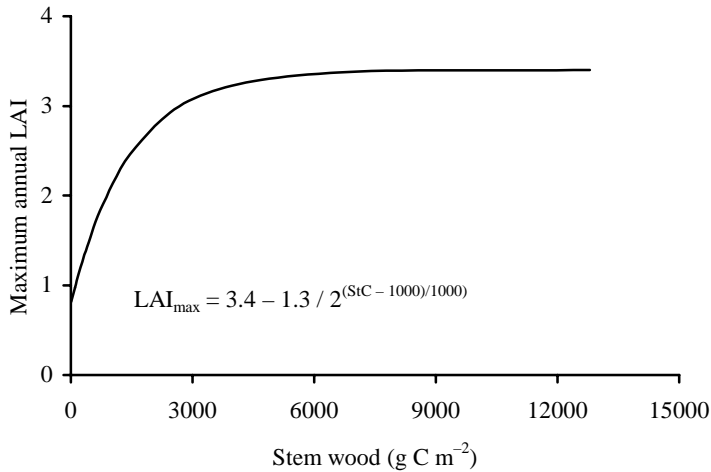
Temperate European forests are nearly all subjected to strict management plans by foresters. Stand thinning is an essential element of forest management, which is conducted throughout the life span of a forest. The first objective of this modelling study was to estimate the effects of 20<sup>th</sup> century stand thinning on current CO<sub>2</sub> fluxes in temperate forests. The second objective was to predict forest responses to 21<sup>st</sup> century environmental conditions under different scenarios of stand thinning. To meet both of these objectives, we used the ASPECTS model because of its ability to predict long-term forest growth while predicting all C fluxes within the forest ecosystem on a half-hourly basis (Rasse et al. 2000; Rasse et al. 2001a).

## **2. Model Development**

### **2.1 General description of the ASPECTS model**

ASPECTS is a mechanistic model designed to predict the dynamics of carbon reservoirs in temperate forest ecosystems (Rasse et al. 2000; Rasse et al. 2001a). Nine carbon reservoirs are represented: sugar, starch, foliage, branches, stems, coarse roots, fine roots, soil litter and soil organic matter (SOM). Because the carbon cycle is closely linked to the water cycle, a complete hydrological module was included in ASPECTS. The evolution of carbon and water reservoirs is computed on half-hourly time steps as the difference between incoming and outgoing fluxes. Although the integration time step is short, ASPECTS is designed to simulate the evolution of carbon and water reservoirs over periods longer than a century. Initial conditions are defined for forests of any age, i.e. from seedlings to mature stands, and ASPECTS simulates the subsequent evolution of each carbon pool, including tree growth.

ASPECTS simulates two phenological phases for evergreen trees: (1) leaf expansion in the spring; and (2) no leaf growth during the rest of the year. Bud burst is triggered when degree-days accumulated since February 10 over a base temperature of 5°C reach a thermal time requirement specific to each tree species, as proposed by Hoffman (1995). Cessation of leaf growth is implemented when the leaf area index (LAI) reaches a maximum value defined by an allometric relationship with wood biomass (Figure 1).



**Figure 1.** Maximum annual LAI as a function of stemwood biomass. After thinning, the function is reduced by an LAI reduction term given in equations 1 to 4.

### Photosynthesis and respiration

ASPECTS computes rates of canopy photosynthesis according to the theoretical model of de Pury and Farquhar (1997), which is a big-leaf model with a separate integration of sunlit- and shaded-leaf photosynthesis. Within the leaf photosynthesis sub-model, stomatal conductance is computed by the semi-empirical model of Leuning (1995), which relates stomatal conductance to net assimilation, the CO<sub>2</sub> concentration at the leaf surface, and the water vapour pressure deficit. In addition, a feedback of soil water stress on stomatal conductance has been introduced. Maintenance respiration costs are computed for each plant organ according to a generalised version of the equation proposed by Zogg et al. (1996), which relates maintenance respiration to the total carbon content of each plant organ, the fraction of living tissue for the reservoir, tissue nitrogen concentration, and plant tissue temperature. Growth respiration is computed as 20% of growth assimilates allocated to each reservoir, as suggested by Hoffman (1995). In addition, growth respiration of fine roots is increased as a function of soil water and temperature stresses (Rasse et al. 2001b). This increase in growth respiration represents the additional energetic costs associated with fine root growth in stress conditions.

### Allocation

Photosynthetically-fixed carbon is stored in the sugar pool. This pool regulates carbon allocation to plant organs, and is necessary because ASPECTS computes photosynthesis at short time steps. Sucrose (*Su*) is allocated to tree organs to satisfy their need for maintenance respiration and growth, which includes both biomass increment and growth respiration. ASPECTS allocates sucrose in priority to maintenance respiration. Growth of tree organs is then conditional upon a supply of assimilates from the sugar pool exceeding the demand from maintenance respiration.

The ASPECTS model partitions assimilates between above- and below-ground organs according to phase-dependent ratios. During the leaf expansion phase, all assimilates are

allocated to non-woody tissues, i.e. leaves and fine roots, as suggested by Lüdeke et al. (1994). During the rest of the year, allocation in ASPECTS is stand-age dependent because studies have shown that the root to shoot ratio of temperate trees decreases asymptotically with time from saplings to mature stands (Vanninen et al. 1996; Lee et al. 1998). Growth assimilates allocated to above-ground organs are then partitioned among leaves, branches and stems, while below-ground assimilates are partitioned between coarse and fine roots.

### **Litter production and decomposition**

Needle senescence from evergreen trees is simulated at a constant rate throughout the year. Turnover time of fine roots is a crucial parameter that drives the largest carbon input to the soil organic matter. Published data for evergreen and deciduous trees indicate that the duration of fine root turnover cycles approximates 10 to 12 months (Harris et al. 1977; Usman et al. 1997). Because the life span of fine roots is difficult to estimate precisely, we chose a value of one year. Litter decomposition is calculated as a simple function of soil water content, temperature, and pH. The temperature dependence of litter decomposition is described in ASPECTS according to an equation specific to forests (Nemry et al. 1996).

### **Hydrology and soil temperature**

In ASPECTS, soil water content is computed in a series of user-defined soil layers. The net flux of water between two adjacent soil horizons is computed by solving Richards' equation for unsaturated flow, according to the methodology of Viterbo and Beljaars (1995). The hydraulic conductivity and the relationship between the volumetric water content ( $\theta$ ) of each layer and its pressure head are parameterised according to Saxton et al. (1986). The bottom water flow boundary condition is free drainage, i.e.,  $\partial\theta/\partial z = 0$ , where  $z$  is the depth. Evaporation from the soil surface which defines the upper boundary condition was computed following Mahfouf and Noilhan (1991). Total water uptake by the tree, which is simulated in the photosynthesis and stomatal conductance subroutines, is distributed among the various soil layers according to the root density and the water and aeration stresses of each layer. ASPECTS also simulates soil temperature for each soil layer by solving the heat diffusion equation, with a bottom boundary condition set to zero heat flux, and an upper boundary condition defined by equating soil surface temperature to air temperature.

## **2.2 Simulation of stand thinning**

Based on literature data, we assumed that the normal planting density for Scots pine is approximately 8000 trees ha<sup>-1</sup> (Kramer and Ross 1989; Ruha and Varmola 1997). This high density is rapidly reduced to less than 3000 trees ha<sup>-1</sup> by pre-commercial thinning operations. Stand thinning is simulated by decreasing all tree carbon reservoirs by the proportion of wood biomass removed from the forest plots. Biomass removal data are not always available. Data on the reduction in stand density at thinning is more readily available than biomass removal data. These two proportions are not equal because foresters tend to eliminate the smallest trees first, which implies that the decrease in wood biomass is smaller than the decrease in stand density. We estimated the proportion of wood biomass removal in thinnings by dividing the reduction in stand density by a factor 1.625, as recalculated from published data (Tullus et al. 1989).

Following thinning, LAI is immediately decreased in proportion to the amount of leaf biomass lost. The removal of trees produces gaps in the canopy, thus reducing the maximum annual LAI reached by the forest canopy for several years following stand thinning. In theory, the relationship built into the model between maximum annual LAI and stemwood biomass (Figure 1) automatically leads to a reduction in LAI when stem wood is harvested. Nevertheless, this relationship alone does not lead to a proper estimation of long-term thinning effects on LAI, because it predicts more important thinning effects on younger stands. No thinning effects would be simulated when the relationship between LAI and stemwood biomass has reached a plateau (Figure 1). In reality, young stands recover from canopy gaps faster than older stands, especially because bigger gaps result from the removal of older trees as compared to younger trees.

In ASPECTS we introduced an LAI reduction term, which is calculated at thinning as:

$$LAI_{red} = LAI_{max,pot} \times f_{cut} \quad \text{[Equation 1]}$$

where:  $LAI_{red}$  is the immediate reduction in maximum annual LAI due to thinning,  $LAI_{max,pot}$  is the potential maximum annual LAI as defined in Figure 1, and  $f_{cut}$  is the proportion of woody biomass that is thinned. After thinning, the recovery in LAI is simulated by a decrease in  $LAI_{red}$  over time, according to:

$$LAI_{red(y)} = LAI_{red(y-1)} \times (1 - 1/t_r) \quad \text{[Equation 2]}$$

where:  $LAI_{red(y)}$  and  $LAI_{red(y-1)}$  are the reduction terms in years  $y$  and  $y-1$ , respectively, and  $t_r$  (yr) is proportional to the half-life of  $LAI_{red}$ . As mentioned above,  $t_r$  is a function of stand age ( $Y$ , yr):

$$t_r = 1.0, \quad Y \leq 5 \quad \text{[Equation 3]}$$

$$t_r = Y / 5.0, \quad 5 \leq Y \leq 100$$

$$t_r = 20.0, \quad Y \geq 100$$

The actual maximum annual LAI ( $LAI_{max}$ ) in year  $y$  is then:

$$LAI_{max} = LAI_{max,pot} - LAI_{red(y)} \quad \text{[Equation 4]}$$

### 3. Climate Scenarios

A set of synthetic weather data was constructed on a half-hourly basis for the 20<sup>th</sup> and 21<sup>st</sup> centuries from: (1) two years of measured meteorological data; and (2) outputs of a general circulation model (GCM) for the period 1900–2100. Six meteorological variables were measured every half-hour during 1997 and 1998 at the Brasschaat experimental forest: solar radiation ( $W m^{-2}$ ), air temperature ( $^{\circ}C$ ), precipitation ( $mm day^{-1}$ ), relative humidity ( $kg kg^{-1}$ ), wind speed ( $m s^{-1}$ ) and atmospheric pressure (Pa), as described by Kowalski et al. (1999).

We used simulated weather data from the Canadian Global Coupled Model (CGCM1) (McFarlane et al. 1992; Flato et al. 2000) of the Canadian Center for Climate Modeling and

Analysis (CCCMA). The CGCM1 model computes the atmospheric general circulation in response to a standard increase in the concentration of atmospheric carbon dioxide (CO<sub>2</sub>) over the 20<sup>th</sup> and 21<sup>st</sup> centuries. The radiative forcing of the climate system includes a warming by greenhouse gases and a cooling by sulphate aerosols (Reader and Boer 1998; Boer et al. 2000a,b). The equilibrium climate sensitivity of CGCM1, i.e., the global-mean temperature response to a doubling of the effective CO<sub>2</sub> concentration, is about 2.5°C. Monthly results are available in a database maintained by the Data Distribution Center of the Intergovernmental Panel on Climate Change (DDC-IPCC). The spatial resolution is 3.75° in longitude and about 3.7° in latitude.

Monthly averages of CGCM1-simulated weather data were extracted for the grid cell corresponding to the Brasschaat research site (51°18' N, 4°31' E). The downscaling was conducted in a two-step approach. First, monthly GCM outputs were modified so that their 1997–1998 average value for each month was equal to that of the measured data. The correction factors obtained for the period 1997–1998 were then applied to the entire period 1900–2100. This operation is necessary because grid-cell GCM outputs do not necessarily correspond to local measurements conducted in a specific set of altitudinal and latitudinal conditions within the grid cell.

The second step of the downscaling procedure consisted in adding the half-hourly variability to the corrected GCM outputs. Simulated monthly averages for the 1900–2100 period were compared to the corresponding month of the year of the 1997–1998 measured data. To each month of the GCM simulations was ascribed the variability of the measured data which presented the closest monthly average to that of the simulations. This process was conducted for each of the six weather variables. The half-hourly measured data were then multiplied by the ratio between the monthly averages of the corrected CGM outputs and the measured data. Because this procedure potentially generates some non-realistic half-hourly values, we constrained the half-hourly values to lie between the maximum and the minimum values of the measured data set for the corresponding hour of all the days of the same month of the year.

In summary, the two-step downscaling procedure produced simulated weather data for the 1900–2100 period which presented: (1) monthly averages identical to that of the measured weather data for the period of measurement; (2) half-hourly variability identical to that of the measured data; and (3) half-hourly values constrained within observable limits.

The atmospheric CO<sub>2</sub> concentration was set to 300 p.p.m.v. in 1900 (a mean value found in the air enclosed between 1883 and 1925 in the ice core of Siple Station), to 355 p.p.m.v. in 1990 in accordance with measurements at various sites in the Northern Hemisphere (Boden et al. 1994), and to 700 p.p.m.v. in 2100 based on scenario IS92a of the Bern model (Houghton et al. 1995). The atmospheric CO<sub>2</sub> concentration was assumed to increase exponentially between these values.

#### 4. Simulation Details

Simulations for the 20<sup>th</sup> century began in 1929, the planting year of the Brasschaat Scots pines. Simulations for the 21<sup>st</sup> century began in the year 2000. Initial values for tree carbon reservoirs were set to that of one-year-old saplings. Three stand thinning scenarios were considered: (1) actual stand thinning (AST), estimated from the forest plot history; (2) a constant annual rate of stand thinning (CST), with initial and final stand densities corresponding to the AST scenario; and (3) no stand thinning (NST).

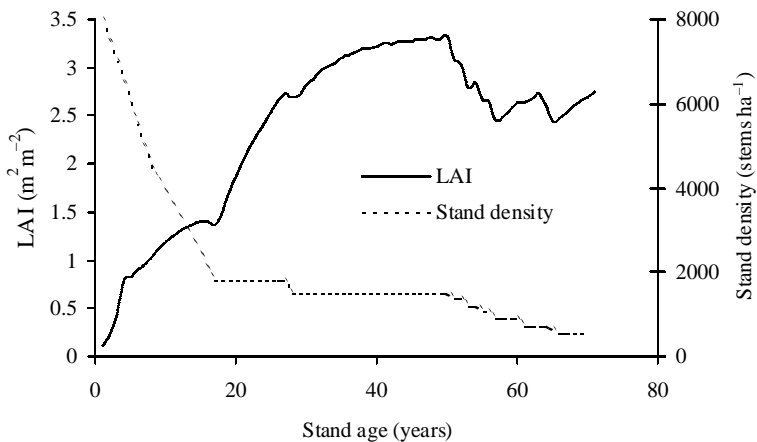
Stand thinning affects the three main CO<sub>2</sub> fluxes in forest ecosystems: (1) decreased photosynthesis (through a reduction in LAI); (2) decreased autotrophic respiration (through a

reduction in standing biomass); and (3) increased heterotrophic respiration (through enhanced litter production). To better understand how each of these three processes separately affects CO<sub>2</sub> sequestration, we ran three additional simulations based on the NST scenario combined with information from the AST scenario. Firstly, for the last three years of the NST scenario, we set the LAI to the value of the AST scenario for the corresponding years. This scenario, which we named 'low LAI', isolates the effect of stand thinning on photosynthesis. Secondly, three years prior to estimating annual NEE with the NST scenario, we set the wood biomass to that of the AST scenario. This simulation run, which we named 'low woody biomass', isolates the effects of stand thinning on autotrophic respiration. Thirdly, three years prior to estimating annual NEE with the NST scenario, we set the amount of woody litter to that of the AST scenario. This simulation run, which we named 'high woody litter', isolates the effects of stand thinning on heterotrophic respiration through the modification of woody litter accumulation.

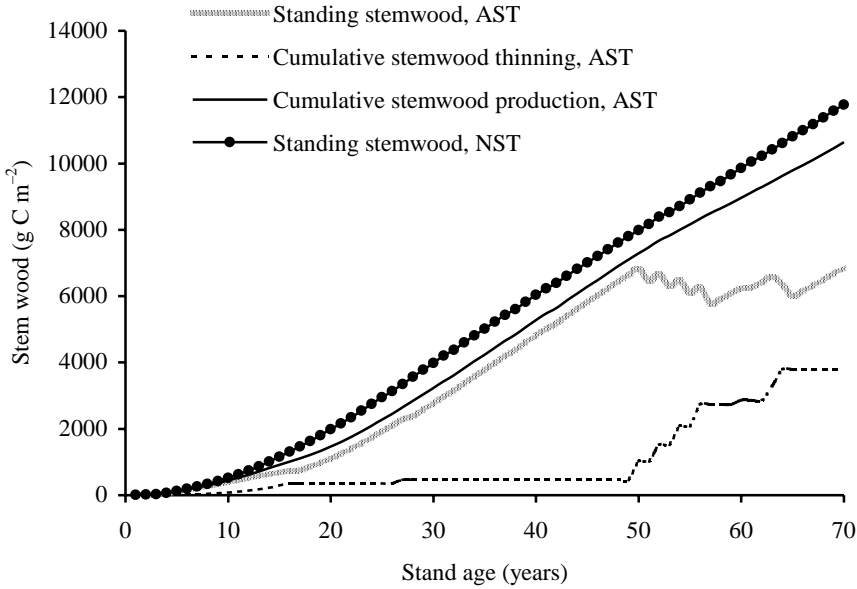
## 5. Results

### 5.1 Thinning effects under 20<sup>th</sup> century climate

Simulated LAI was substantially affected by the progressive decrease in stand density (Figure 2). This effect became more pronounced as the stand grew older. While pre-commercial thinning operations sharply reduced tree population in the first few years following planting, they had only a limited impact on canopy LAI. Although stand density was reduced from 8000 trees ha<sup>-1</sup> at planting to 550 trees ha<sup>-1</sup> at stand age 70, ASPECTS predicted that the standing stemwood biomass of 70-year-old Scots pines was 180% of the total amount of stemwood removed through successive thinning operations (Figure 3). The predicted total amount of stemwood produced during the 70-year growth of the Scots pine forest is similar under the NST and AST scenarios. Nevertheless, predicted stemwood C content of individual trees is 15 kg C without thinning vs. 125 kg C with thinning. The predicted stemwood C exported from the forest during the 70-year growth was 3800 g C m<sup>-2</sup>. Although we considered in all our simulations that stemwood was removed from the forest at stand thinning, the fate of this stemwood C is crucial for a proper estimation of C fluxes in managed forests.



**Figure 2.** Stand density and simulated maximum annual LAI for the Brasschaat Scots pines from age 1 (1929) to age 70 (1999).



**Figure 3.** Simulated cumulative stemwood production, standing biomass, and thinnings for the Brasschaat Scots pines from age 1 (1929) to age 70 (1999). Results of the actual stand thinning scenario (AST) are compared with those of the no stand thinning scenario (NST).

Simulated stand thinning (AST) decreased gross primary productivity (GPP), net primary productivity (NPP) and net ecosystem exchange (NEE), of the Scots pine forest compared to the NST scenario (Table 1). NPP and NEE were decreased by 8% and 19%, respectively, so that stand thinning had less impact on forest productivity than on the net exchange of  $\text{CO}_2$  between the forest ecosystem and the atmosphere. Reduced LAI by thinning, considered separately, decreased GPP by 16%, from 1870 to 1564  $\text{g C m}^{-2}\text{yr}^{-1}$  (Table 1). Predictably, this decrease in GPP was similar to that obtained with the AST scenario. Reduced LAI by thinning also affected NPP and NEE, which were decreased by 25% and 34%, respectively. Therefore, NEE was the most flux most susceptible to reduced LAI by thinning. Reduction of living woody biomass by thinning, considered separately, had little impact on GPP while increasing NPP by 15% due to reduced autotrophic respiration (Table 1). NEE was increased by 29%, from 299 to 385  $\text{g C m}^{-2}\text{yr}^{-1}$ . Higher woody litter biomass due to thinning, considered separately, modified neither GPP nor NPP, as the forest stand was not modified (Table 1) but resulted in a 17% decrease of NEE, due to increased heterotrophic respiration of woody litter.

In summary, thinning affected NEE in three ways: (1) it reduced LAI, which reduced photosynthesis, decreasing NEE by 100  $\text{g C m}^{-2}\text{yr}^{-1}$ ; (2) it reduced living woody biomass, which reduced autotrophic respiration, increasing NEE by about 90  $\text{g C m}^{-2}\text{yr}^{-1}$ ; and (3) it increased woody litter, which increased heterotrophic respiration, decreasing NEE by 50  $\text{g C m}^{-2}\text{yr}^{-1}$ . The combined effect of thinning, therefore, was a reduction of NEE by about 60  $\text{g C m}^{-2}\text{yr}^{-1}$  (Table 1, NST vs. AST).

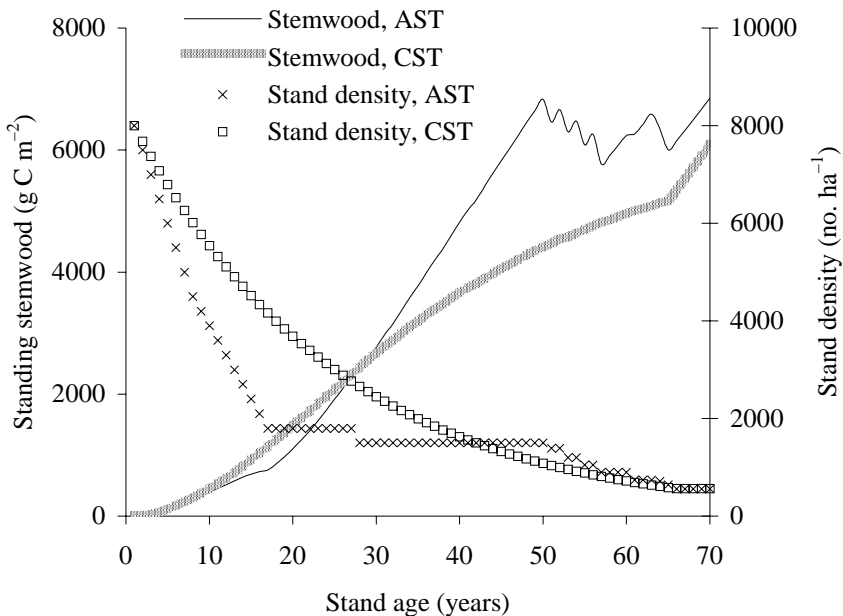
The distribution of thinning operations throughout the life span of the stand affects the simulated standing stemwood biomass. Standing stemwood C at age 70 was decreased by 11% under a constant annual 4% thinning rate (CST) as compared to the actual-thinning scenario (Figure 4). This difference is attributable to the faster recovery rate of canopy LAI



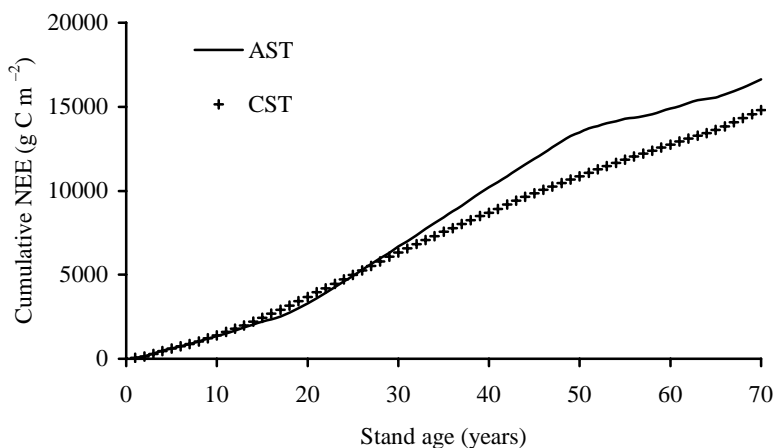
following thinning for younger stands, as in the AST scenario, where most trees are removed as saplings. Most important to C storage, the cumulative NEE was decreased by a similar 11% under the CST scenario as compared to the AST scenario (Figure 5). The heavy thinning of low-biomass saplings with AST resulted in less litter production than with CST, translating into lower levels of heterotrophic respiration with AST. These results suggest that, for an identical number of trees removed from the forest, the timing of previous thinning operations affects the total amount of C that a managed forest sequesters in any given year.

**Table 1.** The effects of 5 different stand thinning scenarios on simulated gross primary productivity (GPP), net primary productivity (NPP) and net ecosystem exchange (NEE) at the Brasschaat forest during 1997–1998. The scenarios are: (1) NST = no stand thinning; (2) AST = actual stand thinning; (3) Low LAI = NST with LAI as in AST from 1996 to 1998; (4) Low woody biomass = NST with woody tissue biomass as in AST from 1996 to 1998; and (5) high woody litter = NST with woody litter biomass as in AST from 1996 to 1998. The three last scenarios isolate the effects of the AST scenario on photosynthesis, autotrophic respiration, and heterotrophic respiration, respectively.

Scenario	GPP	NPP g C m <sup>-2</sup> yr <sup>-1</sup>	NEE
NST	1870	669	299
AST	1570	615	242
Low LAI	1564	503	199
Low woody biomass	1862	772	385
High woody litter	1870	669	249



**Figure 4.** Stand density and simulated standing stemwood biomass for the Brasschaat Scots pines from age 1 (1929) to age 70 (1999). Results of the actual stand thinning scenario (AST) are compared with those of the constant stand thinning scenario (CST).”



**Figure 5.** Simulated NEE for the Brasschaat Scots pines from age 1 (1929) to age 70 (1999). Results of the actual stand thinning scenario (AST) are compared with those of the constant stand thinning scenario (CST).

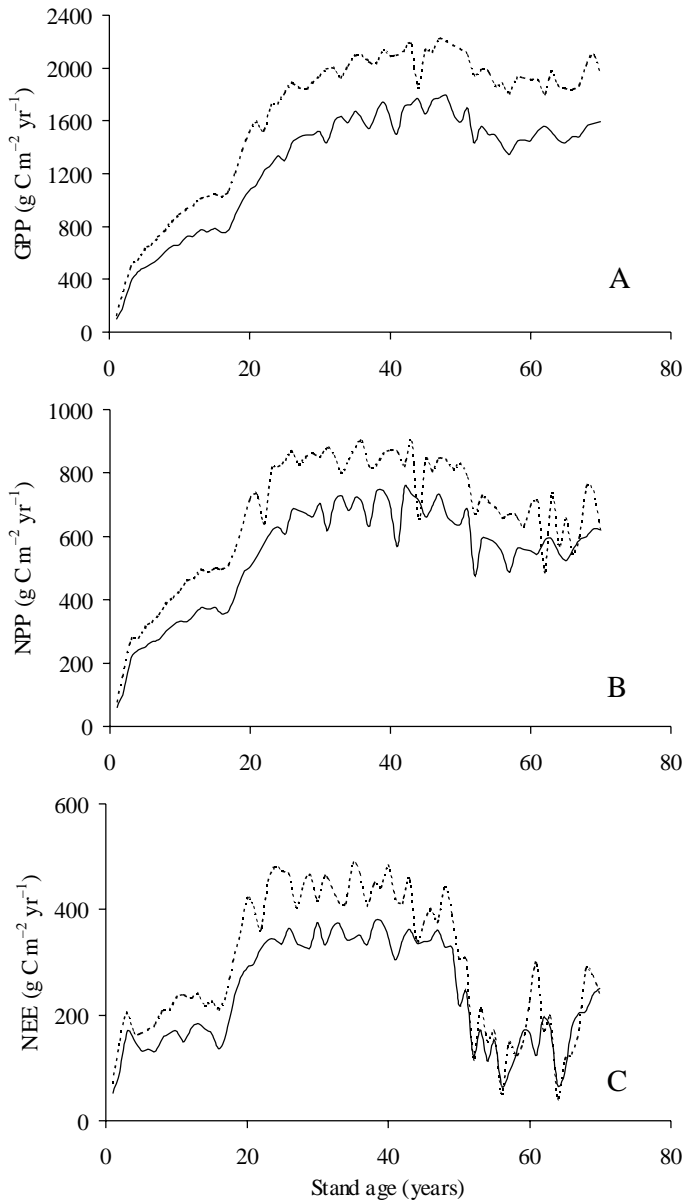
## 5.2 Impact of 21<sup>st</sup> century climate

The GPP of the Brasschaat Scots pines is predicted to increase under 21<sup>st</sup> century climate (Figure 6A). Scots pines planted in 2000 will have an estimated 28% increase in GPP as compared to those planted in 1929, when considering their growth between ages 55 and 70 yr. This effect is mostly attributable to the increase in atmospheric CO<sub>2</sub> concentration (data not shown). NPP will also increase, although the response will be less pronounced (Figure 6B). The inter-annual variability of the downscaled GCM weather simulations implies that for a given stand age the NPP may be lower for trees planted in 2000 than in 1929. The simulated NEE of Scots pines is substantially increased under 21<sup>st</sup> century climate until stand age 50 (Figure 6C). Both 20<sup>th</sup> and 21<sup>st</sup> century simulations were conducted with the AST scenario. After age 50, the severe thinning of the Scots pine stand resulted in a sharp decrease in both 20<sup>th</sup> and 21<sup>st</sup> century NEE curves. In addition, NEE during the last 20 years of Scot pine growth will be similar in the 20<sup>th</sup> and 21<sup>st</sup> centuries, provided that similar thinning operations are conducted.

The NEE simulations with the CST scenario differed substantially from those conducted with the AST scenario for both the 20<sup>th</sup> and 21<sup>st</sup> centuries (Figure 6C and Figure 7). On a multi-year average, the NEE remained higher for 21<sup>st</sup> than for 20<sup>th</sup> century forests. Both scenarios show a peak of NEE at about stand age 25, when the LAI approaches a plateau, while respiring living biomass is not too large.

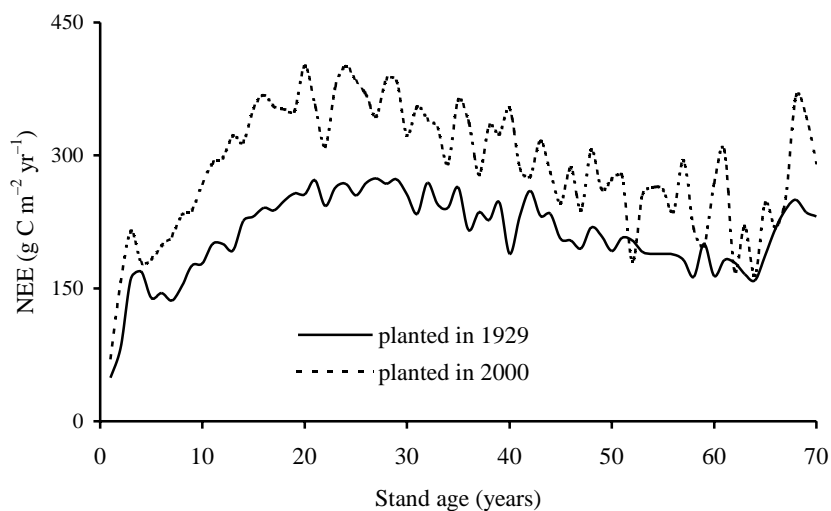
## 6. Discussion

Our ASPECTS simulations with downscaled GCM weather data suggest that GPP, NPP and NEE are substantially modified by stand thinning (Table 1). This result suggests that the time elapsed since the last thinning operation could have a critical effect on measured NEE. Several studies have linked inter-annual variations in NEE to climate variability (Goulden et al. 1996; Goldstein et al. 2000; Wilson and Baldocchi 2000). Nevertheless, little or no attention has been given until now to stand thinning as a factor modifying annual NEE values



**Figure 6.** Simulated GPP (A), NPP (B), and NEE (C) of Scots pines planted in 1929 (solid line) and 2000 (dotted line), and subjected to thinning operations identical to those conducted at Brasschaat from 1929 to 1999.

(Rasse et al. 2001a). Our simulations suggest that stand thinning can generate fluctuations in NEE of greater magnitude than that due to inter-annual climate variability (Figure 6C). The NEE of a 68-year-old Scots pine forest, as managed in Brasschaat, would be similar in 2068 to that measured in 1997 due to the severe thinning operations conducted in the last 13 years prior to measurement (Fig 6C). Hence, ASPECTS predicts that although NPP will be consistently greater during the 21<sup>st</sup> than the 20<sup>th</sup> century, the NEE will be more variable because: (1) enhanced GPP under 21<sup>st</sup> century climate will produce greater standing biomass,



**Figure 7.** Simulated annual NEE of Scots pines planted in 1929 and 2000, and subjected to a constant annual rate of stand thinning (CST).

so that thinning will generate greater amounts of litter, resulting in larger fluctuations in NEE; and (2) the inter-annual variability of the GCM-simulated climate increases towards the end of the 21<sup>st</sup> century. These results have important consequences for the role of forests in atmospheric CO<sub>2</sub> sequestration: the thinning history of the stand is an important consideration when it comes to accurately estimating C fluxes in managed forests.

NPP is modified by stand thinning, although to a lesser extent than NEE (Table 1, Figure 6B and 6C). The simulated reduction in NPP results from two opposite effects: (1) a decrease in photosynthesis due to the reduction in LAI; and (2) a decrease in autotrophic respiration due to the reduction in tree biomass (Table 1). The first effect being greater than the second, ASPECTS predicts an overall reduction in NPP following stand thinning. Thinning-induced reductions in forest productivity have been reported in several studies (Velazquez et al. 1992; Egnell and Leijon 1997; West 1998). Our simulations suggest that early pre-commercial thinning operations decrease stemwood productivity to a lesser extent than when conducted later as the stand matures (Figure 4). These results translate directly into increased carbon sequestration by forests that are heavily thinned at an early stage as compared to those that are thinned later in their life cycle (Figure 5).

In conclusion, our simulations indicate that stand thinning needs to be taken into account both for accurately simulating current NEE measurements and for predicting future NEE and NPP under 21<sup>st</sup> century climate. Although the effects of stand thinning on tree growth have been investigated for decades by foresters, there is a renewed need to examine thinning effects on carbon fluxes in temperate forests.

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# The 3-PG Forest Model: Matters Arising from Evaluation against Plantation Data from Different Countries

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## Abstract

The simple process-based forest production model 3-PG, developed by Landsberg and Waring (1997), requires few parameter values and only readily available data as inputs. It works on monthly time steps and the output variables it produces are those of interest to forest managers. 3-PG has been evaluated using data from experiments and commercial plantations in Australia and New Zealand, the UK, the United States, South Africa, Sweden and Finland. The model generates stem mass and volume, leaf area index, stem diameter and root mass. It can be fitted to data sets that include some or all of these variables. When time series data are not available, 3-PG can be adjusted to produce the values of variables measured only once. Calibration of the model against a wide range of data has provided parameter values and information about model performance in relation to a number of species at a range of sites. The central questions pertinent to the performance of the model are: (1) the importance of an appropriate estimate of soil fertility, which affects carbon allocation and the canopy quantum efficiency used in the model; (2) the need for good estimates of soil water available in the root zone; (3) the importance of Specific Leaf Area. Differences in the performance of species are largely determined by responses to temperature. There is no strong evidence for species differences in carbon allocation patterns but the significant differences that exist in litterfall rates affect the development of leaf area index and hence radiation interception and biomass production. Differences between species in their responses to fertility and to drought are probably important but there is little empirical information available to evaluate these. Using the satellite-driven version of the model to simulate forest growth and productivity over large areas has demonstrated very clearly the great variability of forest plot data, and the need for methods of scaling up to appropriate average values. 3-PG provides an excellent framework for eco-physiological research on forest growth processes at levels below (more detailed) those at which this model is written.

*Keywords: process-based, calibration, biomass, production, weather, site factors*

## Introduction

The model called 3-PG (**Physiological Principles Predicting Growth**), developed by Landsberg and Waring (1997), is a simple process-based model requiring few parameter values and only readily available data as inputs. It is a generalised stand model (i.e. it is not site or species-specific, but needs to be parameterised for individual species) applicable to plantations or even-aged, relatively homogeneous forests, which was developed in a deliberate attempt to bridge the gap between conventional, mensuration-based growth and yield models, and process-based carbon balance models. The output variables it produces are those of interest to forest managers.

The model uses the principles that underlie earlier models such as FOREST-BGC (Running and Coughlan 1988) and BIOMASS (McMurtrie et al. 1990). Like these it is based on the calculation of radiation interception, canopy photosynthesis or Gross Primary Production (GPP), estimation of net primary production (NPP) and the allocation of the resultant carbohydrates to component parts of the trees. One of the major differences lies in the fact that 3-PG uses a time step of a month, so that the descriptions of the biophysical processes that contribute to and control forest growth are necessarily greatly simplified – the model attempts to capture the essential features of these processes, consistent with more detailed descriptions and knowledge about them. The other major difference lies in the carbohydrate partitioning procedure, outlined below.

Since 3-PG is a conservation-of-mass model it provides a useful tool for carbon sequestration calculations, including the amount of carbon that goes below ground. The model can be precisely calibrated to above-ground stand growth, and various sub-models can be independently tested. We argue that, since the simulation of above-ground growth – and hence carbon sequestration – can be demonstrably accurate, the estimates provided of below-ground sequestration are likely to be realistic, and certainly provide clearly-stated and quantitative hypotheses about this process. Work with the model supports the contention that it can provide estimates of site productivity (site index) from soil and climate data without the need for measurements of tree growth on the sites. It is also a useful analytical tool that can be used for exploring the consequences of drought, changes in management practices such as thinning and the influence of varying soil fertility status. However, because of limitations in our knowledge of the interactions between soil chemical characteristics and plant growth, the model cannot be used to indicate the fertiliser that might be needed to achieve particular growth rates and productivity levels. Also, because it uses a Fertility Rating rather than a model of nitrogen mineralisation and uptake, 3-PG is as yet unable to provide assessments of long-term impacts of management on nutrient availability and sustainable productivity (see Cannell and Thornley and McMurtrie et al. 2001).

In this paper we provide a brief outline of the structure of 3-PG, which has been described in detail elsewhere, some examples of calibration and sensitivity analyses, and demonstrations of the performance of the model as a predictive tool at stand and regional scales. We then consider those aspects of the model where the uncertainties are greatest and where research should, perhaps, be focussed, and discuss the questions of species differences and the possible value of this model in relation to carbon sequestration calculations.

## Model Structure

Essentially, 3-PG consists of two sets of calculations: those that lead to biomass values, and those that distribute biomass between various parts of the trees, and hence determine the growth pattern of the stand.



## Inputs

The input data required by 3-PG are weather data, soil water holding capacity in the root zone ( $\theta$ , mm depth equivalent), initial stem number ( $n_{st}$ ), initial total stem ( $w_s$ ), foliage ( $w_f$ ) and root ( $w_r$ ) mass ( $Mg\ ha^{-1}$ ), and an (index) value for soil nutrient status (the fertility rating, FR, which takes values between 0 and 1). Parameter values needed are the constants ( $a_i$ ) and coefficients ( $n_i$ ) of the allometric equations for leaf and stem masses in terms of diameter at 'breast' height ( $B$ ) ( $w_i = a_i B^{n_i}$ ), specific leaf area (SLA,  $\sigma_f$ ), cardinal temperatures (see below), litterfall rate, maximum stomatal conductance and, the most important, canopy quantum efficiency. Monthly average weather data are usually used, but it is possible to provide monthly data for each year of growth.

## Biomass production

Biomass production is determined by radiation interception and carbon fixation by the canopy. Photosynthetically active radiation (PAR,  $\phi_p$ ) is calculated from global solar radiation. Absorbed photosynthetically active radiation (APAR) is calculated using Beers Law and (projected) stand leaf area index ( $L^*$ ). Gross Primary Productivity (GPP,  $P_G$ ) is obtained by applying a canopy quantum efficiency (QE,  $\alpha_c$ ) value to APAR. The actual value of QE at any time is estimated by correcting the potential value for the effects of sub-optimal nutrition (described in terms of a linear correction by FR), temperature, soil drought and atmospheric vapour pressure deficit (which reduces stomatal conductance), and stand age. The model also incorporates the effects of nutrition as part of the carbon allocation procedure, which is based on allometric equations and includes (non-empirical) allocation to foliage, so  $L^*$  varies according to growing conditions.

Net Primary Productivity (NPP,  $P_N$ ) is calculated from a simple ratio of NPP to GPP (see Waring et al. 1998; Malhi et al. 1999). The assumption of constancy in the ratio of NPP to GPP is somewhat controversial. However, Dewar et al. (1998) presented a mechanistic model which predicted that the ratio should be largely independent of environmental factors over time scales of days to weeks, and a detailed analysis by Mäkelä and Valentine (2001) led them to accept that, although they produced indirect evidence the NPP/GPP ratio declines over the course of development of an even-aged forest stand, there is also increasing evidence that there is little variation in the ratio among many even-aged stands. Mäkelä and Valentine said that:

*despite our scepticism (about the constancy of the ratio)... it would appear that the assumption of a uniform value can be put to some good use... and may serve to get physiologically-based models into broader use, particularly in simple management applications.*

## Carbohydrate allocation

Allocation of carbohydrate to roots is determined by growing conditions; the proportion of NPP allocated to roots increases if nutritional status and/or water relations are poor; allocation to stems and foliage is on a single-tree basis, based on the ratio of the derivatives of the allometric equations describing leaf and stem mass in terms of  $B$ . The procedure is dynamic and self-regulating – it is not the same as calculating the masses of particular components, such as foliage or stem mass, from allometric equations and using the ratio of

the values to determine allocation. At the end of each time step the allometric equations are inverted and stem diameters are calculated. Stand basal area and volume are obtained from stem diameters and stem numbers. Initial tree populations are specified and changes in stem populations calculated using the well-established  $-3/2$  power law or specified thinning.

The time course of leaf mass, and hence  $L^*$ , is strongly influenced by litterfall; high litterfall rates can result in a gradual decline in canopy leaf mass if the rate of carbohydrate allocation to foliage is not high enough to maintain  $L^*$ . It is therefore important to use the best available information about litterfall for the species of concern. We know that litterfall rates ( $\gamma_f$ ) are strongly influenced by growing conditions and season: for example Linder et al. (1987) found 8-fold variations, between months, in needle litterfall from *Pinus radiata*, with drought stimulating massive needle fall, particularly from treatments that had been heavily fertilised and developed large canopies; in loblolly pine, new foliage grows during the spring/early summer period (March-June), and the foliage of the previous season starts to fall towards the end of the summer, i.e. the average life of a foliage cohort is about 15 months (Vose and Allen 1991; Hennessey et al. 1992). Leaf area depends on leaf mass and specific leaf area (SLA,  $\text{m}^2 \text{kg}^{-1}$ ). SLA may vary with growing conditions (Specht 1989), with tree age (Sands and Landsberg 2001) and position of the leaves in canopies. Default values of 3.5–4 for broad-leaved species, and 6 for conifers are usually used in 3-PG. The influence of these factors is explored in the next section.

## Water balance

Water balance is calculated monthly, using the well-established, and biophysically rigorous, Penman-Monteith equation. Boundary layer conductance, which is a large value in forests, is taken as a constant. Stomatal – and hence, through  $L^*$  – canopy conductance depends on a maximum value which is modified by atmospheric vapour pressure deficit. Soil water storage in the root zone may be an important factor in the water balance, and in determining site productivity, particularly in areas subject to long dry periods (see later comment). Law et al. (1999) evaluated the water balance sub-model in 3-PG and found that the model accurately simulated soil water balance under *Pinus ponderosa*, over two years. The water balance calculations have also been tested at other sites, among which results from plantations on deep soil in Brazil, where neutron probe measurements were made for a number of years, were excellent.<sup>1</sup>

Software for 3-PG, with supporting documentation (Sands 2000) is available free on the internet (see Acknowledgements).

## Calibration and Sensitivity Analysis

### Calibration

If 3-PG is to be used to simulate the behaviour and responses of a particular species it should, ideally, be calibrated against time series observations of stem mass, stem diameter and leaf area index, with known stem populations. However, such data are seldom available in forestry, and the model can be calibrated against stand volume alone, being constrained to

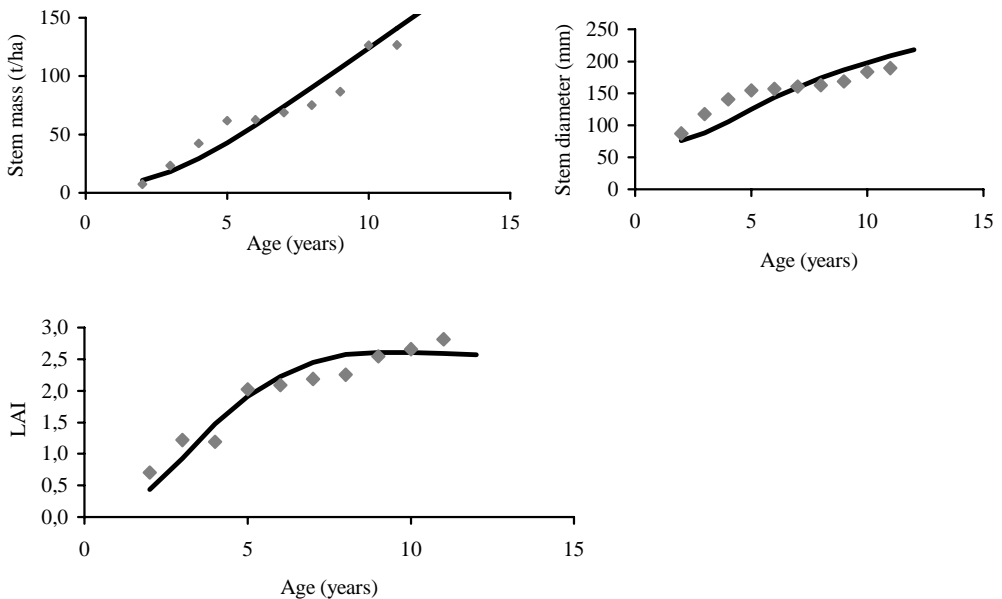
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<sup>1</sup> These data are commercial-in-confidence and not available for publication.

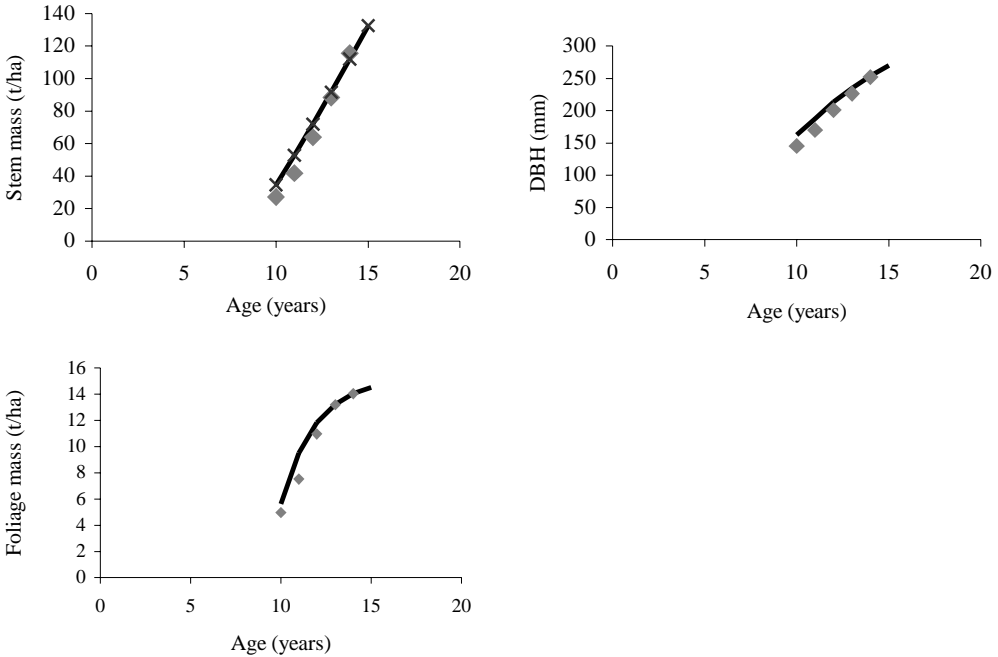
produce ‘reasonable’ values, over time, of stand  $L^*$  and stem diameters. Values have to be selected for the stem allometric parameters, the cardinal temperatures that control growth, litterfall and SLA, soil water holding capacity of the root zone and the fertility rating (FR). The model also requires estimates of the fraction of above-ground growth that consists of branches. Default or tabulated values are available for all of these; generally only the FR is allowed to vary during calibration, on the grounds that the relationship between soil chemical properties and the effects of fertility on tree growth is not well understood. Landsberg et al. (2001) provide a detailed description of calibration procedures and they are also detailed in the software package.

We present here two sets of data against which 3-PG has been calibrated, illustrating results of the type commonly obtained where appropriate time series data are available.

Figures 1 and 2 show the curves produced by 3-PG after optimisation of the parameter values against the data (see Table 1). This is done by hand fitting using an iterative procedure to optimize parameter values. The procedure involves choosing an initial set of parameter values and appropriate inputs, running the model with weather data for the sites, and information about the soils, and comparing output with observed values. Parameter values are then adjusted to improve the fit and the model re-run. Successive adjustments lead, in most cases to good fits between observed and simulated variable values. A number of parameter values can be varied to alter the output of the model, but normal procedure is to select the best values for the species in question on the basis of information available, and to use standard default values for as many as possible of the other parameters; if too much flexibility is allowed it becomes very difficult to identify the factors that are determining growth and yield differences, little is learned and we have little guidance about parameter values to use in the model when predictive calculations are



**Figure 1.** Calibration: time series showing the course of stem mass, stem diameter and leaf area index ( $L^*$ ), from one treatment in a *Eucalyptus grandis* spacing trial in Swaziland. Points are measurements and the lines are obtained in the course of calibrating 3-PG against the data. The aim is to optimise the fit to all variables simultaneously.



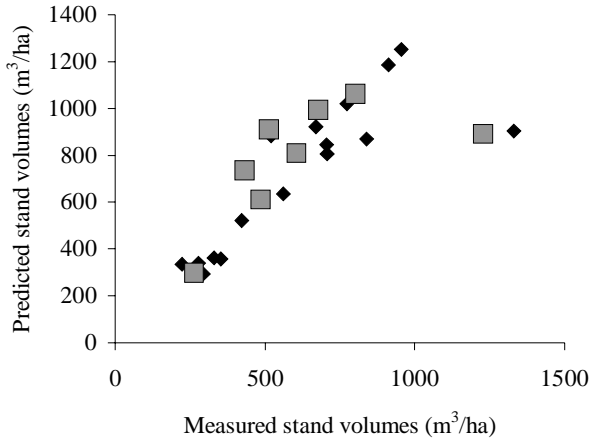
**Figure 2.** Calibration: time series showing the course of stem mass, stem diameter and foliage mass from the Irrigated x Liquid Fertilizer treatment of an ecophysiological experiment on *Pinus radiata* in the ACT, Australia. The points are measurements and the lines are obtained in the course of calibrating 3-PG against the data. The aim is to optimise the fit to all variables simultaneously (data from Snowdon and Benson 1992).

to be made. In most cases the values for the parameters of the stem mass/stem diameter allometric equation are fixed on the basis of empirical data, but because of the non-linear nature of the equations determining carbon allocation, and the fact that they are integrated over whole rotations, it is sometimes necessary to adjust these values to get best fits with time series data consisting of several variables, such as those in Figs 1 and 2.<sup>2</sup>

Tickle et al. (2001) tested 3-PG against conventional forest models applied to plots in 50 000 ha of natural forest, dominated by *E. delegatensis* in New South Wales (see Figure 3). The data available, in this case, were standard stem diameter measurements in 22 plots, ranging in age from 12 to 80 years. Stand volumes were estimated using established relationships with height (site index) and stem number. Good quality information on soil fertility and water holding capacity, and weather data, were acquired as part of the study. There was no information about foliage mass or  $L^*$ . All parameter values were set to the best known (or estimated) default values for the species, except for the values of the foliage mass/stem diameter equation, which were allowed to vary so that the model produced values, and time course, of  $L^*$  that corresponded to estimates and expert knowledge.

Parameter values were optimised (by calibration) for each of eight plots and the (single) set of values that gave the best results for those plots (Table 1) was then applied to all plots, giving the results shown in Figure 3. Note that the relationship between predicted and measured stand biomass, while linear, with a high correlation coefficient ( $r^2 = 0.91$ ), is somewhat biased. Departure from a 1:1 relationship arises because the best overall set of

<sup>2</sup> Work is in progress to automate calibration of 3-PG using a non-linear curve-fitting package (PEST; see <http://www.flowpath.com>)



**Figure 3.** Stand volumes in a natural, *Eucalyptus delegatensis*-dominated forest in New South Wales, predicted using 3-PG, plotted against measured volumes. The squares represent plots against which the model was calibrated using a single set of parameter values that gave the best values for all calibration plots. Diamonds are plots for which there was no calibration. Plots ranged in age from 16 to 80 years.

**Table 1.** Parameter/variable values used in 3-PG to calculate the curves in Figure 1 (*Pinus radiata* in the Australian Capital Territory), Figure 2 (*Eucalyptus grandis* in a spacing trial in Swaziland) and Figure 3 (*E. delegatensis* in a natural forest in New South Wales).

Parameter/variable	<i>P. radiata</i> (ACT, Aus.)	<i>E. grandis</i> (Swaziland)	<i>E. delegatensis</i> (Bago-Maragle, NSW)
Avail. soil water (mm)	170	80	330
Fertility rating	0.9	0.2	0.4
Stem no.	746	1019	550
Max. litterfall rate/month	0.03	0.025	0.02
$T_{\min}$ (°C)	0	2	-2
$T_{\text{opt}}$ (°C)	20	20	15
$T_{\max}$ (°C)	32	32	25
$a_F$	0.016	0.012	0.012
$n_F$	2.55	2.4	2.26
$a_S$	0.005	0.021	0.003
$n_S$	3.2	2.85	2.65

parameter values is not necessarily the best for any individual plot; inevitably there is plot to plot variation. If the concern is to produce a highly accurate prediction tool for the study area the results can of course be corrected empirically, although this process provides no useful insights into the reasons for the variation. On the basis of the (standard mensuration-type) data available for these plots it was difficult to make progress beyond the point demonstrated by the data in Figure 3. Tickle et al. (2001) could find no explanation for the two outlying points. An alternative approach to analysing these data was adopted by Landsberg et al. (2001), who fitted 3-PG to all the plot data together, treating them as a single time sequence. This produced a slightly different set of parameter values from those given in Table 1 and the results were not as good in terms of the relationship between observed and predicted values.

Tickle et al. (2001) found that using GIS with layers for soil fertility and water holding capacity, topography and climate, 3-PG was a better tool for extrapolation across the whole 50 000 ha forest than conventional growth and yield models. They also used 3-PG to estimate site index, using the results to improve predictions with conventional models.

### Sensitivity analysis

The results of some sensitivity analyses are presented in Table 2. They were produced using parameter values and weather data for *E. globulus* plantations growing in Tasmania and illustrate the effects of SLA and  $\gamma_f$  on maximum  $L^*$ , litterfall mass in the year of maximum  $L^*$ , and total biomass production over a rotation. Increasing  $\gamma_f$  from 0.03 to 0.05 per month (equivalent to 36% and 60% leaf loss per year) decreased maximum  $L^*$  from 6.4 to 4.6, reduced the age at which maximum  $L^*$  was reached, and increased the total amount of litterfall, but only reduced total biomass production by 4 tons  $\text{ha}^{-1}$ . This was largely because, in this case, maximum  $L^*$  significantly exceeded 3: with an extinction coefficient of 0.5, Beers Law predicts that 95% of short-wave radiation is intercepted by  $L^*=3$ , so higher values make little difference.

**Table 2.** Sensitivity analysis illustrating the effects of varying litterfall rates on maximum  $L^*$ , the age when maximum  $L^*$  is reached, annual litterfall and total biomass production by a stand of *Eucalyptus globulus*. The analysis was run using optimised SLA values and weather conditions typical of southern Tasmania.

Litterfall Rate (fraction foliage mass $\text{month}^{-1}$ )	Maximum $L^*$	Age of maximum $L^*$ (yr)	Annual litterfall (tons $\text{ha}^{-1}$ )	Total biomass production at 15 years (tons $\text{ha}^{-1}$ )
0.03	6.4	5	3.7	76.6
0.04	5.3	4	4.5	74.3
0.05	4.6	3	5.0	72.0

Table 3 illustrates the influence of stomatal conductance ( $g_s$ ) on annual NPP and water transpired. In this case, since the influence of  $L^*$  again becomes negligible when  $L^*>3$  (Kelliher et al. 1993, 1995), the effect is primarily through canopy conductance, which is determined by  $g_{s,\text{max}}$ . The effects on water use and NPP are in opposite directions, because higher conductance and greater water use lead to longer periods of water stress, and hence reduced NPP. The actual magnitude of the results is of only passing interest; it would vary depending on the conditions pertaining for the analysis.

**Table 3.** Sensitivity analysis illustrating the effects of varying maximum stomatal conductance ( $g_{s,\text{max}}$ ) on annual transpiration and NPP.

$g_{s,\text{max}}$ ( $\text{m s}^{-1}$ )	Annual $E_t$ (mm)	Annual NPP (tons $\text{ha}^{-1}$ )
0.015	1149	45.2
0.020	1213	38.1
0.025	1236	31.7

We have not provided illustrative data, but note here that stand productivity – in practice and as predicted by 3-PG – is not sensitive to available soil water when rainfall is adequate; i.e. when there is seldom a large difference between rainfall and (potential) transpiration. However, if an area is subject to extended dry periods – for example the Pacific North West region of the USA, with mild wet winters and hot dry summers, or the mediterranean rainfall regions, of which the south-west of Western Australia is an example – then large soil water storage capacity can make an enormous difference to the survival and growth of species (see Coops and Waring 2001).

The software package includes a facility for sensitivity analysis of the effects of any parameter in terms of any specified output(s).

### 3-PG as a Predictive Tool

If a model is to be used as a predictive tool it must be driven by simple, readily available input data and must provide reliable estimates of growth and yield using, as far as possible, generic (species independent) parameter values. 3-PG fulfills these requirements: it is driven by monthly average solar radiation values and the environmental modifiers are derived from monthly average rainfall, temperature and humidity data. The default parameter values are surprisingly stable across species. Landsberg et al. (2001) have recently evaluated the performance of 3-PG in relation to about 50 data sets from plantations and commercial forests around the world. They concluded that:

*there are no consistent differences in allometric parameter values for different species derived from fitting 3-PG to experimental or forest plot observation data. This does not necessarily mean that species differences do not exist but rather that, if they do, they are overwhelmed by differences caused by climate, soil conditions and cultural or management practices, such as initial stem populations and factors such as weed control in the early growth stages. The lack of clear differences in carbon allocation patterns between species is consistent with the analysis of Ter-Mikaelian and Korzukhin (1997), who established the allometric equations for 63 north American tree species. Cannell (1989) analysed dry matter partitioning between stem wood and foliage for a number of coniferous species in Japan, England, the USA, Australia and New Zealand and concluded that after canopy closure the proportion allocated to wood is more or less constant.*

With the exception of rainfall, values for the weather variables required by 3-PG can all be estimated with acceptable accuracy using methods outlined by (for example) Thornton and Running (1999) and Coops et al. (2000).

The other requirement for the use of a model as a predictive tool is that it must be able to simulate accurately the growth patterns of stands for which it has been calibrated. This has been demonstrated in a number of cases (see Landsberg et al. 2001). This does not prove that the model can be safely extrapolated, but it gives confidence that all the essential features of the system have been accounted for. Testing ‘blind’, i.e. using the model with parameter values obtained by calibration against a particular species to predict the performance of that species at other locations, without calibration, is difficult because inadequate or inappropriate specification of site variables may lead to lack of coincidence between model output and observation, which may be erroneously attributed to model error. However, Sands and Landsberg (2001) have tested 3-PG thoroughly in this way, using data for *E. globulus* from Western Australia and Tasmania. They concluded that:

*3-PG ...provides a useful tool for modelling the time-course of stand development of intensively managed, even-aged E. globulus, given observed initial biomass data and*

*stocking. Further, it can be used as a predictive tool for modelling long-term stand development when initialised with typical seedling stock at age 0, even though early stand development is quite sensitive to assumed initial biomass data.*

The results shown in Figure 3 indicate that a similar conclusion can be drawn for natural forests, even if initial stand conditions are not well known. Waring (2000) successfully simulated the growth of Sitka spruce at a number of sites in Britain, while Coops et al. (1998a,b, 2001) have demonstrated for a range of forest types that the simplified version of 3-PG, driven by satellite observations, can provide accurate predictions of growth, in terms of NPP, across large areas.

To use 3-PG as a predictive tool, appropriate values of the cardinal temperatures that control growth, litterfall and SLA, appropriate for the species under consideration, have to be selected, as well as values for the stem allometric parameters. Soil water holding capacity of the root zone and the fertility rating (FR) will be estimated from local knowledge and information available about soils and stem populations specified. It is important to ensure that the time course of  $L^*$ , and the maximum values reached, conform to values that can be regarded as 'reasonable' in relation to the water balance of the region. These can be estimated from native vegetation: leaf area will tend to values limited by available water (Specht 1972; Grier and Running 1977); i.e. evergreen vegetation will tend to adjust its  $L^*$  in response to long-term water availability. Plantations may reach higher  $L^*$  than native vegetation because of cultural factors (fertilisation, reduction of competition from other vegetation), and young plantations may violate the condition of hydrologic equilibrium if they either do not occupy a site in a hydrological sense, or they initially mine soil water stored in the profile prior to their establishment (Hatton et al. 1998). Adjustment of the time course of  $L^*$  is simply done by adjusting the foliage allometric parameters (see 'Instruction' sheet in the software package). The model can then be used to estimate probable growth rates under average weather conditions, wood production and carbon sequestration by stands in the area. The influence of abnormal conditions such as drought can be assessed by providing monthly weather data for each year of growth, which allows assessment of the affect of droughts that occurred, or droughts simulated by reducing rainfall by chosen amounts for any period of interest.

Because canopy quantum efficiency is linearly linked to fertility (within constraints), changes in FR will lead to significant changes in biomass production by 3-PG. Despite more than a century of research, our knowledge at the process level of the way soil chemistry affects tree growth is poor; the use of a fertility index in the model is an explicit recognition of this, and indicates that research may need to be focussed in this area, with scientists perhaps using approaches different to those used in the past. One possibility is the use of 3-PG as a heuristic tool – the model could be calibrated against stands on soils of different fertility, for which there is a great deal of information available, and the relationships between soil chemistry and the FR values necessary for accurate description of stand growth, analysed. Meanwhile, varying FR will give indications of probable responses to varying fertility, but will not provide quantitative guidance about the fertilisation needs of plantations.

## Discussion

Enough evidence has been accrued to argue that 3-PG is a robust and reliable model at stand level. It can be used to assess the productivity, on sites where they have not been grown, of



species adapted to the region under consideration; the greatest uncertainty, in most cases, is likely to be the value of FR and hence the value of canopy quantum efficiency to use. Fertility Ratings can be estimated from chemical analysis and the knowledge of people familiar with the soils of the region, but we need continued research focussed on this aspect of process-based modelling. Soil moisture holding capacity may be important in some cases but can generally be estimated to acceptable levels of accuracy by a combination of soil sampling and examination, combined with the use of pedo-transfer functions to estimate water-holding capacity. 3-PG is not sensitive to soil water storage in areas where serious seasonal droughts are infrequent.

We noted earlier that, because 3-PG is a conservation-of-mass model, it can be used to make estimates of below-ground carbon sequestration. The code includes a root turnover parameter, although this makes no difference to the performance of the model – there are no feedbacks. If this term is set to zero the accumulated root mass provides some insight into the amount of biomass that is respired below-ground; clearly this is a very difficult value to test, and the actual values depend on other assumptions in the model – such as the question of the NPP/GPP ratio. Nevertheless, the model used in this way provides a clearly-stated, quantitative hypotheses that can, in principle at least, be tested.

From the management point of view 3-PG, calibrated for the species and region, provides a useful tool for investigating the probable influence of drought, the effects of thinning and questions such as the effects of plantations on local hydrology. From the ecological point of view the model can be used to evaluate the factors limiting tree growth at different times in different locations (see Coops and Waring 2001). Investigation of the importance, and influence, of SLA across ecological gradients may be rewarding.

3-PG provides a useful framework for research. It is a ‘top-down’ model, which has the disadvantage that the environmental modifiers are defined at spatial and temporal scales that differ greatly from those at which environmental variables influence NPP (foliage, seconds, hours...). It is therefore useful to explore the relationships between detailed, short-term, ‘bottom up’ models, supported by detailed measurements, and the performance of robust, simplified models, such as 3-PG, designed for operational use. Such investigations may include detailed modelling of radiation interception by canopies, canopy photosynthesis calculations using a model such as the Farquhar et al. (1980) model and, possibly, detailed modelling and measurements of respiration. A good example of this type of study is provided by Raulier et al. (2000). A model of this type can also perform an important function in providing a means of assessing the importance and limitations of particular processes in relation to growth and the performance of a system (stand) as a whole.

## Acknowledgements

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# Sub-Regional Climate Change Impacts on the Water Balance, Carbon Balance and Primary Productivity of Maritime Pine in South-West France

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## Abstract

The development of sustainable forest management practices must take into account the possible impacts of climate change at the spatial scales of interest to forest managers. We used a simple process-based model of pine forest growth (GRAECO) to examine sub-regional climate change impacts on the water balance, carbon balance and primary productivity of maritime pine in south-west France. GRAECO simulates the energy, water and carbon balances and growth of a monospecific, even-aged stand of maritime pine, represented by soil, understorey and tree layers. Standard and modified ( $2 \times \text{CO}_2$ ) climate data were generated by the ARPEGE model of METEO-France over a 10-year period across a 60 x 60 km grid in south-west France. GRAECO simulated the behaviour of two stand age classes (8 and 18 yr) at seven representative grid points chosen to cover the range of climate conditions within this region. Under  $2 \times \text{CO}_2$ , ARPEGE predicted a mean temperature increase of 2°C, an increase in summer values of air vapour pressure deficit, and a shift in the seasonal distribution of precipitation from summer and autumn to winter. Regardless of age class, GRAECO then predicted a decrease in gross and net primary productivity at 6 grid points, but a slight increase in growth at the most coastal (and humid) site. The negative growth responses reflected increases in air and soil moisture deficits. Increased water stress more than offset the relatively small  $\text{CO}_2$ -fertilisation effect, which was limited by the poor nutrient status of the sites considered. We conclude that sub-regional ( $\approx 10$  km) variations in the impact of climate change on forest growth may be significant, and should be considered in assessments of sustainable forest management.

*Keywords: climate change, forest water balance, carbon balance, pine forest, Pinus pinaster, process-model, sub-regional impact study, south-west France*

## Introduction

Predicting climate change impacts on terrestrial ecosystems remains an important challenge from political, management and scientific perspectives. In particular, an understanding of climate change impacts on the carbon sink strength of forests is crucial for predicting the future course of atmospheric CO<sub>2</sub> concentration. Climate change itself has been studied mainly from a regional to global perspective. However, assessing climate change impacts on forest growth may also require a more local perspective. For example, the development of sustainable forest management practices must take into account the possible impacts of climate change at the spatial scales of interest to forest managers (i.e. sub-regional).

Studies of regional-scale climate change impacts may not be sufficient for this purpose. The European LTEEF-2 project (Long-Term Effects of Climate Change and Atmospheric CO<sub>2</sub> Increase on European Forests) provided a regional overview of expected growth responses for different species and site conditions, predicting, for example, that forest productivity would increase in boreal, northern Atlantic and northern continental regions, and decrease in Mediterranean regions. However, local climate and growth variations may be of the same order of magnitude as the changes predicted over the next 50 years. The question of local variations in climate and site conditions remains unexplored.

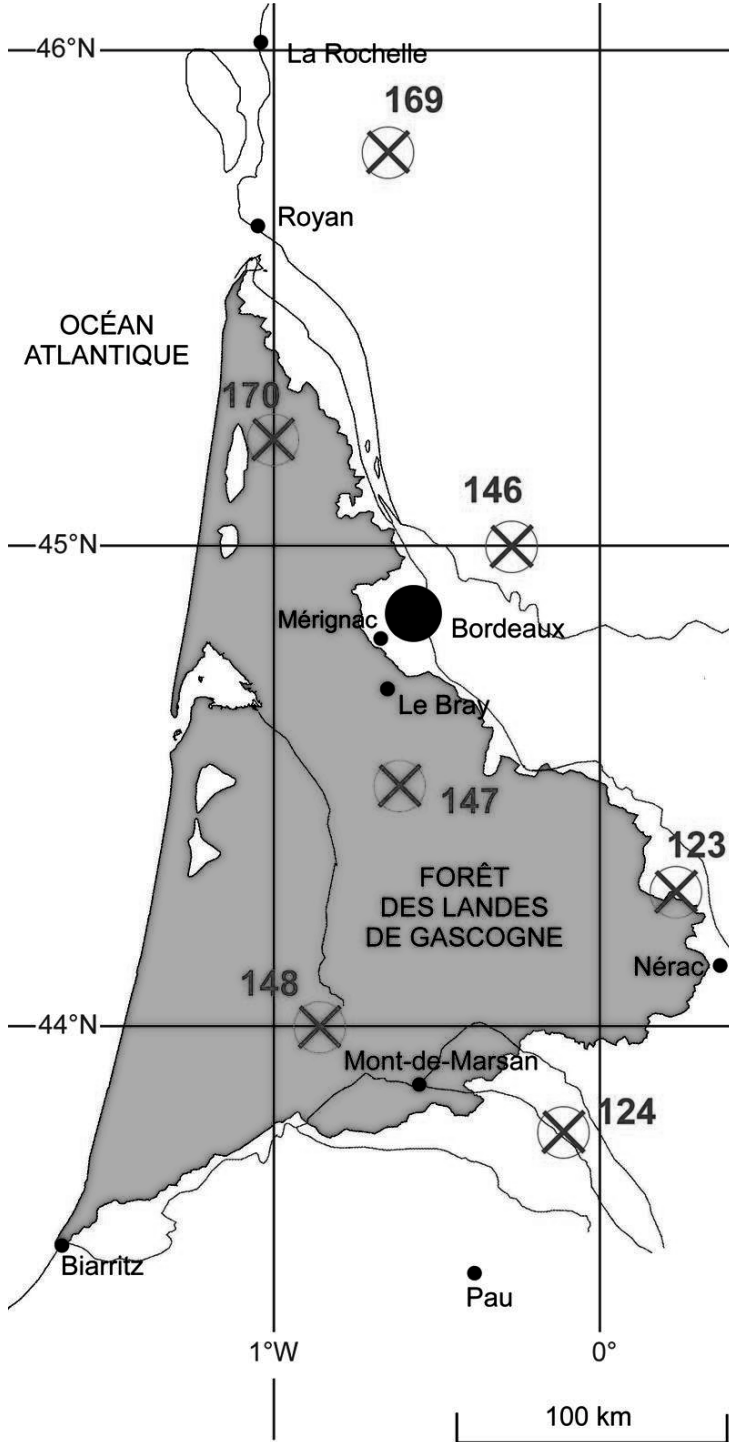
The objective of the present study is to assess the impacts of climate change within the one million hectare Landes de Gascogne pine forest in south-west France. This region is characterised by significant gradients in temperature and humidity from the west (Atlantic coast) to eastern limit (200 km inland), and in rainfall from north (western central plain of France) to south (Pyrenean foothills). Average annual wood production depends strongly on water availability.

We used a process-based forest growth model (GRAECO) to simulate the behaviour of maritime pine (*Pinus pinaster*) forest over a 10-yr period, under standard and modified (2 x CO<sub>2</sub>) climates generated by the ARPEGE climate model of METEO-France at a 60 x 60 km resolution. We present results for the impacts of climate change on the main ecosystem fluxes and tree growth at seven representative grid points covering the climate range of the Landes de Gascogne Forest.

## Regional Characteristics

The region studied was the Landes de Gascogne Forest, covering one million hectares in south-west France (Figure 1). The forest consists mainly of even-aged stands of maritime pine growing on shallow sandy soils with a low nutrient content (pH 4), and a water storage capacity between 100 and 120 mm. Mean annual temperature varies from 10 to 15°C (from west to east) and annual rainfall from 750 to 1250 mm yr<sup>-1</sup> (from north to south).

Precipitation is not uniform over the year, autumn and winter receiving two thirds of the annual total. The Penman evapotranspiration rate exceeds rainfall by 120–150 mm in summer and autumn, and consequently most of the forest region experiences 2–6 weeks of soil drought in 3 out of every 4 years (Choisnel et al. 1987). Mean annual wood production (estimated from measurements by the National Forest Inventory in 1998) is close to 12 m<sup>3</sup>ha<sup>-1</sup>yr<sup>-1</sup>, but varies from 6 to 14 m<sup>3</sup>ha<sup>-1</sup>yr<sup>-1</sup> depending on the water table level and annual rainfall.



**Figure 1.** The Landes de Gascogne Forest (shaded area), showing the location of the seven grid points (⊗) considered in the modelling analysis.

## Climate Scenarios

The ARPEGE-IFS climate model of Météo-France was used to generate two 10-yr series of climate data (for atmospheric CO<sub>2</sub> concentrations of 354 and 708 p.p.m.v., respectively) with a spatial resolution of 60 x 60 km over the French national territory. The ARPEGE climate predictions were calculated using the sea surface temperature predicted by the coupled ocean-atmosphere model HadCM2 of the UK Hadley Centre. The climate data were extracted from long-term simulations (1860–2100) at each CO<sub>2</sub> concentration, assuming the ocean response time to regional climate anomalies is longer than a decade (Déqué et al. 1998). The sulfate aerosol concentration followed the observed trend from 1860 to 1990 and was fixed at its 1990 level thereafter. The concentrations of other greenhouse gases were increased at the same rate as CO<sub>2</sub> (+1% yr<sup>-1</sup>). The soil-vegetation scheme used in ARPEGE is the surface model ISBA (Noilhan and Planton 1989), which provided lower boundary conditions to temperature and moisture.

ARPEGE generated 6-hourly values of temperature and relative humidity, and daily sums of precipitation and global radiation. These data were disaggregated to hourly values using empirical equations derived from a 10-yr series of hourly data measured at Merignac Meteorological Station (Pluviaud 2000) (cf. Rasse et al. 2001).

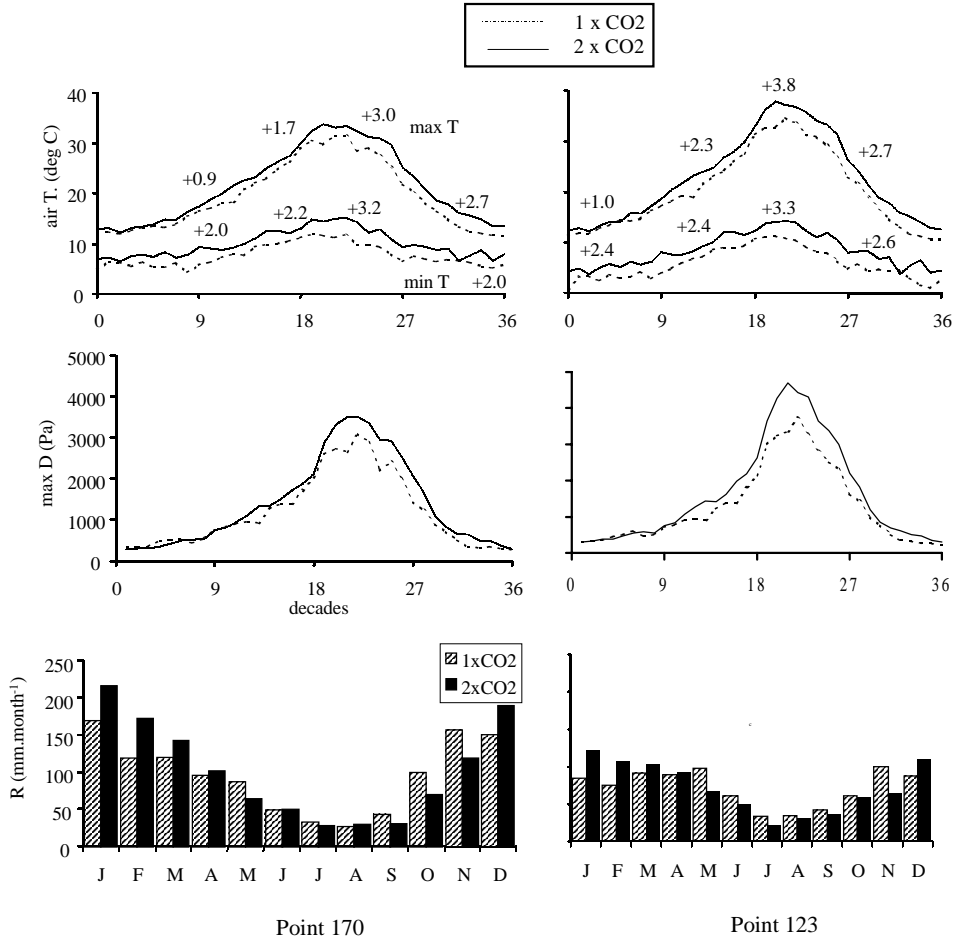
Figure 2 summarises the main climate trends predicted under the standard and modified climate scenarios, for two grid points corresponding respectively to the most western (170) and most eastern (123) points of the Landes Forest area (Figure 1). Doubling CO<sub>2</sub> resulted in a drier climate over the entire region, the greatest impact being in the continental part. The three main climate responses induced by doubling CO<sub>2</sub> were: (i) increased temperature; (ii) increased atmospheric water vapour saturation deficit; and (iii) a shift in the seasonal rainfall distribution from summer and autumn to winter and spring. The increases in temperature and air humidity deficit were most pronounced in summer and autumn. The seasonal difference in precipitation between winter and summer was enhanced under the 2 x CO<sub>2</sub> scenario: winter and spring precipitation increased by 25%, while summer and autumn precipitation was reduced, the annual total remaining largely unchanged. In summary, the seasonal imbalance in rainfall and evaporative demand was amplified under 2 x CO<sub>2</sub>.

## Overview of GRAECO

The forest model GRAECO (GRowth and Allocation based on ECOphysiological processes) simulates the energy, water and carbon fluxes through the soil-vegetation-atmosphere continuum, represented as one soil layer and two vegetation layers (understorey and trees). The model has a time step of 1 hr and predicts carbon allocation and tree growth. The driving variables are standard meteorological variables. GRAECO does not describe nutrient cycling processes and therefore does not predict nutritional limitations on growth (see Cannell and Thornley 2001; Corbeels et al. 2001; McMurtrie et al. 2001).

Canopy processes are described using conventional sub-models such as Rutter's rainfall interception model, the Penman-Monteith equation, and the Jarvis-Stewart stomatal conductance model (Loustau et al. 1992; Granier and Loustau 1994). Light interception is calculated using Beer-Lambert's law with the diffuse and direct components treated separately (Berbigier and Bonnefond 1995). Gross primary productivity is calculated from the light absorbed by each layer and a light-use efficiency coefficient (LUE,  $\epsilon$ ) which depends on environmental variables and stand age (Porté 1999; Porté and Loustau 2001). In particular, the CO<sub>2</sub> impact on pine LUE is derived from the results of the MAESTRO model for a *Pinus radiata* stand on poor nutrient conditions (Medlyn 1996).





**Figure 2.** Average annual courses of minimum and maximum air temperatures (upper graphs), daily maximum water vapour saturation deficit (middle graphs), and monthly precipitation (lower graphs) at two grid points of the Landes de Gascogne, predicted by the ARPEGE climate model under standard (1 x CO<sub>2</sub>) and modified (2 x CO<sub>2</sub>) climate scenarios. Numbers next to the temperature curves indicate the corresponding temperature anomaly.

Heterotrophic respiration is calculated using a  $Q_{10}$  temperature dependence and a seasonally-varying reference rate. Autotrophic maintenance respiration is calculated according to Bosc et al. (in preparation). Growth respiration is a constant fraction (0.28) of the carbon incorporated into new biomass. The net gain in canopy carbon is distributed among individual trees in proportion to their contribution to stand leaf area index (LAI).

Carbon allocation and tree growth are described following the 3-PG approach (Landsberg and Waring 1997; see also Landsberg et al. 2001). Net carbon gain per tree is partitioned between above- and below-ground parts according to a water stress index (i.e. the ratio between potential to actual transpiration of pines), and then further allocated to six above-ground and two below-ground biomass compartments in accordance with allometric relationships established by Porté et al. (2000, 2001) for three maritime pine stands (aged 7, 27 and 32 yr, respectively).

## Model Parameterisation

We used site parameters (e.g. soil depth, hydraulic conductivity – pressure head relationship, water table minimum and maximum depths) measured at Le Bray (Figure 1). Le Bray is part of the Fluxnet network and CarboEuroflux projects, and was a pilot site for calibrating process models in the LTEEF-1 and LTEEF-2 projects. In this study, we ran model simulations for two age classes, 8–17 yr and 18–27 yr, for which GRAECO was initialised with stand data measured at Le Bray at ages 8 and 18 yr, respectively. These data included the LAI of each foliage cohort, and the biomass, circumference and height of each tree within a one hectare area.

## Model Evaluation

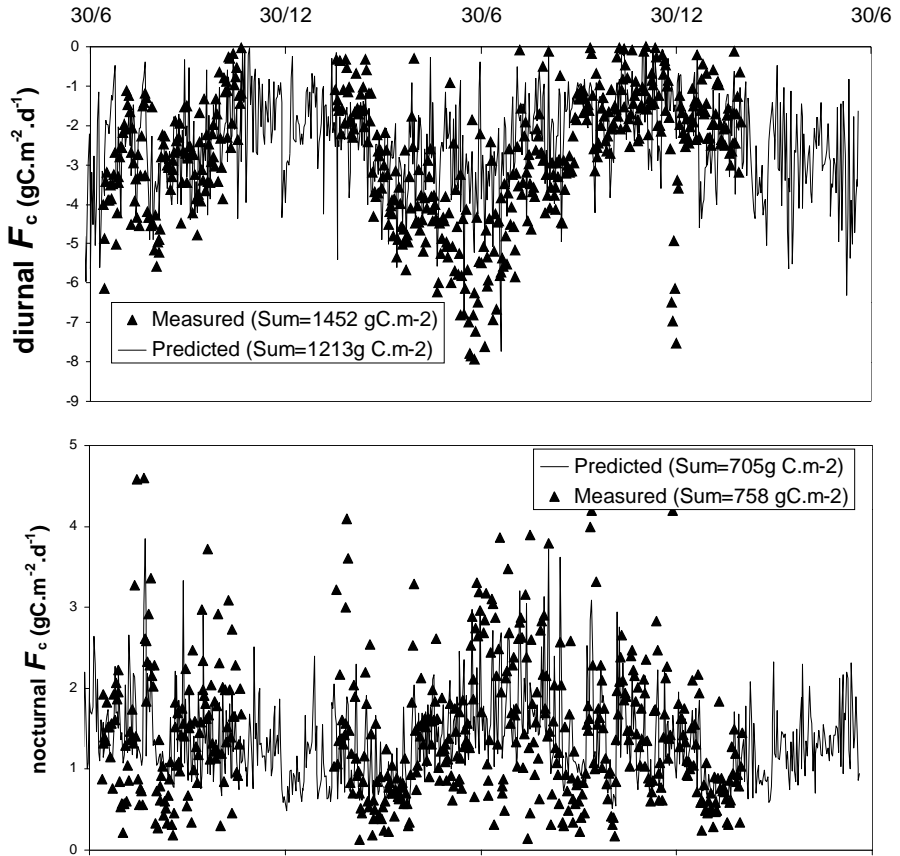
The sub-models for canopy conductance, transpiration and rainfall interception have been evaluated previously (Loustau et al. 1992; Granier et Loustau 1994). GRAECO has also been evaluated in terms of the predicted long-term water-use efficiency, using carbon isotope analysis of the cellulose of early and late wood (Porté and Loustau 2001). Figures 3–5 illustrate how GRAECO predicts with reasonable accuracy the observed short- and long-term trends in carbon fluxes, soil moisture (including duration of soil water deficits) and stem growth, without systematic bias or drift, even if some discrepancies are observed (e.g. in diurnal CO<sub>2</sub> fluxes). These results were obtained with data collected at one site only. Further evaluation over a range of site and climate conditions is necessary.

## Model Responses to Climate Change

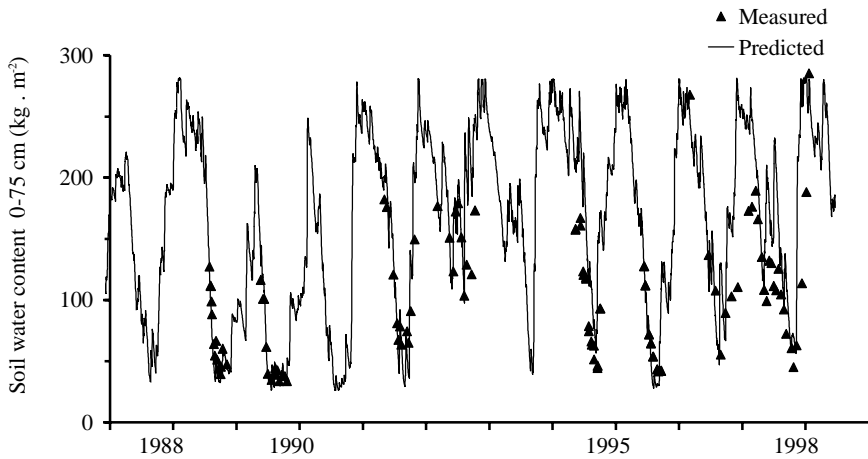
### Evapotranspiration

Despite the significantly enhanced seasonal imbalance in evaporative demand and precipitation under 2 x CO<sub>2</sub>, the annual transpiration rate of trees, averaged over the 10-yr simulation period, was reduced by only 40 mm yr<sup>-1</sup> while understory transpiration was increased by 20 mm yr<sup>-1</sup>. The annual sum of soil moisture deficit was increased by 10%. At the central grid point (147) the maximum annual rates of tree transpiration were remarkably similar under the two climate scenarios (Figure 6). This behaviour is consistent with the sensitivity analysis of the water balance sub-model by Loustau et al. (1998), which showed that tree stomatal closure under increased air humidity deficit (VPD) forces the transpiration rate to plateau when VPD exceeds 2 kPa. Moreover, the stomatal conductance of pine species is generally insensitive to CO<sub>2</sub> concentration (Picon et al. 1996).

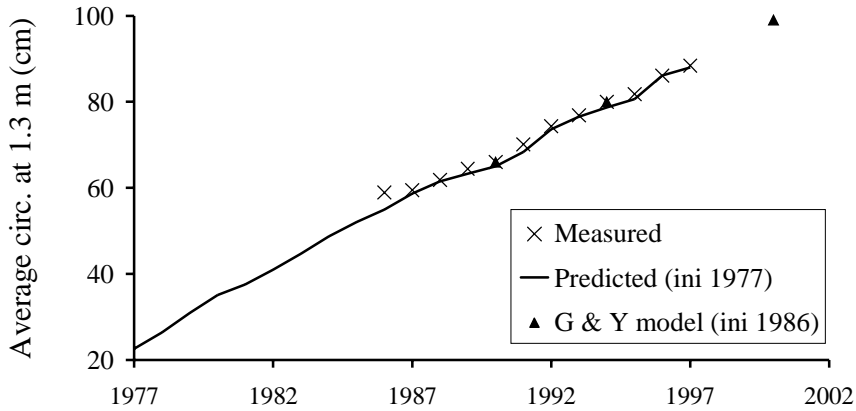
The annual course of soil water deficit, averaged over the 10-yr simulation period, showed a slight increase in summer and autumn under the 2 x CO<sub>2</sub> scenario. The duration of the water stress period – when soil water deficit exceeded 70% – was not significantly extended. In summer, however, soil water was depleted more severely while in winter, the larger excess in rainfall led to increased soil drainage (data not shown). This pattern showed little spatial or age variation. Stomatal responses to air and soil humidity deficits ensured that annual stand evapotranspiration was relatively conservative, consistent with the hypothesis of Roberts (1983).



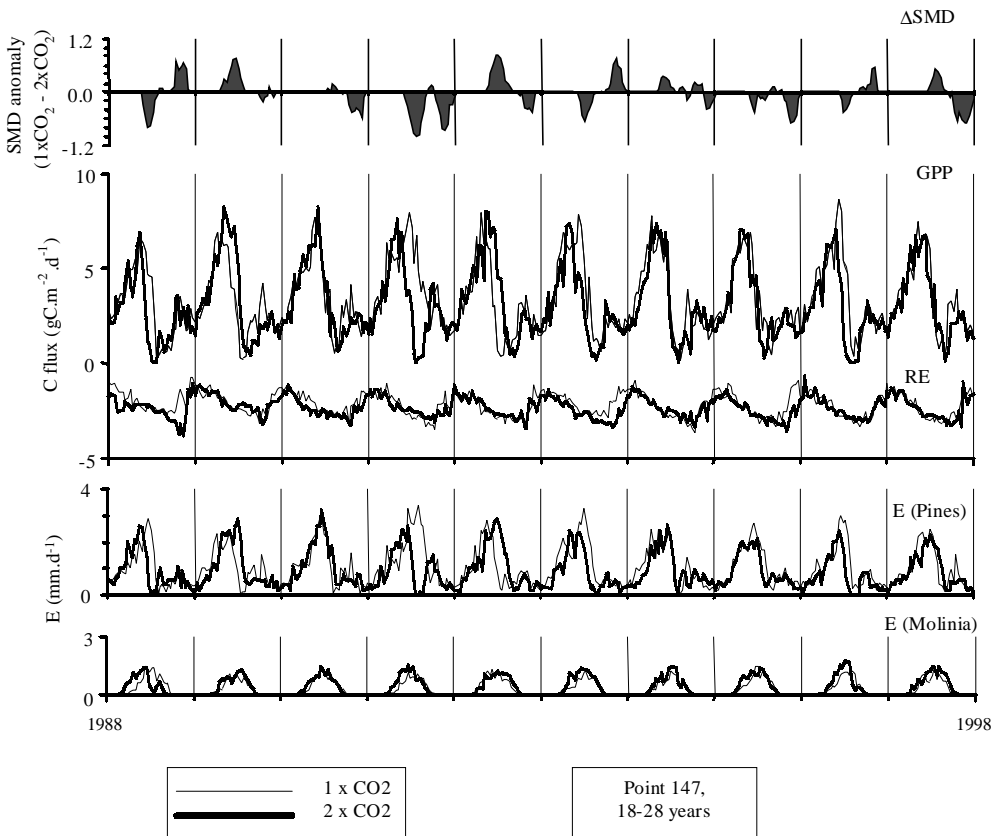
**Figure 3.** Comparison of measured daily net ecosystem C exchange at Le Bray with values predicted by GRAECO. Measurements were performed by eddy covariance from June 1996 as part of the EUROFLUX project (Berbigier et al. 2001).



**Figure 4.** Comparison of measured soil water content from 1987 to 1998 at Le Bray with values predicted by GRAECO.



**Figure 5.** Predicted and measured average circumference of trees at Le Bray. Measurements were performed on the entire population of a 4 ha plot, with  $n$  varying from 3600 to 2000 trees. GRAECO and the Lemoine growth and yield model (Lemoine 1991; 1995) were run using, respectively, 1977 and 1986 data for initialisation.



**Figure 6.** Time course of pine and understorey transpiration ( $E$ , lower graphs), gross primary productivity and ecosystem respiration ( $GPP$  and  $RE$ , middle graphs), and soil moisture deficit ( $\Delta SMD$ , upper graph) from ages 18 yr (in 1988) to 27 yr (in 1998) at grid point 147, predicted by GRAECO under standard ( $1 \times CO_2$ ) and modified ( $2 \times CO_2$ ) climate scenarios.

## Carbon fluxes

Annual GPP decreased slightly under the 2 x CO<sub>2</sub> climate scenario (Figure 6), while the ecosystem respiration flux (RE) was largely unaffected. The direct CO<sub>2</sub>-fertilisation effect on GPP was largely offset by increased air and soil humidity deficits. The CO<sub>2</sub>-fertilisation effect was expressed mainly through a 10% increase in LUE, derived from a modelling analysis of radiata pine growing in comparable site conditions (Medlyn 1996).

The climate change impact on ecosystem respiration reflects two opposing effects. Increased temperature tends to enhance both heterotrophic and autotrophic respiration. Conversely, slower growth and lower standing biomass lead to a reduction in growth and maintenance respiration per unit ground area. Table 1 shows that the combined climate and CO<sub>2</sub> effects resulted in a reduction in NEP at six grid points of the Landes Region, and a slight increase for point 170, the latter being closest to the coast where climate change impacts on soil moisture are negligible.

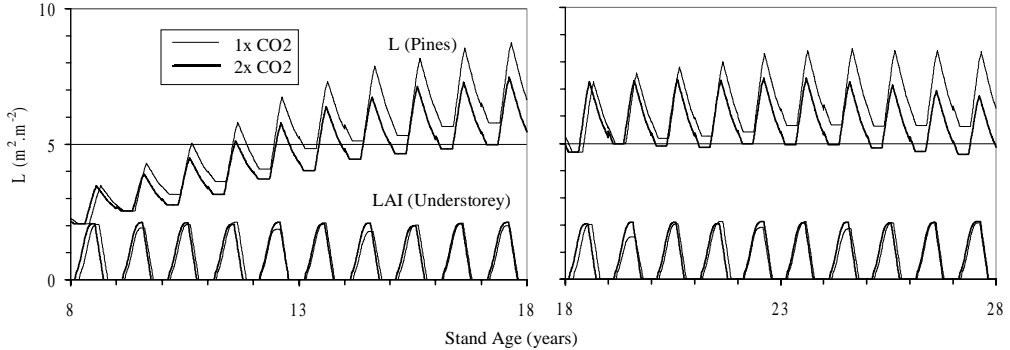
**Table 1.** Annual mean NEP, final biomass (Wa, above-ground; Wr, below-ground), mean DBH and height predicted by GRAECO for the standard (1 x CO<sub>2</sub>) climate scenario (with % change under 2 x CO<sub>2</sub>) at the seven grid points. Ten-year simulations were run for two age classes, 8–17 yr (a) and 18–27 yr (b).

Grid point	NEP [gC m <sup>-2</sup> yr <sup>-1</sup> ]	Wa [kg m <sup>-2</sup> ]	Wr [kg m <sup>-2</sup> ]	DBH [cm]	Height [m]
(a)					
123	360 -10%	13.8 -11%	7.1 -4%	15.7 -6%	10.9 -1%
124	362 -4%	14.0 -5%	7.3 —	15.8 -3%	10.9 -1%
146	425 -8%	15.8 -10%	7.9 -3%	16.8 -6%	11.1 -1%
147	370 -6%	14.0 -8%	7.1 -1%	15.8 -7%	10.9 -1%
148	329 —	12.9 -2%	6.7 +3%	15.2 -1%	10.8 —
169	385 -6%	14.4 -8%	7.4 -2%	16.0 -4%	11.0 -1%
170	360 +3%	13.5 —	6.9 +5%	15.5 —	10.9 —
(b)					
123	416 -9%	21.8 -6%	10.7 -3%	24.8 -3%	16.7 -1%
124	418 -3%	21.8 -3%	10.8 —	24.9 -2%	16.7 —
146	492 -9%	24.1 -7%	11.6 -3%	26.2 -4%	16.9 -1%
147	428 -6%	22.1 -5%	10.8 -1%	25.0 -3%	16.8 —
148	381 +2%	20.8 —	10.2 +2%	24.2 —	16.6 —
169	445 -8%	22.5 -6%	11.1 -2%	25.3 -3%	16.8 —
170	417 +1%	21.8 -1%	10.7 +2%	24.8 —	16.7 —

## LAI and tree growth

Figure 7 shows that, under the 2 x CO<sub>2</sub> scenario, the predicted maximum needle area was lower (by 23% after 10 yr) and was reached sooner. The tendency of needle area to plateau between ages 8 and 18 yr is explained by the model through a shift in C allocation from shoot to root with increasing age and water stress. Climate change effects on the water stress index (the ratio

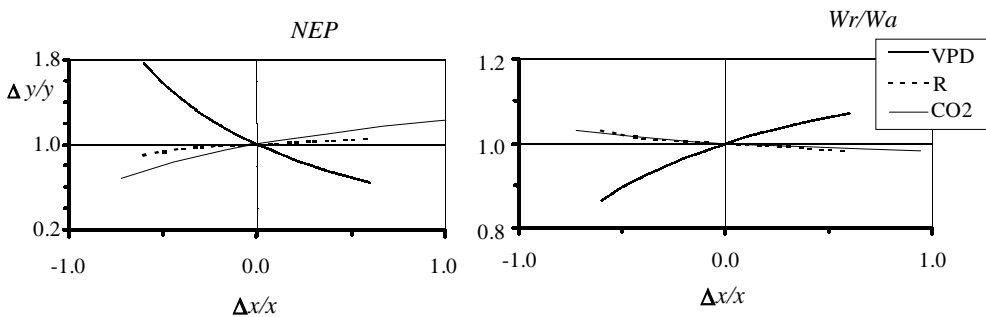
between actual and potential tree transpiration) were the main cause underlying the reduction in needle area. Understorey LAI was not affected. It is noteworthy that the annual peak in pine and understorey LAI was advanced by 5 to 10 days, due to higher temperature. The reduction in LAI led to decreased absorption of incident visible radiation by the pine canopy.



**Figure 7.** Time course of pine leaf area (all-sided) and understorey leaf area index (projected), predicted by GRAECO under standard ( $1 \times \text{CO}_2$ ) and modified ( $2 \times \text{CO}_2$ ) climate scenarios.

Above-ground tree biomass changed by between  $-11$  and  $+3$  % depending on stand age and site location (Table 1). The growth decrease was greatest for the most continental sites and younger stands. Carbon allocation shifted to below-ground components, and below-ground biomass was relatively unaffected, changing by between  $-4$  and  $+5$  %.

Figure 8 shows the sensitivity of annual NEP and root-shoot ratio ( $W_r/W_a$ ) to selected climate variables, as determined from a 1-yr simulation of an 8-yr-old stand. Both NEP and  $W_r/W_a$  were more sensitive to air humidity deficit than to rainfall or  $\text{CO}_2$  concentration, underlining the importance of air and soil water conditions to ecosystem functioning for this species and region.



**Figure 8.** Relative responses of annual net ecosystem C exchange (NEP, left) and root-shoot ratio ( $W_r/W_a$ , right) to atmospheric water vapour saturation deficit (VPD), rainfall (R) and atmospheric  $\text{CO}_2$  concentration. The responses were obtained from a one year simulation of an 8-yr-old stand at grid point 147. The baseline climate data were taken from an average year of the  $1 \times \text{CO}_2$  climate scenario.

## Discussion

The simulations presented here represent a consistent and coherent description of the impact of *one possible* climate change scenario over the next 50 years, rather than firm predictions of what *will* happen. For example, the standard (1 x CO<sub>2</sub>) climate scenario generated by ARPEGE is drier than the observed climate (Pluviaud 2000), and the present study does not represent the full range of humidity conditions observed in the region's coastal zone.

Further caveats concern GRAECO's representation of soil-vegetation-atmosphere feedbacks. Pine forest functioning is largely controlled by water availability at the atmospheric and soil boundaries of the ecosystem (i.e. by air and soil humidity deficits). Conversely, air and soil moisture contents both depend on canopy fluxes at the stand and regional scales (Jarvis and McNaughton 1986). The feedbacks acting across the ecosystem boundaries may play a major role in the Landes de Gascogne Forest, which extends over one million hectares of flat ground with low soil water retention capacity. At the stand level, the soil water feedback is incorporated in GRAECO (for which closure of the water balance has been checked). However, regional changes in drainage flow, hydrology and related impacts on the water table have not been accounted for. Similarly, the atmospheric feedback was not fully accounted for, the surface parameterisation of ARPEGE-IFS (the ISBA model) having been kept constant between the two scenarios. The results presented here should therefore be regarded as an assessment of only the direct, primary impacts of climate change on ecosystem water and carbon balances.

Despite these shortcomings, some useful insights have been gained. The empirical functions used to model the soil-vegetation-atmosphere water fluxes appear to capture the conservative hydraulic behaviour of the forest (Loustau et al. 1998). Pine transpiration and soil water balance are relatively unaffected by climate change, despite the dramatic changes in evaporative demand and rainfall distribution involved. It has been proposed that the functional significance of such hydraulic homeostasis lies in the requirement to maintain tree water potentials above the threshold for runaway cavitation (Tyree and Sperry 1988; Jones and Sutherland 1991; Cochard et al. 1996). This interpretation merits further study. As for most pine species studied so far, the stomata of *Pinus pinaster* do not respond to CO<sub>2</sub> concentration (Picon et al. 1996). CO<sub>2</sub>-induced alleviation of drought effects is therefore limited in the Landes de Gascogne Forest. In our simulations, the homeostatic behaviour of the water balance under climate change is mainly due to a water stress induced reduction in pine LAI and stomatal conductance. Understorey LAI is not affected.

The reduction in pine LAI and stomatal conductance decreased absorbed PAR and LUE, thereby reducing GPP. The increase in air and soil temperatures enhanced both autotrophic and heterotrophic respiration rates, but this was offset by the reduction in standing biomass, so that ecosystem respiration was relatively unaffected. Acclimation of soil respiration to temperature (Giardina and Ryan 2000) was not implemented in the model, although soil respiration was constrained by the annual carbon input to the soil (Janssens et al. 2001). These changes in GPP and RE led to reductions in NEP and tree growth at five of the seven grid points, with commercial wood production being further reduced by the drought-induced increase in below-ground C allocation.

The results presented here depend on the model assumptions and parameter values used, including their sensitivity to climate change. With these caveats in mind, it is worth noting the predicted importance of increased air humidity deficit, which dominates the direct impact of CO<sub>2</sub>. Another important and novel result of our study is that climate change impacts on NEP and tree growth could change sign over distances of only tens of kilometres.

We conclude that sub-regional variations in the impact of climate change on forest behaviour may be significant, and should be considered when horizontal gradients in climate

are important (e.g. due to coastal edges or surface discontinuities such as mountains and lakes). Such local variations may be of crucial concern to forest managers in their assessment of sustainable forest management practices. In regions where future water availability may be significantly reduced over a rotation, by climate change or other factors, the opportunity for optimising forest water-use by management should be considered. The effects of drought on forest might be alleviated by management practices such as increasing soil water storage capacity through deep ploughing, or reducing evapotranspiration through thinning control of LAI, understorey removal and soil mulching.

## Acknowledgements

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# Modelling Long-Term Changes in Forest Productivity and Soil Nitrogen Supply Following Conversion of Pasture to *Pinus radiata* Plantation

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## Abstract

Recent experimental studies have revealed that soil carbon (C) and nitrogen (N) content and soil N availability often decrease following conversion of improved pasture to *Pinus radiata* plantations. These decreases are a concern partly because of possible negative consequences for future forest productivity. This issue is investigated by applying the G'DAY model of C and N cycling in pasture and forest ecosystems to simulate the replacement of improved, legume-rich pasture by *P. radiata* plantations that are grown over several harvest cycles at a site in New Zealand. We illustrate how process models can be used for analysing constraints on long-term productivity by performing a sensitivity analysis of G'DAY's response to various rates of N removal. We find that simulated productivity declines over successive forest rotations and that the rate of decline is sensitive to N losses through wood harvesting, slash removal, and N leakage (i.e. leaching and soil gaseous emission). However, even when these N losses are zero, forest productivity still declines because of a gradual depletion of labile soil N reserves. Simulations are used to evaluate the mean annual increment (MAI) in wood volume over forty 30-year forest rotations. With harvesting only, simulated MAI declines from 44 m<sup>3</sup> ha<sup>-1</sup>yr<sup>-1</sup> in the first rotation to 18 m<sup>3</sup> ha<sup>-1</sup>yr<sup>-1</sup> in the fortieth rotation. The simulated MAI in the fortieth rotation is 13 m<sup>3</sup> ha<sup>-1</sup>yr<sup>-1</sup> with harvesting and leakage, and 11 m<sup>3</sup> ha<sup>-1</sup>yr<sup>-1</sup> with harvesting, leakage and 50% slash removal. If the simulation with harvesting only is modified so N removed in harvests is replaced by an equivalent fertiliser

addition, MAI is  $28 \text{ m}^3 \text{ ha}^{-1}\text{yr}^{-1}$  in the fortieth rotation. Results presented are tentative and should be regarded with caution until the model is fully tested.

*Keywords: forest productivity, G'DAY, harvesting, modelling, nitrogen cycling, Pinus radiata, sustainability*

## Introduction

Many empirical studies have quantified nitrogen (N) losses by forest ecosystems through harvesting, fire, leaching and soil gaseous emissions (see McMurtrie and Dewar 1997 and references therein). The link between ecosystem N losses and forest productivity has been modelled by Dewar and McMurtrie (1996) (henceforth referred to as DM) who developed a theoretical analysis of long-term changes in forest growth. DM considered a forest stand undergoing a repeated cycle with forest harvests followed by slash fires and re-planting. Over successive rotations, soil N supply may change due to N losses, but in the long-term the system approaches a steady state at which total N losses equal N inputs over a rotation. DM defined sustainable yield as the stemwood productivity achieved at that steady state, and developed a graphical method – together with some simplifying assumptions (see below) – for evaluating the steady state, based on the balance between N inputs and outputs. Using this method sustainable yield can be evaluated without the need to run simulations of forest growth over multiple rotations.

The simplicity of DM's steady-state analysis makes it a powerful tool for quantifying sustainability. However, it has the limitation that the analysis does not determine the timescale for achieving steady state. Another limitation to DM's analysis is their grossly simplified N model which consists of a single equation balancing cumulative N inputs over a rotation versus N removals in harvests and fire at the end of the rotation, and which does not simulate soil processes. In particular, one of their simplifying assumptions (referred to above) is that the total N supply rate (net mineralisation + external inputs) to the soil inorganic N pool is constant within each rotation, which may have important consequences for the rotation-averaged N loss rate.

This paper aims to overcome these two limitations to DM by running simulations over several forest rotations of a fully coupled model of C and N dynamics in plant and soil (G'DAY, Comins and McMurtrie 1993). We apply G'DAY to a 30-year-old stand of *Pinus radiata* planted onto grazed hill pasture at the Glenlean forest site near Masterton, New Zealand. The transition from pasture to *P. radiata* is a major land-use change in New Zealand, occurring at a rate of tens of thousands of hectares per year (Ford-Robertson 1997). Recently, several empirical studies have been undertaken to investigate changes in soil properties under this transition. Most of these studies, summarised in Table 1, found that soil carbon (C) and nitrogen (N) contents were lower and soil C:N ratios were higher under pine than pasture (Halliday et al. 2002). Other studies have measured lower levels of soil net N mineralisation and soil inorganic N under mature pine stands than pasture, though both may be elevated in the early stages of stand development (Nambiar 1996; Parfitt et al. 1997; Halliday et al. 2002). These decreases raise the question of whether the high productivity achieved by first rotation *P. radiata* stands planted on improved pasture will be maintained in subsequent rotations.

G'DAY has previously been applied to the Glenlean site by Halliday et al. (2002) who investigated changes in N cycling, litter quantity and quality as mechanisms for altered soil C and N following pasture-to-pine land-use change. In this paper G'DAY is used to simulate the transition from pasture to pine followed by a series of forest rotations with stands harvested at age 30 years and then re-established. Changes in simulated productivity over successive forest

**Table 1.** Published results on changes in total soil C and N after conversion of pasture or grassland to *P. radiata* plantation. Total C is a measure of total organic C (excluding roots), while total N is a measure of both organic and mineral N. Percentage changes are expressed as pine relative to pasture. It should be noted however that many of these studies sampled soils to a depth of 10cm or less (modified from Halliday et al. 2002).

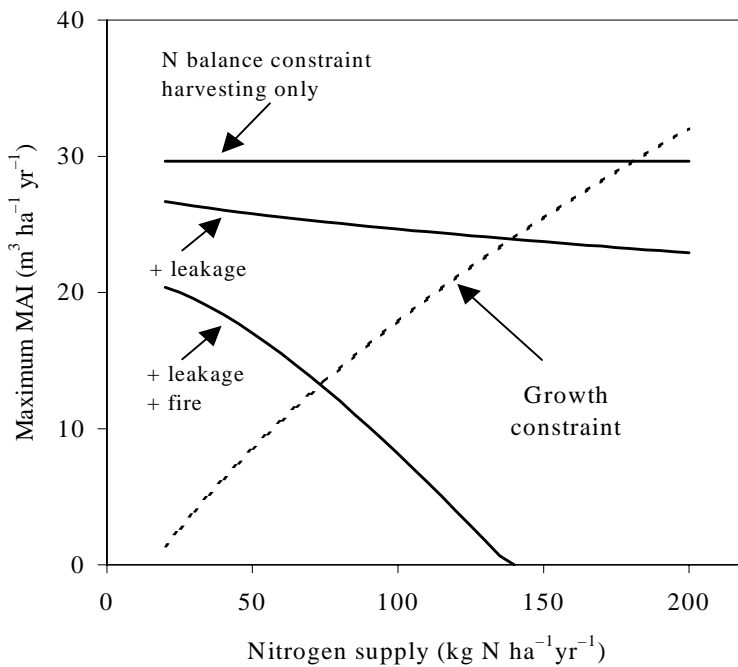
Source	Location	Details of pine site	Soil depth (cm)	Change in soil C	Change in soil N	Change in soil C:N ratio
Giddens et al. 1997	10 sites in NZ	13–30 yrs, 100–500 stems ha <sup>-1</sup>	10	-42 to +21%	-77% to +9%	-14% to +154%
Halliday et al. 2002	Glenlean site, Masterton, NZ	30 yrs, 350 stems ha <sup>-1</sup>	30	-1%	-39%	+51%
Parfitt et al. 1997	Palmerston North, NZ	20-yr-old stand, 250 stems ha <sup>-1</sup>	20	-26%	-31%	+6%
Perrott et al. 1999	Tikitere, NZ	19-yr-old stands, 50–400 stems ha <sup>-1</sup>	7.5	-35 to -22%	Not measured	Not measured
Ross et al. 1999	Puruki, NZ	19-yr-old stand, 1000 stems ha <sup>-1</sup>	20	-29%	-38%	+11%
Scott et al. 1999	NZ	Data from national databases and 4 detailed site studies	30, 50	-37 to -16%	Not measured	Not measured
Yeates and Sagggar 1998	Glendu Forest, NZ	12-yr-old stand, 1230 stems ha <sup>-1</sup>	5	-24%	-30%	+9%

rotations are compared with changes in soil N availability and results are related to DM's framework for quantifying sustainable productivity. The paper is structured as follows. We begin with a summary of DM's framework followed by an overview of the G'DAY model, and a description of how it was applied to *P. radiata* growing at the Glenlean site. Then we present results from simulations over multiple rotations of forest response to different rates of N removal in harvesting, leaching, soil gaseous emission and residue management.

## DM's Theoretical Framework for Quantifying Sustainability

DM developed a simple graphical analysis for quantifying the long-term sustainable productivity of managed forests. Their model takes account of the balance between N inputs (from fertilisers, atmospheric deposition and biological fixation) and N removals in wood harvesting, end-of-

rotation slash burns and 'leakage' (i.e. leaching and soil gaseous emissions). They assumed that external inputs of N from fertiliser additions, fixation and deposition occur at constant rates, and that a repeated management cycle is imposed consisting of harvesting at maximum mean annual stem volume increment (MAI), followed by slash burning and re-establishment. Over successive rotations, a nutrient feedback operates whereby depletion of site N reserves over one rotation leads to decreased stemwood productivity, and hence decreased N losses in harvesting and slash fires, in the next rotation. Eventually, this feedback leads to a steady state at which, cumulative N losses over a rotation equal cumulative N inputs from fixation and deposition. The growth constraint shown in Figure 1 represents the positive relationship between maximum MAI and plant-available N supply ( $S$ ) in any given rotation, where  $S$  includes external inputs (assumed constant) and net mineralisation (which varies between successive rotations).



**Figure 1.** Illustration of DM's graphical analysis of long-term sustainable yield, showing the growth constraint (dashes) and N-balance constraints (solid lines). Sustainable yield is given by the height of the intersection between the two constraint curves.

The steady-state N balance constraint, also shown in Figure 1, is the relationship between steady-state values of maximum MAI and  $S$ , imposed by the condition that over a rotation N inputs to the system equal N losses associated with removal of harvested wood, leaching, gaseous emissions and slash burns:

$$N_{\text{input}} = N_{\text{loss}} \text{ or } A T_h = H + F + L \text{ or } A = \frac{H}{T_h} + \frac{F}{T_h} + \frac{L}{T_h} \quad (1)$$

where:  $A$  is mean annual N addition rate from deposition and fixation (assuming no fertiliser input),  $T_h$  is rotation length (yrs),  $H$  is N loss in harvested stemwood,  $F$  is N loss due to fires that consume slash residue, and  $L$  is N loss due to leaching and gaseous emissions.

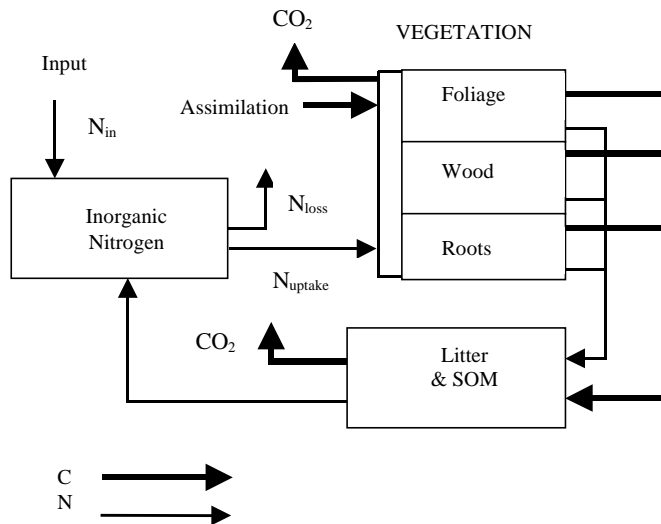
Because mean annual N removal in harvested wood ( $H/T_h$ ) is proportional to MAI, while mean annual N losses from slash burning ( $F/T_h$ ) and leakage ( $L/T_h$ ) depend positively on N supply ( $S$ ), Equation (1) (with  $A$  fixed) imposes a negative relationship between MAI and  $S$ , as illustrated by the N-balance constraint curves shown in Figure 1.

The system must be on the growth constraint curve in every rotation. If the system is above the steady-state N balance constraint curve, then N outputs exceed N inputs, so that N supply and maximum MAI will decrease over successive rotations with the system moving down the growth constraint curve. Below the N balance constraint curve, the opposite occurs, so that N supply and maximum MAI increase. In either case, the system moves along the growth constraint curve towards the steady state given by the intersection where both constraints are satisfied. Sustainable wood yield is then defined as the maximum MAI at the steady state. Using this approach, DM quantified the sustainable yield of managed *Eucalyptus* stands under various scenarios of N loss; sustainable yield was  $30 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$  if harvesting was the only N loss,  $24 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$  with N losses from harvesting and leakage,  $15 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$  with N losses from harvesting and slash burning, and  $13 \text{ m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$  with the combined effects of harvesting, fire and leakage (Figure 1). These results suggest that slash burning may represent a greater limitation to sustainable productivity than harvesting or N leakage.

## The G'DAY Plant Soil Model

### Overview of G'DAY

G'DAY is an ecosystem model describing C and N dynamics in plant and soil. It is based on the plant production models of McMurtrie (1991) and Medlyn et al. (2000) and the soil carbon and nutrient cycling model, CENTURY (Parton et al. 1987, 1993). The version of G'DAY used here does not include the revised decomposition sub-model (MIT) described in this volume by Corbeels et al. (2001).



**Figure 2.** Pools and fluxes of C and N in the G'DAY model. In pasture simulations the wood pool is omitted.

The pools and fluxes of C and N in G'DAY are shown in Figure 2. The forest version of G'DAY has three biomass pools: foliage, wood (which includes stems, branches and coarse roots) and fine roots. The pasture version has only foliage and root pools. The model has four litter pools (surface structural, surface metabolic, soil structural and soil metabolic) and three soil organic matter pools (active, slow and passive SOM). Processes represented include net primary production (NPP), allocation, tissue senescence, litter and soil decomposition, plant N uptake, N retranslocation, net soil N mineralisation, N input by biological fixation and atmospheric deposition, and N loss by leaching and soil gaseous emission. Brief descriptions of each process, and differences between the pasture and pine versions of G'DAY are presented in Table 2. For further details see Comins and McMurtrie (1993), Medlyn et al. (2000) and McMurtrie et al. (2001). G'DAY can be run either with a daily time step using daily meteorological data, or with constant mean-annual weather. In this paper constant mean-annual weather was used under the assumption that the Glenlean site (described below) is not water-limited.

### Parameterisation of G'DAY for the Glenlean site

Parameterisation of G'DAY was based on paired pasture and pine sites located at the Glenlean site in Ngaumu State Forest, near Masterton in the North Island of New Zealand (41° 01' S, 175° 57' E, elevation 320 m). The mean annual rainfall is 1347 mm, and mean annual temperature is 11.4°C. The pine site was a 30-year-old first rotation stand of *P. radiata* with a stocking density of approximately 350 stems ha<sup>-1</sup>. The pasture site, located on adjacent farmland, was grazed by sheep and cattle at a rate of approximately 10 animals ha<sup>-1</sup>, and fertilised. The sites were on similar silt loam soils. Further details are in Halliday et al. (2002).

G'DAY was parameterised for the pasture and pine systems using measurements at the Glenlean site and published data (Halliday et al. 2002). Many parameter values for the pine system were derived from recent applications of G'DAY to *P. radiata* growing at the 'Biology of Forest Growth' site near Canberra, Australia (Medlyn et al. 2000). Assumed rates of N uptake and leakage from the inorganic N pool under pasture, defined in Table 2, were  $\lambda_u = 10.2 \text{ yr}^{-1}$  and  $\lambda_{\text{loss}} = 1.8 \text{ yr}^{-1}$ , respectively, so that 15% of mineralised N was lost through leakage with 85% taken up by plants. A comprehensive list of parameter values is in Halliday et al. (2002). Because values of several parameters are uncertain, results presented below are tentative and should be regarded as illustrations of G'DAY's qualitative response to N removals.

### Simulations of Forest Response to Different Rates of N Loss

To simulate the transition from pasture to pine, the model was initially parameterised for pasture and run to equilibrium. At time zero, the pasture was replaced by pine trees, grazing ceased and N-fixation was reduced from the high rate under pasture (150 kg N ha<sup>-1</sup> yr<sup>-1</sup>) to zero. Pasture biomass was transferred to above- and below-ground structural and metabolic pools based on C:N ratio and lignin content using equations in Parton et al. (1993). Rates of N uptake and leakage from the inorganic N pool were altered, as specified below. The lignin contents of foliage and root litter were modified gradually over the first four years under pine to reflect a gradual increase in litter lignin content under forest (Halliday et al. 2002). At age 30 years trees were harvested with the C and N contents of wood removed from the site.

We simulated the following four scenarios for N losses: (1) harvesting only (with zero N leakage under forest and with harvest residue retained on site); (2) harvesting + leakage;



**Table 2.** Description of processes in G'DAY for pine and pasture. Full details are given in Comins and McMurtrie (1993), Medlyn et al. (2000), McMurtrie et al. (2000, 2001) and Halliday et al. (2002).

Process	Description in the model
Light interception	Beer's law, with separation into sun-lit and shaded foliage for forest, but not for pasture.
Leaf photosynthesis of pine	Rectangular hyperbolic function of light and leaf [N], responsive to temperature and [CO <sub>2</sub> ] (Medlyn et al., 2000).
Gross primary productivity (GPP) of pine	Derived by integration down the canopy with N declining exponentially (Medlyn et al., 2000), giving GPP analytically as a function of total foliage C, mean foliage N:C ratio, incident photosynthetically active radiation (PAR), mean growing season temperature.
GPP of grass	Proportional to absorbed PAR with light use efficiency a function of mean foliage N:C ratio (McMurtrie 1991).
Respiration	Fixed fraction of GPP.
Net primary production (NPP)	GPP minus Respiration.
C allocation	Fixed fractions of NPP allocated to foliage, wood, fine roots.
Senescence	Constant litterfall rates for foliage, wood and fine roots.
Plant N:C ratios	Fine root and wood N:C ratios proportional to foliage N:C.
Litter N:C ratios	Foliage and fine root litter N:C proportional to live foliage N:C.
N retranslocation	Fixed fractions of leaf, non-structural wood and fine root N.
Soil C dynamics	Derived from the CENTURY model (Parton et al. 1987, 1993). Decomposition rates responsive to temperature and soil moisture.
Soil N dynamics	Rate of change of soil inorganic N ( $N_{inorg}$ ) = gross N mineralisation + N fixation - N immobilisation - plant N uptake - N loss through leaching and soil gaseous emission.
Soil N:C ratios	N:C ratios of substrate entering active, slow and passive SOM represented as linearly increasing functions of $N_{inorg}$ .
N input	Constant inputs from atmospheric deposition and symbiotic fixation.
N loss	Leaching and soil gaseous emission rates proportional to soil inorganic N pool ( $N_{loss} = \lambda_{loss} * N_{inorg}$ ).
N uptake	Proportional to soil inorganic N pool ( $N_{uptake} = \lambda_u * N_{inorg}$ ).
Grazing	Consumption rate proportional to foliage biomass.
C loss in grazing	Fixed fraction of C eaten, remainder entering soil as faeces.
N input to soil by grazers	Fixed fraction of N eaten, as either faeces with constant N:C ratio or urine.

(3) harvesting + leakage + 50% harvest residue removed from site; and (4) harvesting + N fertiliser addition equal to N removed in harvest. Scenarios 1 to 3 represent increasing N losses, and (4) is the simulation with least N loss. Simulations were conducted over 40 rotations with a fixed rotation length of 30 years with the same management practices imposed after each harvest.

### Scenario 1: harvesting only

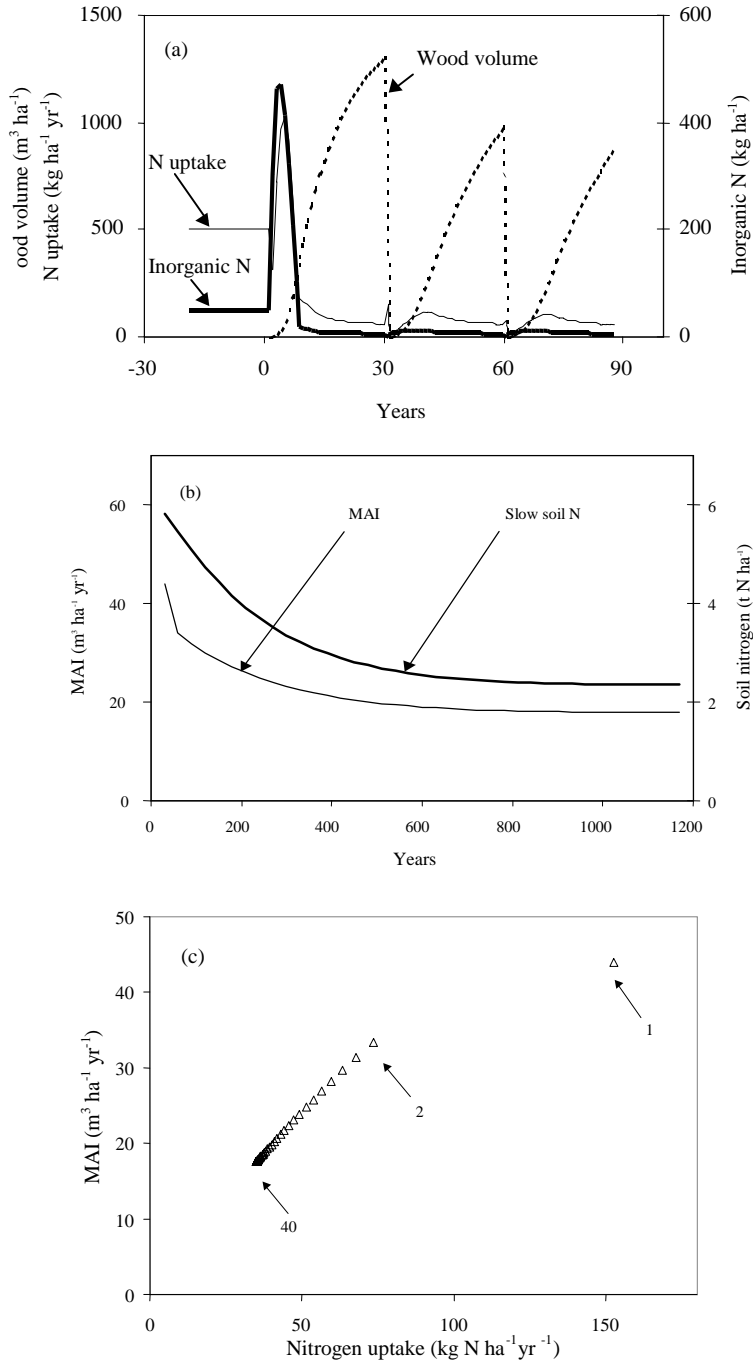
For Scenario 1 we assumed that all wood C and N were removed in harvests with no other N loss under forest ( $\lambda_{\text{loss}} = 0$ , Table 2). The rate of N uptake was proportional to soil inorganic N (Table 2), with  $\lambda_u$  assumed to increase from 1.8 yr<sup>-1</sup> at the start of each forest rotation to a maximum rate of 11.4 yr<sup>-1</sup> after 10 years (Halliday et al. 2002), reflecting expansion of the root system. After each harvest the residue (leaves and fine roots) was transferred to the above- and below-ground litter pools based on the C:N ratio and lignin content of residue. Simulated wood volume, N uptake rate ( $N_{\text{uptake}}$ ) and soil inorganic N ( $N_{\text{inorg}}$ ) were highest in the first rotation (Figure 3a). Values of  $N_{\text{inorg}}$  were elevated early in the first rotation when net N mineralisation, enhanced as a residual effect of the highly fertile pasture system, exceeded tree uptake requirements, and when soil N immobilisation was reduced because of low tree litter input. Within 10 years, however,  $N_{\text{inorg}}$  declined to below its value under pasture. The reduced  $N_{\text{inorg}}$  under pine was a consequence of the increased quantity of low quality (structural) litter, resulting in reduced N input to soil and increased N immobilisation during its decomposition (Halliday et al. 2002). Reduced  $N_{\text{inorg}}$  in turn caused an increase in C:N ratio of substrate entering slow SOM, resulting in a gradual decline in slow soil N (Figure 3b), and hence in the rotation-averaged N uptake rate (Figure 3c). Both MAI and N uptake were highest in the first rotation and declined over subsequent rotations (represented by the sequence of points 1, 2... 40 progressing down and to the left in Figure 3c, analogous to DM). Simulated MAI declined rapidly at first (from 44 to 23 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> in rotations 1 and 10, respectively), then gradually until it reached a steady-state value of 17 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>.

### Scenarios 2–4: effects of different rates of N removal

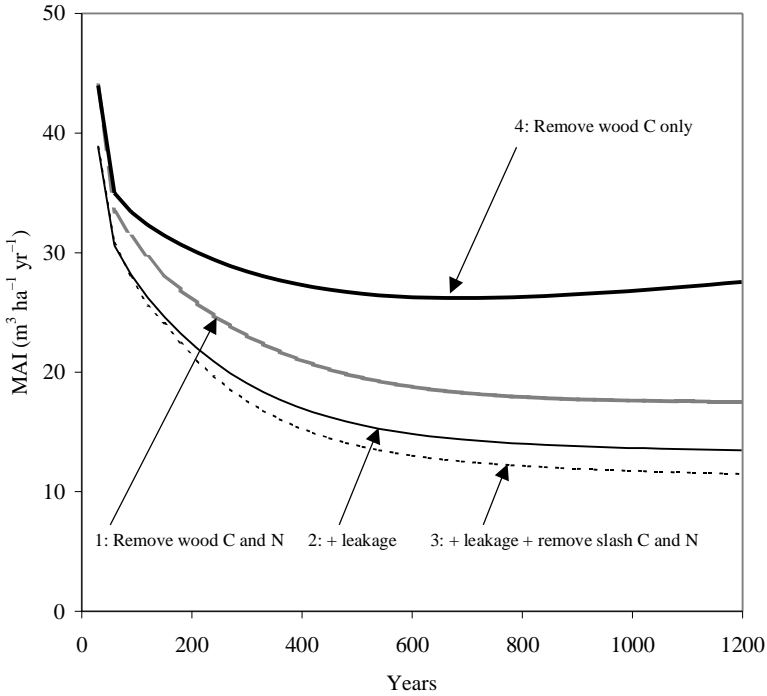
Under Scenario 2 (harvesting + N leakage) the rate of N loss through leakage ( $\lambda_{\text{loss}}$ ) was set to 0.6 yr<sup>-1</sup> under forest. Thus, N loss represented 25% of N loss + N uptake at the start of each forest rotation, declining to 5% at age 10 years. Simulated N losses were large early in the first rotation because of elevated levels of soil inorganic N (as illustrated in Figure 3a for Scenario 1). Consequently, in the first rotation under Scenario 2 tree N uptake and MAI were considerably lower than under Scenario 1, as illustrated in Figure 4. Between rotations 1 and 10, MAI declined from 39 to 19 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>, and was 13 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> at steady state.

Scenario 3 considered the effect of removing 50% of slash C and N at harvest. The slash removed included both shoot biomass and above-ground litter. Under this scenario MAI was 39 and 18 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> in the first and tenth rotations, respectively, and 11 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> at steady state.

Simulations of Scenarios 1 to 3 indicate that the decline of MAI over successive rotations is sensitive to N losses in harvests, leaching/gaseous emissions and slash removal. This conclusion raises the question of whether MAI will still decline if there is no ecosystem N loss. This question was investigated by Scenario 4 where wood C and N were removed in harvests (as under Scenario 1), but N was added to the soil inorganic N pool so there was no net N loss. Under this scenario simulated MAI declined from 44 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> in the first rotation to 28 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> after 10 rotations (Figure 4). This simulation illustrates that MAI declined even when ecosystem N loss was zero. Here MAI declined because of a gradual decrease in soil N supply due to a shift of soil N reserves from relatively available soil organic matter (active and slow SOM) to passive SOM. The decline of slow soil N, illustrated in Figure 3b for Scenario 1, occurred primarily because the slow pool's C:N ratio, which depends on  $N_{\text{inorg}}$  (Parton et al. 1993; Halliday et al. 2002; McMurtrie et al. 2001), increased from 12 to 30 during the simulation. The simulation for Scenario 4 (Figure 4) shows a



**Figure 3.** Simulation with harvesting only (Scenario 1). The simulation was initiated by running the pasture to equilibrium. At time zero pasture was replaced by a pine stand. (a) Simulated soil inorganic N (thick solid line), plant N uptake (thin solid line) and wood volume (dashes) for pasture (time -20 to 0) and three 30-year forest rotations (time 0 to 90 years). (b) Simulated mean annual wood volume increment (MAI) at harvest and slow soil N content over 40 rotations. (c) The relationship between MAI and rotation-averaged plant N uptake over 40 rotations.



**Figure 4.** Simulated mean annual increment (MAI) in wood volume over forty 30-year forest rotations under Scenarios 1 to 4: 1 – Harvesting only; 2 – Harvesting + N leakage; 3 – Harvesting + N leakage + removal of 50% of slash C and N; 4 – Harvesting + N addition equal to N removed in each harvest. Simulated MAIs in the fortieth rotation were 18, 13, 11 and 28  $\text{m}^3 \text{ha}^{-1} \text{yr}^{-1}$ , respectively for Scenarios 1 to 4.

gradual increase in MAI after 700 years. This increase occurred under Scenario 4 because N input was constant whereas output was zero, so that the system accumulated N and did not achieve DM's steady state where input equals output.

## Discussion

DM's analysis provides a theoretical framework for assessing constraints on sustainable forest yield. Our simulations are complementary to DM's analysis in several respects. Our simulated changes in forest productivity over successive rotations are related to changes in plant N uptake, and follow a unique curve (Figure 3c) that is analogous to DM's growth constraint between MAI and soil N supply (Figure 1). We find that the same unique relationship between MAI and plant N uptake over successive rotations applies to Scenarios 2 to 4 (results not shown). Under Scenarios 2 and 3 with higher N loss than Scenario 1, MAI and N uptake decreased more rapidly over successive rotations and values reached at steady state were lower than indicated in Figure 3c for Scenario 1 (cf. DM's analysis illustrated in Figure 1). As predicted by DM, MAI approached a steady state under Scenarios 1 to 3, but not under Scenario 4 where the system accumulated N and did not achieve N balance.

There are several differences however between our results and DM's. One difference arises because DM considered only N inputs and losses at the ecosystem level, whereas G'DAY

also considers internal N fluxes into and out of soil pools. Thus, at the steady state achieved by G'DAY under Scenarios 1 to 3 (Figure 4), rotation-averaged N fluxes into and out of each soil pool were equal, in addition to the balance between inputs and losses at the ecosystem level. In our simulations of G'DAY changes in N fluxes into and out of slow-turnover soil pools have a considerable impact on soil N supply and hence on forest productivity, as shown previously by McMurtrie et al. (2000, 2001) in the context of modelling carbon sequestration by forests. The effect is illustrated by Scenario 4 (Figure 4) where productivity declined over several rotations although ecosystem N loss was zero. This result identifies a limitation to DM's analysis of sustainability, and indicates that a plant-soil model with internal C and N cycles is required for predicting changes in productivity over multiple rotations. A further illustration of DM's shortcomings is that the soil N supply and plant N uptake rates simulated by G'DAY varied considerably within each rotation (Figure 3a), whereas DM's theoretical analysis assumes that soil N supply rate is constant within each rotation, which may lead to an underestimate of N loss rates, especially for sites where leakage is important.

Consequently, another difference between DM and our work concerns the importance of N leakage through leaching and soil gaseous emissions. DM assumed that only 5% of N not taken up by trees is lost through leakage, reflecting their focus on *Eucalyptus* growing on relatively dry sites. Consequently, DM predicted that sustainable MAI was relatively insensitive to N leakage. In contrast, we predict a large effect of N leakage on MAI (compare Scenarios 1 and 2, Figure 4). Our prediction applies to forest planted onto N-rich pastures where N leakage was high during the first forest rotation, leading to greatly reduced MAI after the first rotation (Figure 4). In subsequent rotations, although N leakage under Scenario 2 was reduced, the difference in MAI between Scenarios 1 and 2 was maintained (Figure 4). For this site, a dynamic treatment of N supply and leakage within each rotation is appropriate.

A useful insight from our modelling concerns the timescale for the decline of MAI, which was outside the scope of DM's analysis. Simulations in Figures 3 and 4 suggest that several centuries are required for stabilisation of MAI.

DM derived an equation for the N addition required to sustain a given yield. A similar analysis could be performed by running simulations of G'DAY. This analysis would be based on the relationship between MAI and N supply illustrated in Figure 3c. This relationship could be used to determine the N supply corresponding to a given yield, and model runs could be used to determine the N addition required in each rotation to achieve that rate of N supply.

The simulations presented here illustrate the model's sensitivity to different levels of N removal. The scenarios considered are not intended to closely represent specific management practices. For instance, our harvesting strategy, involving removal of all wood C and N including branches, stems and structural roots, is extreme. Because our scenarios are unrealistic and because the model has not been fully tested, our prediction of declining productivity over time should be viewed as tentative. Before specific model predictions can be taken seriously, the model requires further testing and development, including application of G'DAY to long-term experiments where carbon and nutrient cycles are monitored over multiple rotations. We also emphasise that processes not considered in this paper may be important in explaining changes in productivity over successive rotations.

Our simulations do, however, illustrate a potential role for plant-soil models as tools for analysing constraints on long-term forest productivity, and for evaluating the benefits of management strategies (e.g. residue retention, fertilisation, legume inter-cropping) that may maintain site N capital. Furthermore, our evidence of the G'DAY model's sensitivity to altered N fluxes into and out of SOM pools highlights the importance of research into mechanisms of soil N release and immobilisation, leading to improved representations of these mechanisms in plant-soil models (Aber et al. 1998; Corbeels et al. 2001; McMurtrie et al. 2001).

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# Modelling the Effects of Harvest Residue Management on Soil Nitrogen Supply in Short Rotation Eucalyptus Plantations in Southwestern Australia

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## Abstract

An updated version of G'DAY (Generic Decomposition and Yield model) with a revised decomposition model is presented. The model is parameterised to simulate C and N dynamics of *Eucalyptus globulus* stands. Data from litterbag studies on decomposition of eucalypt slash residues are used to calibrate the decomposition model. The whole system model is tested against data on tree growth and N mineralisation of a first-rotation *E. globulus* stand in southwestern Australia. The model is then used to assess the effect of harvest residue management on soil nitrogen supply during growth of second rotation eucalypt. The model simulations suggest that retention of harvest residues will favour enhanced soil N supply for the next rotation. The risk of N stress due to N immobilisation only occurs during the first year of the second rotation.

*Keywords: Eucalyptus globulus, G'DAY, model, N mineralisation, slash residues*

## Introduction

Hardwood plantation forestry is a new and expanding agro-industry in southwestern Australia. *Eucalyptus globulus* is the most commonly planted species, primarily because of its high potential growth rate, short rotation length (10 years) and favourable pulpwood properties (Cromer 1996). More than 150 000 ha had been established by 1999 and the planting rate currently exceeds 25 000 ha yr<sup>-1</sup> (National Forest Inventory 2000). The region, which has a Mediterranean climate with hot dry summers and cool wet winters, has a high productivity potential because of favourable temperatures and abundant incident radiation. In this climate, growth rates in summer depend largely on the capacity of the soil to store water from winter rains (Hingston et al. 1998).

The natural fertility of soils in the southwest region of Australia is generally low. However, plantations are usually established on agricultural land, where soil fertility has been elevated by past annual fertiliser application and enrichment of soil organic matter by legume-based pasture systems. Sustaining plantation productivity in the future will largely depend on maintaining current levels of soil fertility in subsequent rotations. Evidence from agricultural research (e.g. Glendining and Powlson 1995) further suggests that soil fertility declines markedly within a period of some years where fertiliser application ceases or becomes less frequent. There is also a concern that high harvest intensity with increased exports of nutrients from the site will exacerbate soil fertility decline and result in a long-term reduction in site-productivity (see also McMurtrie et al. 2001).

Management of slash residues during the inter-rotation period following stand harvest is one option for manipulating site fertility. The effect of harvest residue management on soil N supply is a key issue, because N often limits productivity of fast growing *E. globulus* plantations (Judd et al. 1996). Experimentation dealing with this topic is limited, in part because of the time scale (several years) over which the effects may occur. Process-based models provide an alternative approach and can be valuable tools in analysing the medium to long-term consequences of alternative strategies for managing harvest residues in plantations. To be valuable, the model must include all relevant processes and driving variables that affect tree growth and N cycling (Ryan et al. 1996)

In this paper, we report on the use of an updated version of G'DAY (Generic Decomposition And Yield model) (Comins and McMurtrie 1993) to simulate tree growth and N cycling in *E. globulus* plantations, and analyse the impact of alternative harvest residue management options on N supply.

## Model Formulation

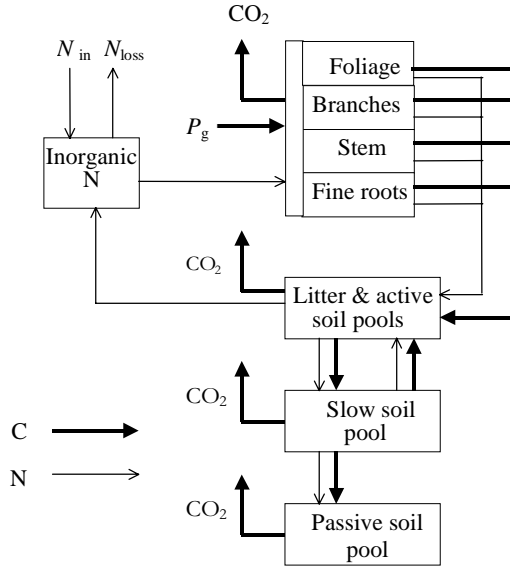
G'DAY is a process-based model of plant-soil ecosystems, that was initially developed to simulate C and N dynamics in forest ecosystems (Comins and McMurtrie 1993) (Figure 1).

The original version of G'DAY links the plant production model of McMurtrie and Wolf (1983) to the decomposition sub-model of CENTURY (Parton et al. 1993). The plant production model essentially assumes that net primary production (NPP) is proportional to intercepted photosynthetically active radiation (PAR) with modifications for soil water and nitrogen stress. It comprises four tree biomass pools: foliage, branches (plus coarse roots), wood (stems) and fine roots. These pools receive C through allocation of NPP, and N through allocation of N uptake and retranslocation. The CENTURY model simulates decomposition of plant litter and soil organic matter (SOM) (Parton et al. 1993). It was originally developed to investigate SOM dynamics in grassland ecosystems (Parton et al. 1987), but has more recently been extended for simulating soil C dynamics of tropical (Vitousek et al. 1994) and boreal (Peng et al. 1998) forests. This model has been extensively validated and applied in regional and global studies.

In the present study, we modified the decomposition model to better describe the interactions between C and N dynamics in decomposing litter. This adapted version simulates N mineralisation-immobilisation turnover (MIT) and is in part based on the structure of the soil C and N transformation model NCSOIL (Molina et al. 1983).

The MIT model consists of seven organic matter pools: four litter pools and three SOM pools. The C litter pools defined in the model are: (1) metabolic, easily decomposable compounds; (2) cellulosic compounds (not encrusted in lignin); (3) ligno-cellulosic compounds; and (4) a woody litter pool (Figure 2a). Each litter pool is divided into a surface and sub-surface component. Decomposing foliar and fine root litter is characterised in the





**Figure 1.** Pools and fluxes of C and N in the G'DAY model.  $P_g$  = gross photosynthesis;  $N_{in}$  = N input through deposition and/or biological fixation;  $N_{loss}$  = N output through leaching and/or gaseous losses (modified from Comins and McMurtrie 1993).

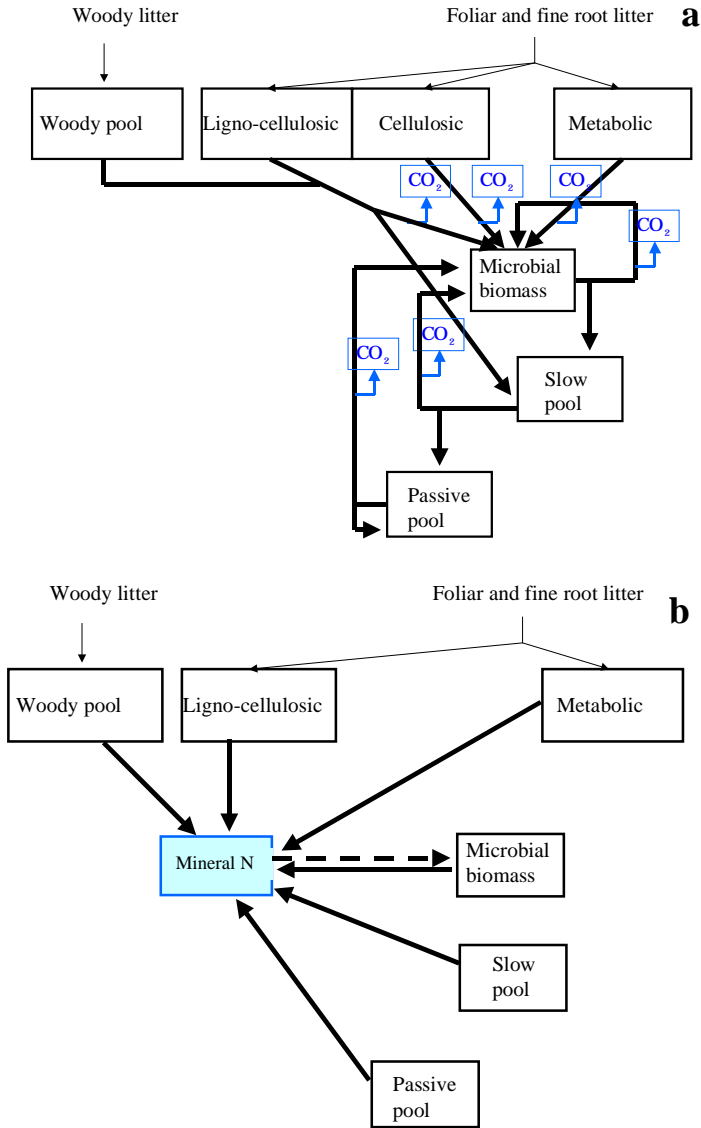
model by the first three litter pools, while branch, stem and coarse root litter enter the woody litter pool. The SOM pools in the model are: (1) a microbial biomass pool; (2) a 'slow' pool of humified products of microbial debris and lignin; and (3) a 'passive' or stable humus pool. The microbial biomass is split up into two components: (1) a labile component; and (2) a stable, more recalcitrant component (Molina et al. 1983).

Nitrogen is associated with each C pool (except the cellulosic pool). The N pools (including mineral N) and mineralisation-immobilisation fluxes are shown in Figure 2b. It is further assumed that the N:C ratio of the ligno-cellulosic pool equals the N:C ratio of the overall litter, the remainder of N in the litter is allocated to the metabolic pool.

The model assumes that from all litter and SOM pools a fraction ( $Y_i$ , the microbial efficiency factor) enters the microbial biomass pool. The fraction of the C leaving the microbial biomass pool, that re-enters this pool, simulates microbial succession. Another fraction from the microbial biomass is transferred to the slow pool and simulates the synthesis of hydrolysable humified microbial products. The fraction of decomposed C from the slow pool, which is incorporated in the passive pool, simulates the stabilisation of microbial products. Fractions not incorporated in SOM pools are lost as  $CO_2$ . The model further assumes that lignin (in the ligno-cellulosic pools) is not used for microbial growth and is transferred directly into the slow pool, when ligno-cellulosic material is decomposed. The C decomposition rate of each litter and SOM pool is calculated according to first-order rate kinetics:

$$D_i = k_i * S_i * r_w * r_T * r_L * r_N$$

where:  $D_i$  is decomposition rate of pool  $i$ ,  $S_i$  is substrate C in litter or SOM pool  $i$ ,  $k_i$  is the intrinsic decomposition rate constant of pool  $i$ , and  $r_w$ ,  $r_T$ ,  $r_L$  and  $r_N$  are rate modifiers representing, respectively, effects of moisture, temperature, lignin and nitrogen.



**Figure 2.** (a) C flow diagram and (b) N mineralisation (solid lines) and immobilisation (dotted lines) for the modified decomposition model of G'DAY.

The moisture factor is calculated as a function of cumulative potential soil evaporation for surface litter, and as a function of moisture content of the surface soil in case of belowground litter and SOM. The response function of Kirschbaum (2000) was used to describe the effect of temperature on the decomposition rate of each of the C pools. The decomposition rate of structural compounds is further modified as a function of its lignin content (see Parton et al. 1987), with lower decomposition rates at higher lignin contents. Decomposition flows in the model are also controlled by availability of inorganic N, based on the assumption that

microbial growth is limited, when N is limiting. The rate modifier  $r_N$  is determined by balancing N availability against N needs of the decomposing microbial biomass (see below).

The model assumes that all organic N in substrate passes first through the inorganic N component before microorganisms can assimilate it and that there is thus no direct assimilation of organic N. The change of N in litter or SOM pools is given by the difference between N input (calculated from C flux multiplied by the N:C ratio of the donor pool) and N output. Inorganic N availability is further determined by N inputs from biological fixation, fertilisation or atmospheric deposition and by N outputs via plant uptake and losses through gaseous emission or leaching.

Microbial growth incorporates N from the soil inorganic N pool based on the assumed N:C ratio of the microbial biomass ( $n_{bio}$ ). The rate of gross immobilisation by microbial biomass ( $IM_{bio}$ ) is:

$$IM_{bio} = \sum Y_i * D_i * v_{bio}$$

where:  $Y_i$  is the microbial efficiency factor for litter or SOM pool  $i$ .

The N:C ratio of the decomposing microbial biomass is a function of the metabolic fraction of the incoming litter, with higher values for litter with a larger fraction of metabolic compounds. Following Parton et al. (1993), the N:C ratio of newly formed slow SOM is simply set to the N:C ratio of new microbial biomass plus a constant increment. The N:C ratio of newly formed passive SOM is set equal to the N:C ratio of the microbial pool. By imposing a N:C ratio for new slow SOM, the transfer of lignin to the slow pool during decomposition of the ligno-cellulosic pool results in inorganic N being immobilised in the slow pool to sustain its N:C ratio. This simulates the abiotic or chemical incorporation of inorganic N in humic materials.

Balancing microbial N demand or immobilisation ( $IM_{bio}$ ) against inorganic N availability (INA) will determine whether the system is N limited. INA is represented as inorganic N content multiplied by the rate of microbial N uptake. In the model, we differentiate between above- and belowground N cycling in calculating this balance. We assume that all inorganic N is available for microorganisms decomposing below-ground litter, whereas only a fraction of the actual soil inorganic N is available for microorganisms decomposing above-ground litter.

If  $INA \geq IM_{bio}$ , indicating no N limitation, microbial N immobilisation occurs at its potential rate and the value of  $r_N$  is set to 1. If in contrast,  $INA < IM_{bio}$ , indicating inorganic N is insufficient to meet microbial N demand, microbial N immobilisation is limited by N and equals the total amount of N available. Decomposition of each substrate is reduced accordingly with:

$$r_N = \frac{INA}{IM_{bio}}$$

In this case, all C fluxes are balanced proportionally to the N fluxes and N availability drives the C decomposition rates.

The key features of the present decomposition model (MIT) that distinguish it from CENTURY (e.g. as used in McMurtrie et al. 2001) are: (1) N immobilisation occurs only via the active (microbial) organic matter pool; (2) microbial biomass succession is simulated by a C flow from the active pool feeding back into itself; and (3) the C:N ratio of the active pool is a function of litter quality.

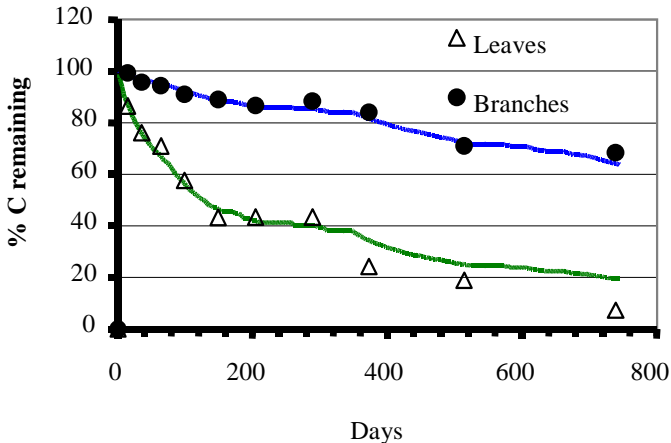
## Model Parameterisation

### MIT sub-model

The experimental data used in this study to parameterise the decomposition (or MIT) model were obtained from a litterbag experiment on decomposition of *E. globulus* slash residues under field conditions (Shammas 1999). The dataset consists of mass loss and N content data from decomposing slash leaves and branches (1 cm and 2 cm diameter) over a period of 2 years at Manjimup (34°20'S, 116°00'E; 1023 mm rainfall per annum) in southwestern Australia.

The initial distribution of plant residue C over the litter C pools was based on proximate fractionation analysis as described in Allen et al. (1974). The intrinsic decomposition rate constants of the various litter pools were derived from the literature (Van Veen and Paul 1981). The critical cumulative soil evaporation calculated between two rainfall events at which decomposition of surface litter ceases, was determined by tuning the model to the observed C loss data from decomposing leaf slash residues. The decomposition rate of the branch woody litter pool (10 yr<sup>-1</sup>) was obtained by fitting the model to observed C loss data of decomposing *E. globulus* branches. The N:C ratio of the microbial biomass pool is arbitrarily set to decrease linearly from 1/6 to 1/20, as the metabolic fraction of the incoming litter decreases from 0.5 to 0. The C:N ratio of slow SOM was set equal to the C:N ratio of microbial biomass plus 6. The external N input was set to 4 kg N ha<sup>-1</sup>yr<sup>-1</sup>.

As an example, the observed and simulated C loss data of decomposing leaf and branch (1 cm) slash residues are shown in Figure 3. The N dynamics in decomposing leaf and woody slash residues were contrasting, with leaves exhibiting net N mineralisation, and branches showing net N immobilisation. These different patterns were well captured by the model simulations (data not shown).



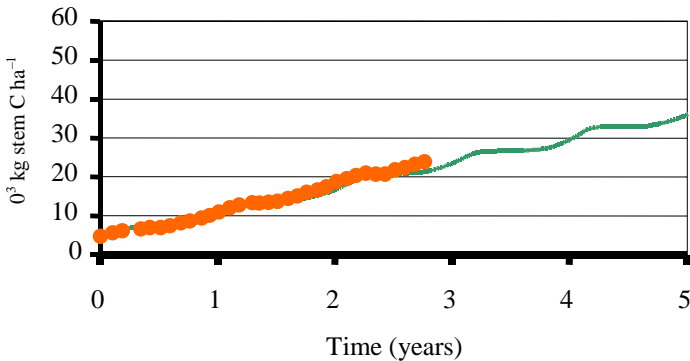
**Figure 3.** Observed (points) and simulated (line) C decay of *E. globulus* leaf and branch (1 cm) slash residues.

### G'DAY model

The G'DAY model was parameterised for *E. globulus* using data from a first-rotation plantation in Mumballup (33°33'S, 116°4'E; 950 mm) in southwestern Australia. This site was selected because the dataset provides both tree growth and N mineralisation data

(Hingston and Galbreith 1998; O'Connell and Rance 1999). The soil at the site is a shallow Xanthic Ferralsol (FAO classification) with a sandy loam texture at the surface. *E. globulus* was planted in 1988 and in 1990 the stand had a stocking of 1250 stems  $\text{ha}^{-1}$  with a basal area of 5.6  $\text{m}^2 \text{ha}^{-1}$ .

Cumulative stem growth, simulated over a 3 year period starting from September 1990, is illustrated in Figure 4. For these calculations we used a PAR utilisation efficiency ( $e_p$ ) of 2.7 g dry weight (dw)  $\text{MJ}^{-1}$ . To match simulated values with stem biomass values estimated from monthly measurements of stem diameter and tree height, allocation coefficients for the various tree components were set as shown in Table 1. Stem growth showed a seasonal pattern with slow growth rates in summer caused by water limitation.



**Figure 4.** Measured (points) and simulated (line) stem growth of a *E. globulus* stand (year 0 on x-axis corresponds to September 1990, when the stand was 2 years old).

**Table 1.** Carbon allocation coefficients from model parameterisation.

Tree component	Allocation coefficient (%)
Stem	38
Branches	25
Leaves	10
Fine roots	27

Experimental results indicate that leaf litter production was variable in time without any noticeable pattern (Hingston F.J. unpublished data). Based on these data, in the model we assumed a constant leaf litterfall rate of 0.3  $\text{yr}^{-1}$ . For branches a constant of 0.03  $\text{yr}^{-1}$  was derived from analogous measurements (Hingston F.J. unpublished data). The litter production rate for fine roots was set at 2.0  $\text{yr}^{-1}$  based on data from Fabião et al. (1985).

The N:C ratios of the various tree biomass pools were derived from measurements on the site or, if measured data were not available, from data in the literature. A crucial parameter for soil N mineralisation fluxes is N retranslocation from senescing leaves and fine roots. Nitrogen retranslocation was on average 42%, as determined from the difference between N concentration in green foliage and litter measured on the site between September 1992 and June 1993 (Hingston F.J. unpublished data). Exact information on nutrient retranslocation from fine roots is lacking. In the model, we set root N retranslocation to 10% (Nambiar 1987).

We compared simulated N mineralisation with measurements made in the top 20 cm soil layer over the period from 1992 to 1994 (Table 2- O'Connell and Rance 1999). We assumed that decomposition rates of belowground litter or SOM decrease linearly with soil moisture below a relative moisture value of 0.7, and are 0 below a threshold value of 0.1 (Paul K. and O'Connell A. M. unpublished results). The model was able to reproduce the N mineralisation pattern at the site (data not shown). Nitrogen mineralisation rates were highest during late winter and spring when soil moisture was not in deficit and soil temperature was rising. Rates were low or zero in summer, when the topsoil dried below wilting point. In calculating the amounts of net N mineralised (on an annual basis, see Table 2) the initial soil C and N status are important, as they determine the contribution of the SOM pools to the net N mineralisation fluxes. Direct determination of the initial size of the slow and passive SOM pools is problematic, since these pools are defined on a conceptual basis. The initial level of total SOM was set at 65 t C ha<sup>-1</sup>, based on measurements in the top 30 cm layer (Hingston F.J. unpublished data). The initial level of microbial biomass was set at 1% of total SOM (personal communication, D. Mendham) and the amount of slow C was set at 15%. With these initial parameter values, annual net N mineralisation was reasonably well predicted over 3 years by the model (Table 2). Overestimation by the model is reasonable, since measured data represent only the net N mineralisation in the top 20 cm soil layer (O'Connell and Rance 1999).

**Table 2.** Observed and simulated annual N mineralisation in a 4-yr-old (in 1992) *E. globulus* stand.

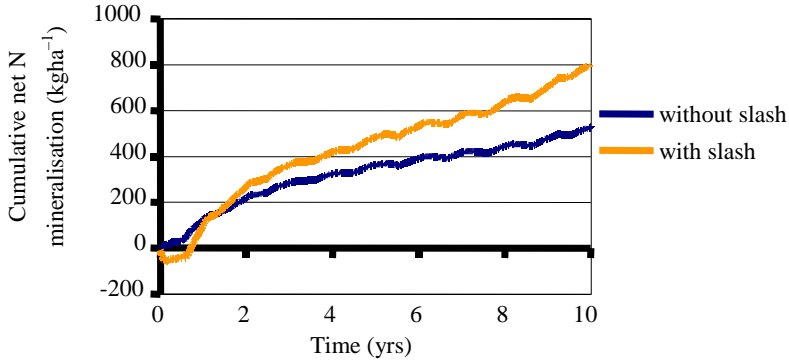
Year	N mineralisation (kg N ha <sup>-1</sup> yr <sup>-1</sup> )	
	measured	simulated
1992	72	78
1993	81	107
1994	81	86

### Impact of harvest residues on N mineralisation

Using the above model, we simulated the effect of slash residue retention on site productivity in terms of N mineralisation. Results from a study by O'Connell et al. (2000) indicate that slash loads following harvest of first-rotation *E. globulus* stands vary largely with site productivity. Slash loads were 13 and 21 t dw ha<sup>-1</sup> of leaves, and 18 and 30 t dw ha<sup>-1</sup> of woody material, respectively on a fertile red earth soil and a less fertile grey sandy site. The corresponding amounts of N in the slash residues ranged from 220 to 350 kg N ha<sup>-1</sup>, which are considerable amounts in relation to the stores in the surface soil (O'Connell et al. 2000).

In the following example we considered two cases: (1) no slash residues retained on the site; and (2) leaves and small woody residues (branches) retained at load levels of 16 and 20 t dw ha<sup>-1</sup>, respectively. The C:N ratio of slash leaves and branches were set to 40 and 150, respectively.

Results from model runs with and without slash retention are given in Figure 5. We simulated the growth and N cycling of a *E. globulus* stand during 1 rotation (10 years) with the parameter values as obtained from the model calibration (see above). With these parameter values, retention of slash residues led to a net N immobilisation during the first year of the rotation (Figure 5). This phase of N immobilisation was followed in subsequent years by an increased N mineralisation compared to the situation where slash residues were removed. At the end of the rotation (10 years), the model predicted that about 250 kg ha<sup>-1</sup> of extra N was mineralised in the scenario with slash retention compared to slash removal.



**Figure 5.** Simulated cumulative net N mineralisation in an *E. globulus* stand with and without retention of slash residues.

The impact of retention of harvest residues on site productivity is currently being experimentally investigated in two second rotation *E. globulus* plantation in southwestern Australia (O'Connell et al. 2000). Results show that annual N mineralisation rates during the third and fourth years after harvest on both sites are significantly higher when harvest residues are retained on the site.

## Conclusions

We have described the structure and parameterisation of a detailed, process based C and N cycling model (G'DAY) for *E. globulus* stands and investigated the impact of harvest residues on N mineralisation. Model simulations suggest that retention of residues will favour enhanced soil N supply for the next rotation. The risk of N stress due to N immobilisation appears to occur only during the first year of the second rotation.

The present formulation of the model is based on the current knowledge of the processes underlying tree growth, decomposition and N cycling in eucalypt plantations. The results of the model predictions on slash residue management have to be tested against experimental data. This will allow further improvements in model structure and eventually emerge in better quantitative understanding of underlying processes of N cycling.

## Acknowledgements

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# **The Use of Management-Oriented Growth and Yield Models to Assess and Model Forest Wood Sustainability. A Case Study for Eucalyptus Plantations in Portugal**

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## **Abstract**

This paper demonstrates, using a large eucalyptus plantation as a case study, how management-oriented growth and yield models may be used within a prototype decision support system to assess forest wood sustainability. The growth model used is GLOBULUS, which predicts whole stand development for different combinations of region, initial stand density and site index. GLOBULUS also includes a series of equations for new stand initialisation (by planting or coppice), essential for long term prediction of wood availability. After describing GLOBULUS, we illustrate its application within the framework of the decision support system developed at CEF, which simulates various management alternatives in a eucalyptus forest area. The system is used to find the optimal stand management prescription that maximizes net present value subject to pulpwood even flow constraints.

*Keywords: Growth and yield model, decision support system, wood sustainability, Eucalyptus globulus, Portugal*

## **1. Introduction**

Decision support systems for forest management are among the most important applications of growth and yield models. The objective of such systems is to help in the selection of management alternatives that meet certain user-defined goals and criteria. Management alternatives (prescriptions) consist of schedules of activities over the planning horizon of a forest stand. Davis and Johnson (1987) identified three essential elements of a management alternative: (i) the classification of the forested landscape into homogeneous management units (stands), (ii) the description of the management activity by which the vegetation and

other resources will be manipulated or disturbed to achieve desired outputs, and (iii) a quantitative growth and yield projection that describes stand development under each management activity.

This paper demonstrates the use of a growth and yield model developed for eucalyptus plantations in Portugal– the GLOBULUS 2.1 model – within the framework of a decision support system that was developed at CEF (Centro de Estudos Florestais). The growth model and the decision support system are briefly described and then applied together to simulate management alternatives for a large eucalyptus forest area in the South of Portugal. This forest area represents an interesting case study because it is a large area (4510 ha) without any harvesting during the previous 3 years. Therefore, a ‘financial optimum’ solution would imply the harvesting of a large area in the first years (all stands older than optimum rotation age), thus producing an uneven flow of wood. The decision support system is used in order to find prescriptions that lead to a more reasonable solution, both in terms of present net value and even flow of wood.

## 2. The GLOBULUS 2.1 Model

The GLOBULUS 2.1 model is the present version of a growth and yield model that systematises all the growth and yield information available in Portugal for eucalyptus. It represents the product of a co-operative research project between industry and universities, initiated in 1995, with three main objectives:

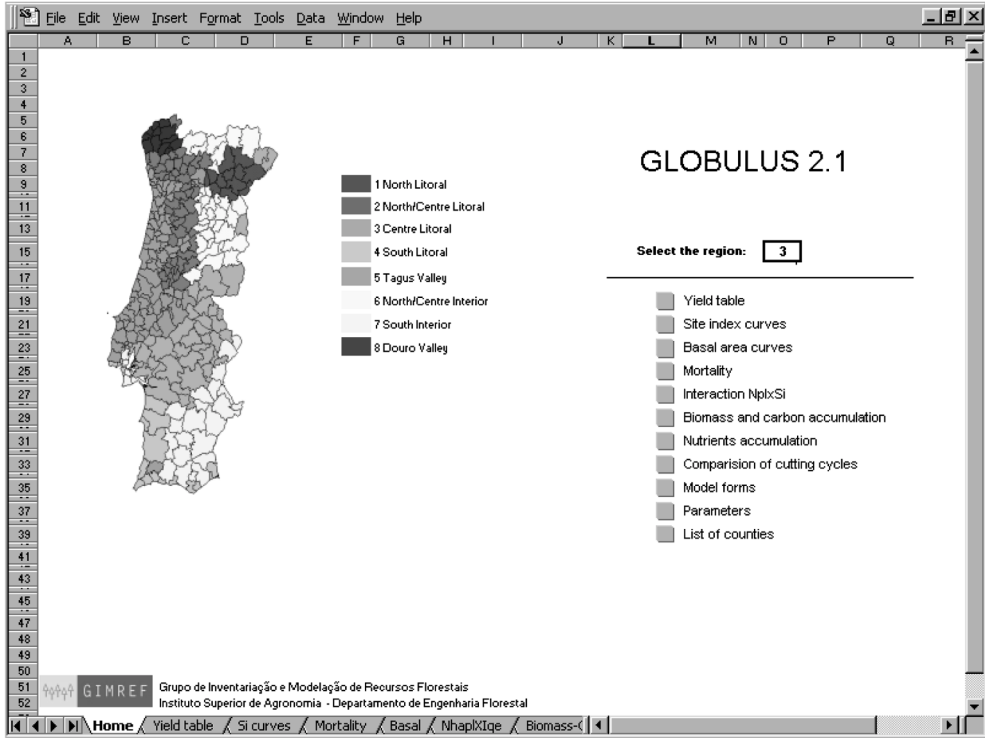
1. to combine all growth data available in Portugal for eucalyptus;
2. using these data, to develop the ‘best’ growth and yield model for eucalyptus plantations in Portugal, applicable to the whole country, but reparameterised, if needed, for different regions; the model should be aimed at: operational planning at stand or company level; strategic planning at country or regional level (long-term wood availability for industry); the definition of forest policy measures; and the definition and monitoring of sustainability criteria;
3. to identify the need for additional data and trials.

Previous versions of the model have already been described (Tomé et al. 1998a; Tomé 1999). Details of the present version of the model are given in Tomé et al. (2001).

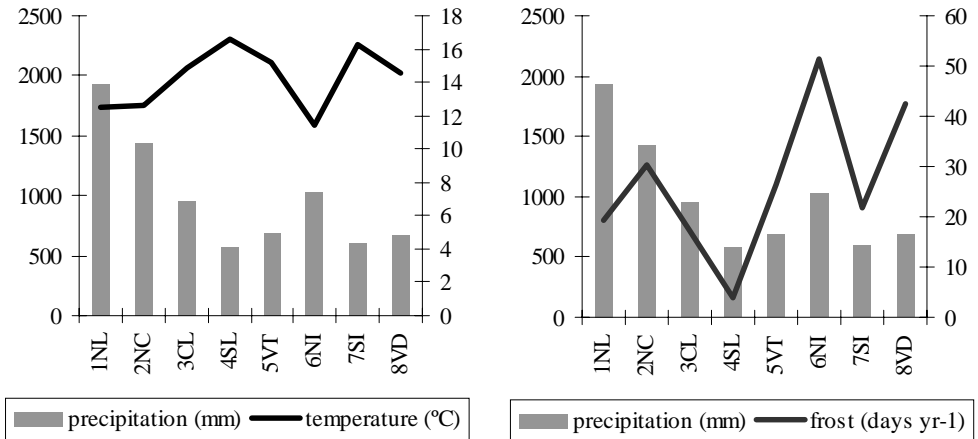
### 2.1 Model regionalisation

The need to reparameterise the model for different regions was examined using a purpose-designed climatic classification of Portugal (Ribeiro and Tomé 2000) based on climatic information in digitised maps. Because not every plot included in the database was geo-referenced, the unit of analysis was the county, an administrative division of the country (Figure 1). The climate of each of the 275 counties in the country was characterised with the aid of a GIS. Principal component analysis followed by hierarchical classification was then used to divide the country into 8 climatic regions (Figure 1).

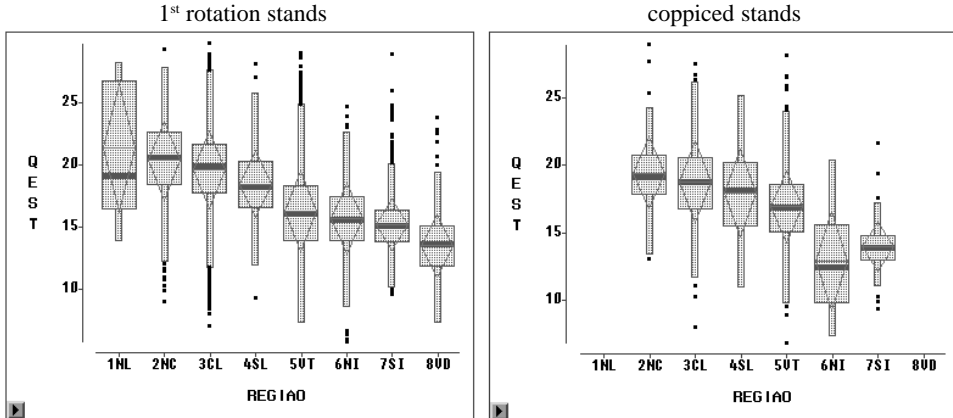
Figures 2 and 3 characterise the eight regions in terms of climate and productivity, respectively. There is a strong impact of precipitation on the productivity in each region. The number of frost days, however, can offset water availability, thus inverting the yield ranking for some regions. A set of 7 dummy variables was defined, in order to assess the need to parameterise each model component for the different regions (Tomé et al. 2001).



**Figure 1.** Climatic classification of the Portuguese territory (*Home* page of the EXCEL interface for the GLOBULUS 2.1 model).



**Figure 2.** Summary of the mean annual climate in each one of the eight climatic regions. Regions are ranked according to mean site index (Figure 3).



**Figure 3.** Box-plots for productivity, represented by site index (base age 10), in each one of the eight climatic regions. The central line in each box indicates the mean site index while the bold line indicates the 50% percentile and the edges the 25–75% percentiles. Only one point per permanent plot is represented.

## 2.2 The database

The model was developed using a large amount of information on the growth of eucalyptus stands in Portugal from several sources – permanent plots, experimental trials, and continuous forest inventory by pulp companies – organised into a common relational database. At present the database includes 31 416 measurements from 11 371 growth periods. The coverage of data on the control variables used in GLOBULUS – age, site index, initial stand density, region and rotation – is far from complete (Tomé et al. 2001) because (i) experimental trials do not cover the full range of other control variables, (ii) some regions and old ages are poorly represented, and (iii) coppiced stands are poorly represented in some regions (sometimes not at all).

## 2.3 Model structure

The overall structure of GLOBULUS 2.1 is shown in Figure 4. The model includes:

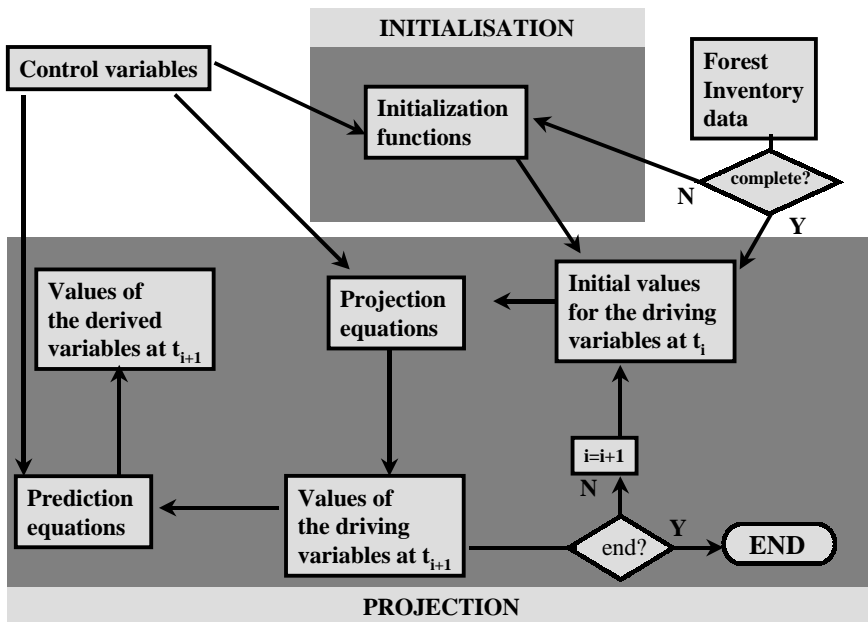
- state variables, that define the state of the stand over time; these are divided into driving variables (directly predicted from a growth function) and derived variables (indirectly predicted from allometric or other equations);
- control or external variables, that control the development of the state variables; control variables can be environmental, cultural or intrinsic to the stand.

Table 1 lists all variables currently included in the model.

GLOBULUS 2.1 has two main modules (Figure 4): initialisation and projection. The projection module includes a system of compatible functions to predict the growth of each driving variable as a function of its starting value as well as the control variables and other driving variables (through growth functions formulated as first order non-linear difference equations). The derived variables are predicted as a function of state and control variables. The initialisation module predicts each stand variable as a function of the control variables that characterise the stand. This module is essential for the decision support system because it

**Table 1.** Variables currently included in the GLOBULUS 2.1 model.

Control variables	Driving variables	Derived variables
Environmental: Site index (base age 10) Climatic region	Dominant height  Number of trees ha <sup>-1</sup>	Merchantable volumes (over and under bark)
Cultural: Initial stand density (1 <sup>st</sup> rotation) Stand density after thinning (coppice) Rotation age	Stand basal area Total volume (over and under bark)	Total biomass (aboveground) Biomass per plant component: branches, leaves, wood, bark
Stand: Rotation Age		Carbon stock per plant component



**Figure 4.** Structure of the GLOBULUS 2.1 model.

allows for new stand initialisation (by planting or coppice). Site index in the first coppice is similar to the site index of the respective first rotation, slightly higher in very good sites and smaller in poor sites (Tomé and Soares 1998). Site index (base age 10) in the second cutting cycle is modelled as a region-specific function of site index in the first cycle.

Further details of the model functions, and the statistical methodologies used to parameterise them, are given in Tomé et al. (2001). The EXCEL version of the model, described in section 2.4, is available from the GIMREF internet site (<http://www.isa.utl.pt/DEF/CEF/GIMREF>).

## 2.4 The EXCEL interface

GLOBULUS 2.1 is implemented in the decision support system described in the next section. It is also available in an EXCEL workbook that can be used for qualitative evaluation of the model by users and for teaching and demonstration purposes. Such an interface is very useful to users of the decision support system, so that they may understand how the model functions and become confident (or not) about its use.

Figure 1 shows the initialisation sheet of the EXCEL interface of GLOBULUS 2.1. In this sheet the user selects the region in which the object stand is located. From there it is possible to access a series of sheets containing relevant information or model outputs:

- Yield table sheet (Figure 5), in which the site index and initial stand density are specified. The user may estimate the site index, provided that he knows a pair (age, dominant height). It is also possible to choose the volume type for the yield table: i) total volume or merchantable volume, by choosing the top diameter (0, for total volume); ii) over or under bark volume, indicated by the dummy variable *bark* (0 – without bark; 1 – with bark).
- Site index curves (Figure 6) over the range of site indices that may occur for both cutting cycles in the chosen region.
- Basal area curves and mortality functions for the first and second cutting cycles for different combinations of initial stand density and site index.
- Plots of volume growth, mean annual and current annual increments, for 9 combinations of initial stand density and site index, for the first and second cutting cycles (Figure 7). The model reproduces well the combined effect of site quality and initial stand density on stand age at maximum mean annual increment as observed in spacing trials (Ribeiro et al. 1997).
- Biomass, carbon and nutrient accumulation by plant component: wood, bark, leaves and branches.
- Comparison of first rotation stands with the respective first coppice (in the present version other coppices are assumed to be equal to the first).
- Model functions and parameter estimates for each region.
- List of counties as an aid to selecting the appropriate region.

## 3. The Decision Support System

The decision support system developed at CEF for multifunctional forest ecosystem management is based on a set of integrated modules and sub-modules (Figure 8). The primary target level of the system is the management area – a forest project impact area – which is subdivided, according to ecological and productive criteria, into several forest stands.

Miragaia et al. (1996; 1998; 1999) described the first module of the decision support system, a forest management information system – *inFlor* – that stores all the data that may be relevant for forest resource management according to various goals and criteria. The Portuguese Forest Service, the Nature Conservation Agency, forest industry, forest researchers, and environmental groups provided diverse information on the data requirements for forest ecosystem management in Portugal, including the identification of entities, attributes and the relationships between them.

In addition to the management area and management units (stands), the system also considers the entity plot that represents a sampling unit for field data gathering. Other entities include ecological features of forest stands (e.g. soil characteristics, occupancy, development and water resources), events that may occur in a forest stand (e.g. site preparation, thinning, clearcut) and plot data (e.g. tree measurements and dbh distributions) as well as technical and economic data.

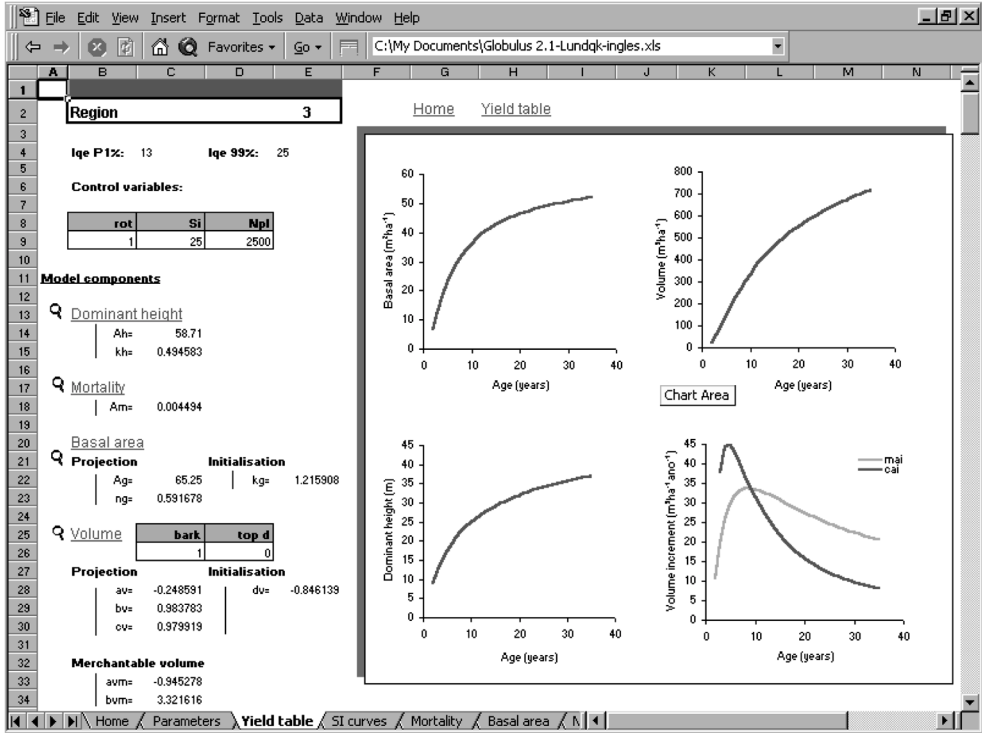


Figure 5. Yield table sheet of the EXCEL interface for the GLOBULUS 2.1 model.

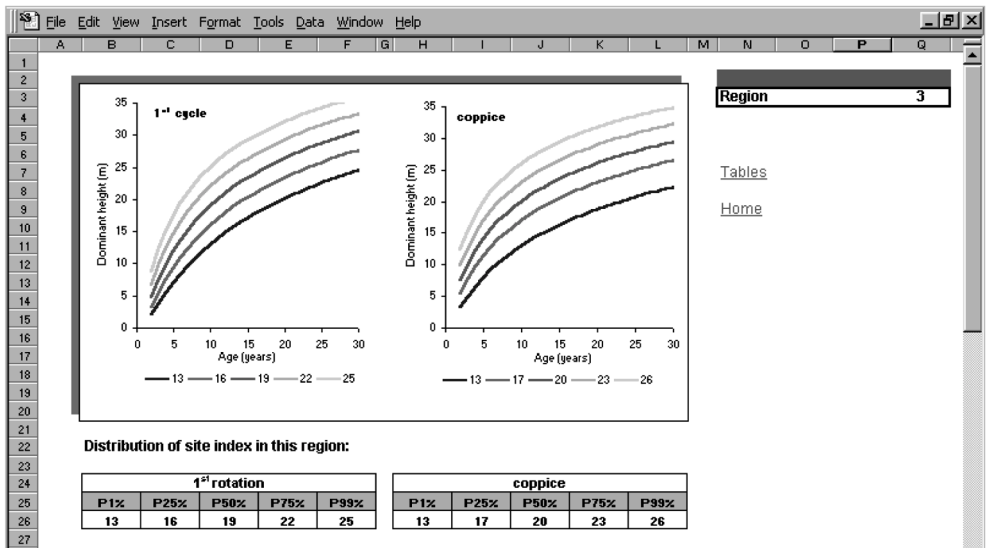


Figure 6. Site index curves sheet of the EXCEL interface for the GLOBULUS 2.1 model.

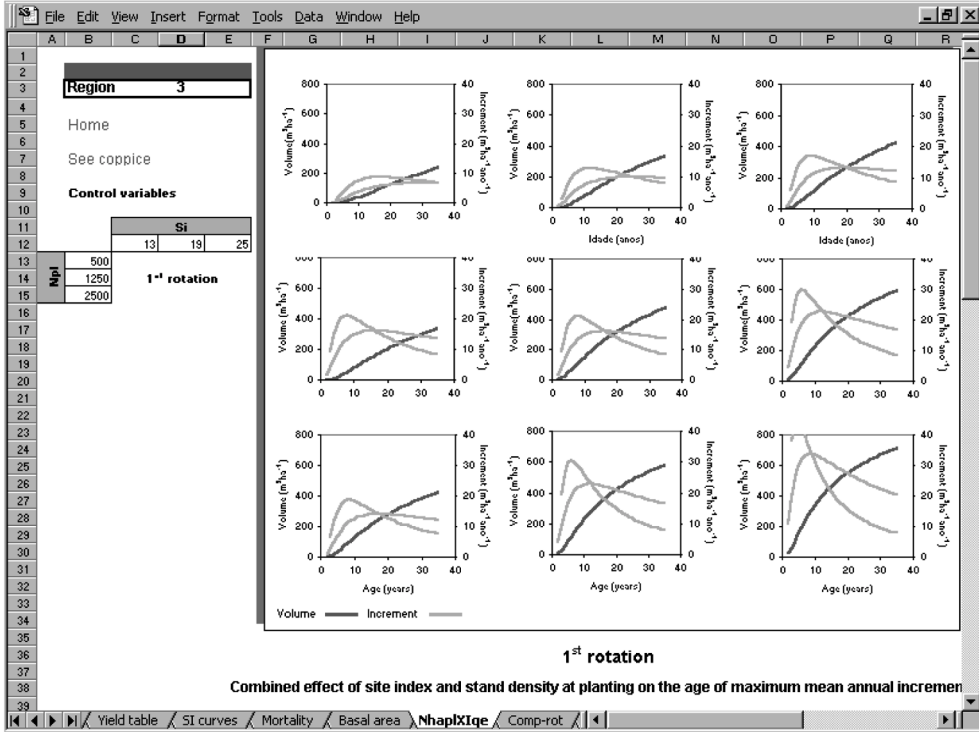


Figure 7. Interaction between number of trees at planting and site index as shown in the *Interaction NplxSi* sheet of the EXCEL interface for the GLOBULUS 2.1 model.

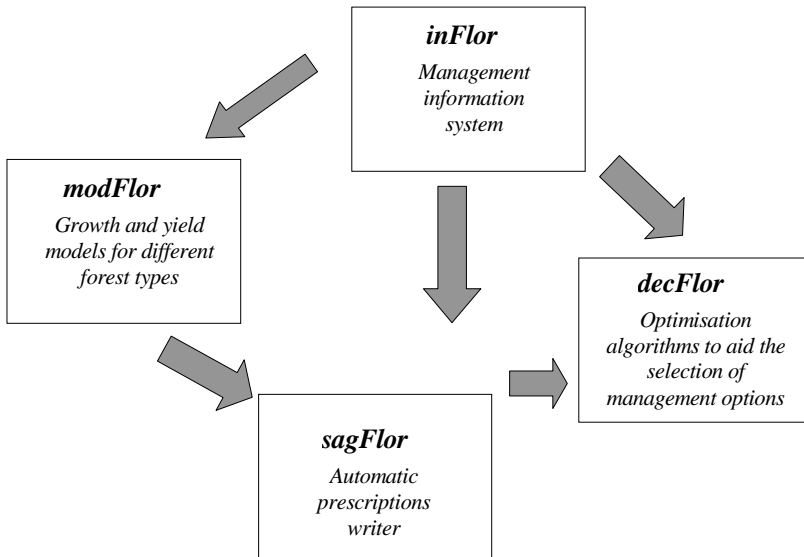


Figure 8. Structure of the CEF decision support system.



Forest management problems generally involve a large number of management units and an extended planning horizon. Thus, the automation of the process of describing management activity schedules in each unit and of quantifying resulting product flows is crucial for decision analysis. For example, currently, *inFlor* stores data for more than ten thousand management units. If 20 prescriptions were considered for each unit over a planning horizon extending over 100 years, the number of management options would reach 200 thousand. The quantification of the corresponding product flows would be extremely inefficient if the overall process were not automated. For that purpose, the decision support system second module – *sagFlor* – automates the process of generating forest management prescriptions for management units stored in the information system *inFlor* (Borges and Falcão 1999; Falcão et al. 1999). The module *sagFlor* integrates a set of forest growth and yield models (from a third module *modFlor*): GLOBULUS (Tomé et al. 2001, described in this paper) for eucalyptus; the maritime pine yield table of Oliveira (1985) and the DUNAS model (Falcão 1999) for maritime pine; and SUBER (Tomé et al. 1998b; 1999) for cork oak.

A fourth module, *decFlor*, allows the selection of models (*via sagFlor*) for solving a wide range of forest ecosystem management problems, such as the optimisation of timber net present value, either unconstrained (financial optimum) or subject to various types of constraints (e.g. even flow, adjacency, minimum harvest patch size) in various combinations. Details of the optimisation algorithms are given in Falcão and Borges (in press).

#### 4. Modelling Forest Wood Sustainability in a Large Eucalyptus Area

Here we illustrate the use of GLOBULUS within the decision support system for a large eucalyptus plantation – Serra de Ossa – located in the South Interior region of Portugal (region 7 of the GLOBULUS model). The plantation has 4510 ha and includes 660 management units – stands – with areas between 1 and 106 ha. This area has not been harvested during the previous 3 years, and therefore has a large percentage of management units older than the optimum rotation age. The objective is to find a set of prescriptions that guarantees a more or less even flow of wood, minimising the loss in present net value. The *sagFlor* interface is used to find this set of prescriptions.

The sheet for selecting management units (Figure 9) offers a set of options that enable the user to specify selection criteria, e.g. *Inventário Posterior a* (date of last forest inventory), *Idade Superior a* (average age greater than), *Área de Gestão* (Management Area), *Utilização* (major use) or *Espécies* (forest species). Figure 9 illustrates the selection of all the eucalyptus in Serra de Ossa. The buttons to the lower right (Figure 9) lead to a map and table of the selected management units.

The definition of management options available for each forest type is interactive. For that purpose, sheets were designed for defining management options for each forest type. For example, Figure 10 displays all options available for eucalyptus management (*Parâmetros dos modelos de silvicultura*). These include spacing (*compassos de instalação*), cutting cycle age (*Idades de corte*), average number of shoots per stool (*Número de varas por toíça*) and number of cutting cycles in a rotation (*cortes de talhadia*).

After defining the management options, the grid in the lower left corner (Figure 10) will then display all silvicultural models that were generated according to the user-defined parameters. In the present case study *sagFlor* automated the generation of 512 alternative models that differ only in cutting cycle age. The grid displays the twenty-first such model (Figure 10).

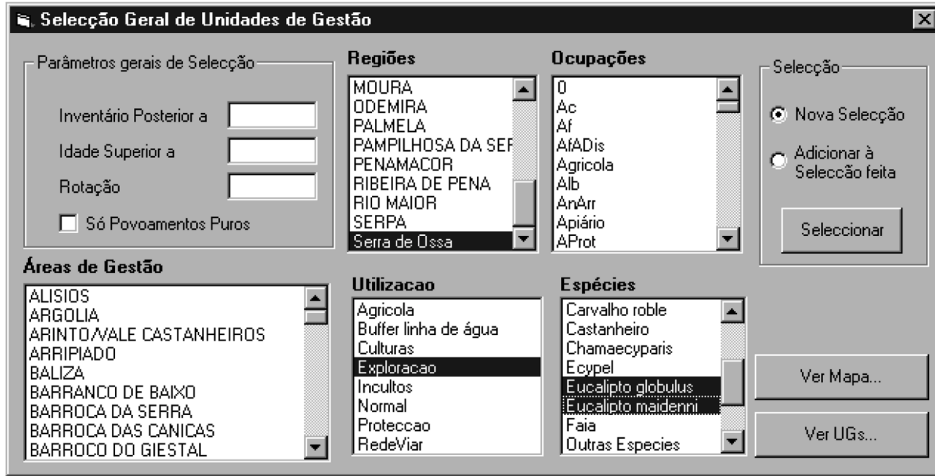


Figure 9. *sagFlor* sheet for selecting management units.

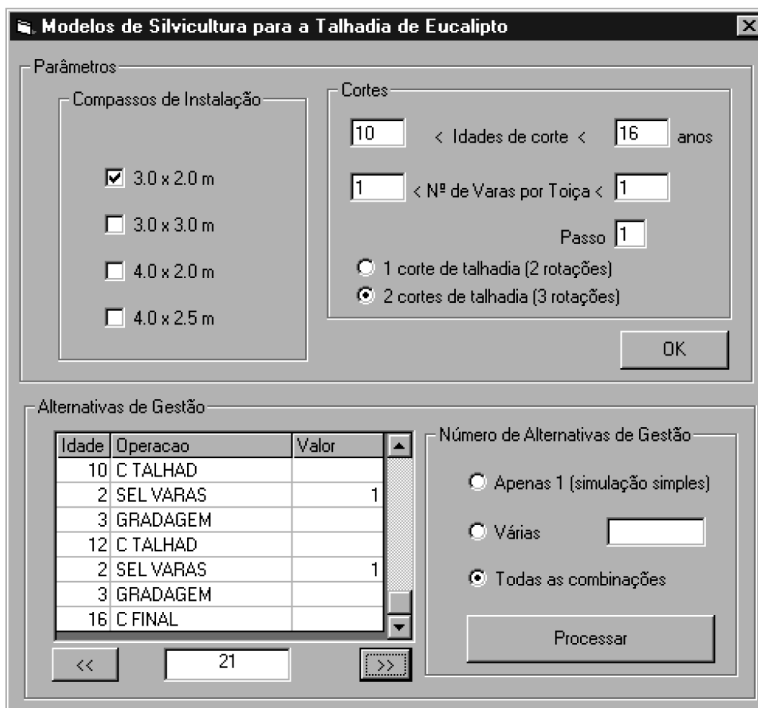


Figure 10. *sagFlor* sheet for defining management alternatives in eucalyptus stands

At this point, the user may decide to analyse forest management planning based on stand-level technical decisions, or to generate information for management alternatives. In the first case, the user may simulate one unique management alternative for each unit (Falcão et al. 1999). For that purpose, after selecting one silviculture model, the user may check the option button (*Apenas 1 (simulação simples)*) in the frame *Número de Alternativas de Gestão* (number of management alternatives) in the lower right corner (Figure 10). In the second

case, the user may choose between two options; *sagFlor* may be used either to generate a limited number of management alternatives in each unit based on Monte Carlo simulation, or to generate all feasible prescriptions (Falcão et al. 1999). In the example shown, the user checked the option button *Todas as combinações* (all feasible prescriptions). Thus *sagFlor* will generate all feasible rotation combinations to define the management alternatives in the Serra de Ossa eucalyptus units. The total number of management alternatives will depend on the planning horizon defined by the user.

For each particular problem the user may use the system several times, changing the parameters used to define the management alternatives allowed for each management unit (Figure 10). In this way the system may also help the user in defining these parameters. Considering the Serra de Ossa case study, suppose that an annual wood consumption of  $40 \times 10^3 \text{ m}^3$  is set as a harvesting goal. Let us set the initial spacing for new stands to  $3 \times 2 \text{ m}$ , the number of shoots per stool in the range 2–3 and set a maximum of 3 cutting cycles.

As an illustration, *sagFlor* was used to simulate management alternatives under three silviculture scenarios that differ in the range of allowable cutting cycle length (Table 2). Two decision algorithms were considered, both with the objective of maximizing total net present value. In the first, no constraints were considered, and a simple computational procedure was used to select the optimum prescription for each stand (FO). In the second, pulpwood even flow constraints were applied, and a genetic algorithm was used to assign one prescription to each stand (GA).

The first algorithm (FO) led to higher net present values but annual harvests fluctuated substantially, thus deviating from the target annual pulpwood supply level of  $40\,000 \text{ m}^3$  (Table 2). The second algorithm (GA) produced solutions with slightly lower net present value, but volume flows approximated the target values (Table 2).

As expected, widening the range of allowable cutting cycle lengths leads to better solutions using either algorithm. The associated increase in management flexibility may contribute to higher financial returns, and allows for easier compliance with pulpwood even flow constraints (Table 2). With a range of cutting cycle length of 10–19 yr, a third algorithm was considered in which pulpwood flows might deviate by up to 5% from the targets. A genetic algorithm was used to approximate the solution. As expected, relaxing the flow constraints leads to an increase in net present value, approximating the value achieved under the financial optimum algorithm (FO).

**Table 2.** Results from the *sagFlor* interface for different ranges of cutting cycle length and different optimisation algorithms: financial optimum (FO), and genetic algorithms to maximise net present value subject to an even flow of wood (GA).

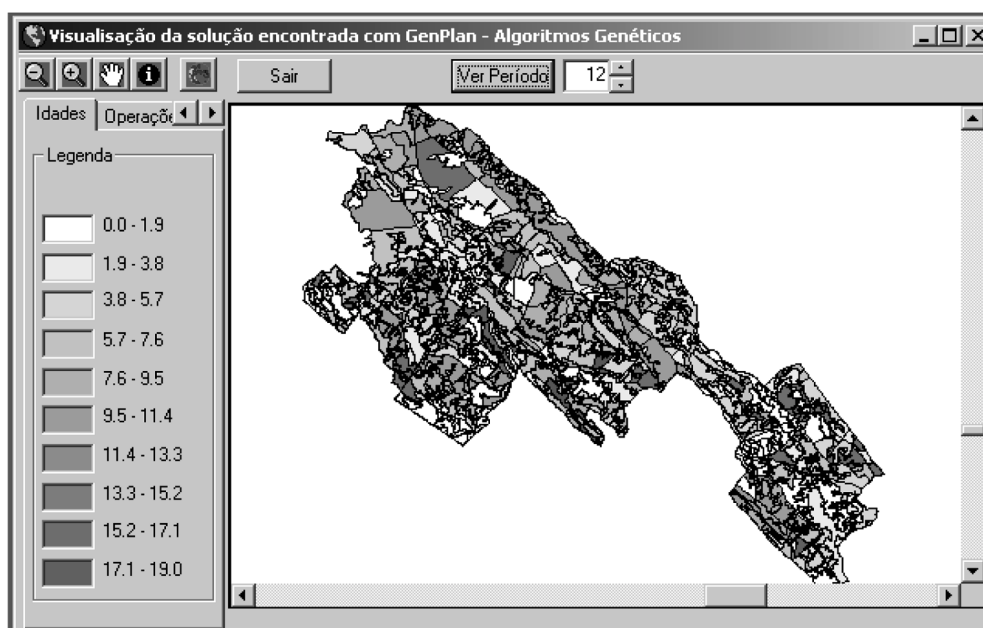
Length cutting cycle (range)	Consumption level		Optimis. Alg.	Net present value $10^9$ PTE	Annual harvest of wood ( <i>ahw</i> )		
	annual $10^3 \text{ m}^3$	Tolerance %			max*	min	d40>5 <sup>†</sup>
10–13	–	–	FO	2.496	211	0	29
10–13	40	0	GA	2.143	102	0	2
10–16	–	–	FO	2.564	166	0	29
10–16	40	0	GA	2.286	55.9	36.6	1
10–19	–	–	FO	2.610	174	0	27
10–19	40	0	GA	2.289	39.8	38.2	0
10–19	40	5	GA	2.325	41.9	38.1	0

\*maximum *ahw* always occurs in the first year; <sup>†</sup>d40= $|ahw-40|$

Because the decision support system is integrated within a geographical information system, the solutions may be visualised in map form. Figure 11 shows the age distribution in year 12 when the decision support system was used to optimise timber net present value subject to flow constraints, using simulated annealing. It is also possible to visualise the operations (thinning, harvesting) to be applied, for each period in the planning horizon, in each management unit.

## 5. Final Remarks

This paper demonstrates the use of a growth and yield model developed for eucalyptus plantations in Portugal– the GLOBULUS 2.1 model – within the framework of a decision support system that was developed at CEF (Centro de Estudos Florestais).



**Figure 11.** Map displaying the age class in each unit in year 12 of the planning horizon if the maximum net present value management alternative was selected for each unit in Serra de Ossa.

The GLOBULUS 2.1 model is the present version of a growth and yield model that systematises all the growth and yield information available in Portugal for the species. It predicts all the stand variables important in terms of pulpwood yield, taking into account the region where the stand is located and allowing for different planting densities and site indices. It is also possible to initialise new stands – both first cutting cycle and coppice – as a function of a few control variables: region, initial stand density and site index.

*sagFlor* is a computational module that efficiently reads data stored in a management information system (*inFlor*), and provides a user-friendly interface for generating forest

management prescriptions. Currently, this set of programs integrates the most recent versions of Portuguese forest species growth and yield models. Its modularity allows easy updating of existing models and inclusion of models for other forest species.

In this paper, we have demonstrated how the *sagFlor* interface may be used to select and analyse stand data organised in the management information system; to define the management options available for each forest type; to analyse forest management planning based on technical stand-level decisions; and to generate information for management models. The ability to take account of the complex spatial and temporal interactions between decisions made at the stand level is crucial for decision analysis (Falcão et al. 1999).

## Acknowledgements

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# Optimization Models for Scheduling Silvicultural Operations

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## Abstract

Among forestry activities in Chile at present, silvicultural tasks involve the greatest amount of labour instability. Due to their seasonal character and short duration, a continuous turnover of non-specialized workers is generated. To manage this situation it is necessary to predict the number of workers required to perform such tasks, considering cost and labour stability simultaneously. The objective of this study was to develop and compare the performance of several optimization models for solving this problem, based on different objectives and constraints. Our results indicate that constrained optimization models can aid the effective scheduling of silvicultural activities at an operational level, thus providing an example of how models can be used to address the socio-economic aspects of sustainable forest management.

*Keywords: operational planning, activity scheduling, labour stability, Chile*

## Introduction

This paper concerns the application of models to the socio-economic aspects of sustainable forest management. At present, the forestry sector is one of the most dynamic components of the Chilean economy. In an open economy, however, the productive factors are constantly changing, and forestry activities are becoming more competitive and less profitable. Moreover, the productive and social constraints of sustainable forest management are of increasing concern. These constraints have led forestry companies to develop new techniques for improving the return of their investments by means of increasing yields, optimizing production systems, and minimizing operational costs.

Mathematical models may be used to solve a variety of problems such as selection of management regimes, log merchandising, and transportation scheduling. To date, however, the problems analyzed by forestry companies have mainly concerned the efficient allocation of resources at a strategic (Barros and Weintraub 1982; García 1984) and tactical level (Laroze and Greber 1991; Weintraub et al. 1994). Simulation and optimization models have been used much less frequently for solving operational problems, the scheduling of silvicultural activities being a particular case in point (Muñoz and Andalaft 1991).

At present, silvicultural activities involve the greatest amount of labour instability within the Chilean forestry sector. The seasonal character and short duration of these tasks leads to a continuous turnover of non-specialized workers. One way forward would be to schedule silvicultural activities by considering cost and labour stability simultaneously; that is, by maintaining as regular a workforce as possible in order to reduce monthly labour fluctuations, without affecting operational costs. Such an approach might improve the use of forest camps, increase companies' administrative efficiency, and achieve a labour stability that could stimulate a worker-training program by the forestry contractors.

In this study, several optimization models were developed for scheduling silvicultural tasks over a one-year planning horizon. The lack of proven models of this type made it necessary to design and evaluate alternate formulations. By implementing different objective functions and constraint sets, and comparing the results in terms of cost and labour stability, we were able to generate effective management solutions at an operational level.

## Data on Silvicultural Tasks

Forestal Mininco's Department of Forest Management (Concepcion Region) provided the data for silvicultural activities. Data on 20 tree-farms representing diverse conditions were extracted from the 1995 annual plan. For each tree-farm, the data consisted of the area, cost, labour productivity, and feasibility periods for eight silvicultural interventions. A 12-month planning horizon was considered.

## Objectives and Constraints

Based on the annual program of silvicultural activities, mathematical programming techniques may be used to efficiently tailor the workforce for each task at each tree-farm, on a monthly basis. The input required consists of the periods in which it is feasible to perform the tasks, the expected labour productivity and cost, and the available budget.

Because the problem of scheduling silvicultural tasks was not clearly defined, several models were evaluated, based a combination of different objective functions and constraint sets. Thus we were able to examine different scenarios and compare the optimal solutions obtained in each case. The following *objective functions* were considered:

- *Minimization of total cost*: the solution minimizes the total cost incurred by all the silvicultural tasks at every tree-farm.
- *Minimization of total labour*: the solution minimizes the total workforce required to execute all the silvicultural tasks considered.
- *Minimization of workforce variance*: the aim here is to minimize the variance in the number of man-days hired throughout the season – a proxy for maximizing labour stability.



- *Minimization of the maximum workforce required per month (MMW)*: the aim here is to minimize the peak labour requirements in each month, and thus indirectly to reduce the total annual workforce and to standardize the workload in different periods. This objective function is complemented by restrictions that ensure a positive difference between the objective value and the number of man-days hired in each period of the season.

The following constraints were considered:

- *Annual program (A)*: this constraint, implemented in all models, ensures that the solution performs the annual program of silvicultural activities in every tree-farm.
- *Range of allowable man-days per month (R)*: this forces the solution to hire a number of man-days per month that does not exceed a maximum bound nor is lower than a minimum target, taking into account all the tasks in the different tree-farms. This range allows regulation of the workload distribution in absolute terms.
- *Workforce fluctuation (F)*: this forces the solution to maintain the relative differences in the number of man-days hired in consecutive months within a specified percentage.
- *Minimum and maximum monthly size per activity at each tree-farm (S)*: These restrictions force each silvicultural task to have a minimum areal extent (defined by a practical limit that makes its execution possible) and a maximum areal extent (set by an operational bound), in each month and at each tree-farm.
- *Continuity condition (C)*: this condition acts to reduce task interruptions, by associating a cost to the beginning and ending of each task. If the assigned costs are high enough, the continuity conditions become implicit constraints.
- *Maximum total cost (P)*: It restricts the solution not to exceed a pre-defined budget for executing the annual program of silvicultural activities.

The constraints were parameterised as follows. A maximum variation of  $\pm 10\%$  was accepted for the workforce hired in successive months. For the total workforce hired per month, the minimum (maximum) value was 5850 (7150) man-days for all the periods. These limits correspond to approximately 10% variation in the average number of hired man-days per month (6517), derived from the solution that minimizes workforce variance subject to execution of the annual program (A). For each task, minimum and maximum areal extents per month were assigned in the ranges 10–25 ha and 80–250, respectively, depending on the task. The constant determining the condition of continuity was 0.1 ha, and the maximum total budget was \$1 675 525. This last value was used as a reference, and corresponds to the solution which minimizes the total cost subject the execution of the annual plan (A).

A large number of combinations of objective functions and constraints is possible. The following section presents the results obtained from a selection of these.

## Comparison of Objective Functions

We first compared the solutions obtained for the four objective functions, subject only to execution of the annual program of silvicultural tasks according to the company's operative targets (A). Then we examined the solutions obtained by including the additional constraint regulating the minimum and maximum task sizes per month (S).

### Solutions under constraint A only

Table 1 shows that minimization of total cost (the most efficient in economic terms) results in a high monthly variation in budget and workforce. Minimization of total labour also generates

a high operative instability and, even though a reduction in required man-days of 3.5% is achieved, the total cost is increased by 3.6%. This result is due to the fact that periods of greater task productivity do not necessarily coincide with lower operational costs.

**Table 1.** Solutions obtained by minimizing various objective functions subject to different constraints.

Analyses		Cost		Labor		Iterations
Minimize:	Subject to:	[000-\$]	CV [%]	[man-days]	CV [%]	
Total Cost	A	1676	87.2	76490	94.3	1
Total Labor	A	1736	91.6	73793	92.9	1
MMW	A	1794	13.5	77088	6.6	221
Workforce Variance	A	1821	11.9	78208	0.5	220
Total Cost	A, S	1709	48.7	78405	58.4	954
Total Labor	A, S	1759	49.4	75915	48.3	991
MMW	A, S	1816	16.3	78403	15.1	26802
Workforce Variance	A, S	1825	14.1	78519	8.5	250000
Total Cost	A, F	1717	17.4	76757	18.5	259
Total Cost	A, R	1737	13.6	76981	10.2	194
Total Cost	A, F, R	1738	14.1	77077	9.3	212
Total Cost	A, S, C	1729	55.3	78763	64.6	20316
Total Cost	A, S, F	1753	16.3	78177	18.7	54162
Total Cost	A, S, R	1780	14.6	78149	10.2	13844

A = Annual program; S = Minimum and maximum size per task ; F = Maximum workforce fluctuation; R = Range of allowable man-days per month; C = Continuity condition.

Figure 1 shows the workforce distribution per month for the solutions obtained from each of the basic models. Minimization of the maximum monthly workforce (MMW) and minimization of workforce variance both generate a homogeneous distribution throughout the year. However, this greater labour stability implies an important increase in costs (greater in the case of variance minimization). Minimization of workforce variance is insensitive to total cost and number of man-days, and thus achieves greater labour stability by means of an inefficient assignment of tasks – that is, to even out the workforce distribution in months of least activity it allocates to the tasks of poorest productivity. Consequently, minimization of workforce variance leads to an increase of 8.7% in costs and 5.4% in man-days compared to the minimization of total cost and labour, respectively.

### Solutions under constraints A and S

When analyzing the solutions at the tree-farm level, it was observed that the minimization of cost and labour concentrate each task in the month which is most convenient. In several cases this results in the contractors' actual workload capacity being exceeded. In contrast, MMW and minimization of workforce variance generate solutions involving such low activity per month on some tree-farms as not justify keeping a task open. To address this problem, we added the constraint limiting the minimum and maximum task sizes each month (S).

Table 1 shows that this additional restriction generates an important increase in operational costs under minimization of costs and labour, but it also reduces the monthly variability in

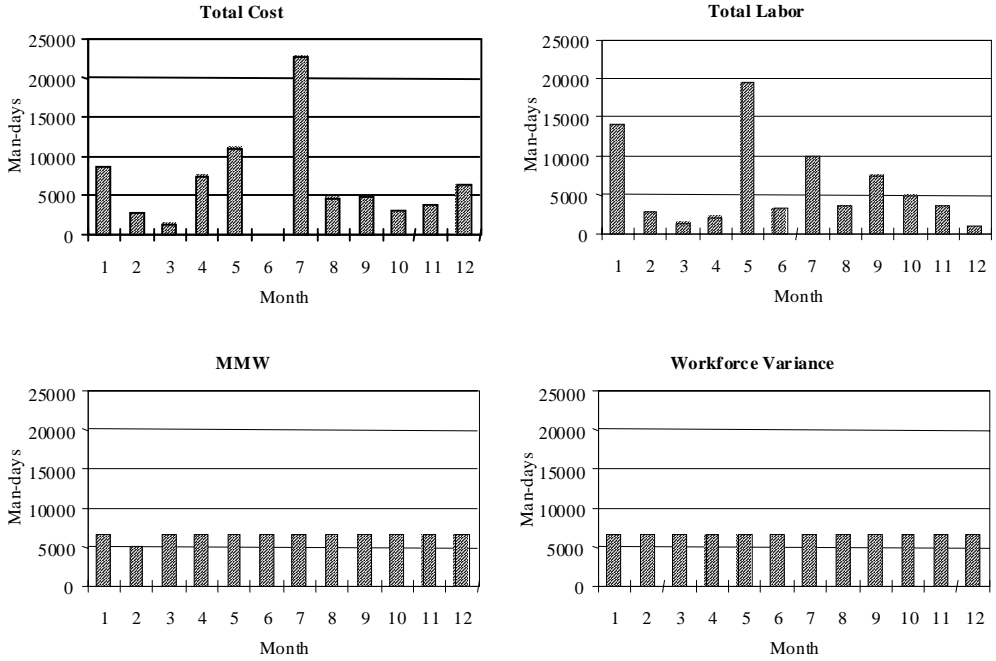


Figure 1. Monthly workforce distribution according to the selected objective function.

costs and labour. In the case of the remaining two objective functions (MMW and minimization of workforce variance), adding constraint  $S$  had a lower impact on costs but decreased labour stability.

The processing time is negligible for the basic models, including minimization of workforce variance (which is the only non-linear programming model). For the problems involving integer variables, a significant increase in processing time took place for the MMW model, although the solution time did not exceed 10 minutes on a Pentium Intel-166 MHz computer. However, in the problem of variance minimization (a non-linear integer model), the processing time exceeded 24 hours even when the solution corresponding to the MMW model was used as a starting point. In this case, the best feasible solution obtained within a limit of 250 000 iterations was selected.

### Comparison of Constraints

Several constraint combinations were examined for the objective function of minimizing total costs. In each formulation, new restrictions were added successively to the basic problem – minimizing total costs subject to completion of the annual program ( $A$ ). Table 1 shows how economic interests oppose those of labour and operative stability, by augmenting total costs as the constraints of the problem increase.

For example, limiting the range of man-days ( $R$ ) has a larger impact on cost than restricting the workforce fluctuation ( $F$ ), but also makes the workforce distribution more homogeneous. By restricting the fluctuation in the number of man-days between consecutive periods ( $F$ ), it is possible to achieve a considerably more uniform distribution of activity than in the base

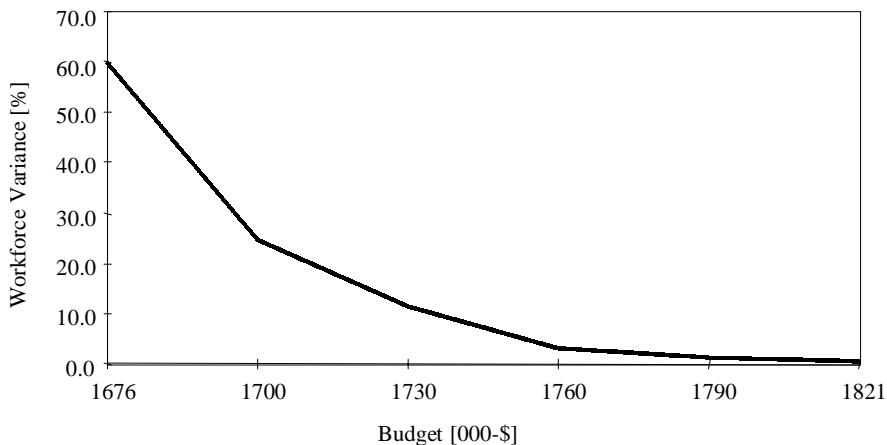
case (the coefficient of variation decreases from 94.3% to 18.5%). However, the changes in the number of man-days present a trend, the accumulated difference over 6 months being close to 35% for a tolerated fluctuation of 10% (4555, 7336, and 5374 man-days for months 1, 6 and 12, respectively). Combining constraints  $F$  and  $R$  has little effect on costs but a significant effect on labour stability.

Combining the constraints of minimum and maximum task area per month ( $S$ ) and the condition of continuity ( $C$ ) avoids a series of operative problems, but increases the total cost by 3.2%. Combining also the constraints on workforce range and fluctuation ( $R$  and  $F$ ), the total cost increases by 8.5% compared to the basic model (from thousand-\$ 1676 to 1818), while the coefficient of variation of the workforce is reduced from 94.3% to 9.2%.

The processing times in Table 1 indicate that when the complete set of constraints is used a combinatorial problem takes place, which is difficult to solve due mainly to a conflict between the continuity condition ( $C$ ) and the restrictions related to workforce distribution ( $R$  and  $F$ ). Under this situation it was not possible to find an optimum solution in less than 2.5 million iterations, equivalent to approximately 12 hours of computer time. However, if a 3.5% tolerance is considered for the theoretical limit, a reasonable solution can be obtained in approximately 30 000 iterations. All other formulations of the problem had a solution time of less than 20 minutes.

### Trade-off Between Cost and Labour Stability

The problem of minimizing the workforce variance was solved for different allowed maximum total costs. Figure 2 presents the results.



**Figure 2.** Trade-off curve cost-labour stability.

This formulation of the problem allows us to determine the best possible labour stability not exceeding a given total budget. For example, with a budget of \$1.79 million, a workforce variance of 1.2% is obtained, to be compared with the 6.6% obtained for the MMW model having the same cost (Table 1). In the case of minimizing total cost subject to range and

fluctuation constraints (*R* and *F*), the workforce variance is 9.3% with a cost of \$1.74 million (Table 1). With the same budget, the variance is 8.5% when the workforce variance is minimized, increasing to 11.4% as the maximum total cost allowed is decreased to \$1.73 million.

The curve in Figure 2 corresponds to the most efficient transaction points, in terms of the cost of improved labour stability. In the case illustrated, the curve shows a relatively favorable cost/labour stability trade-off up to \$1.73 million. From that point on, improved stability is achieved only with considerable increase in total cost.

### Dominant Solutions

In the above we examined the problems of minimizing workforce variance and MMW under the constraints of annual program (*A*) and minimum/maximum task areas (*S*). Here, for each problem we included an additional constraint that limits the budget allowed for the solution (PB = the reference budget of \$1.676 million, or PS = a larger budget of \$1.709 million).

Table 2 indicates that it is possible to improve labour stability without increasing costs. In models with constraint *A* only, the workforce variance decreased from 94.3% to 67.0% (or 59.7%) for MMW (or minimizing workforce variance). For the models that also include size restrictions (*S*), gains in labour stability are less because these restrictions themselves have a regulating effect when the total cost is minimized. However, when the operative budget is larger (constraint PS), there is more scope for finding a more stable labour solution (Table 2).

**Table 2.** Dominant solutions.

Minimize:	Analyses Subject to:	Cost [000-\$]		Labor [man-days]	
		Total	CV [%]	Total	CV [%]
Total Cost	A	1676	87.2	76490	94.3
MMW	A, PB	1676	60.9	76903	67.0
Workforce Variance	A, PB	1676	50.7	76875	59.7
Total Cost	A, S	1709	48.7	78405	58.4
MMW	A, S, PS	1709	46.5	77863	53.5
Workforce Variance	A, S, PS	1709	40.1	78287	48.5

A = Annual program; S = Minimum and maximum size per task; P = Maximum budget. (PB = 1676 thousand-\$, PS = 1709 thousand-\$)

For a given budget, optimal labour stability is achieved by minimizing the workforce variance. However, one should bear in mind that this objective function is a quadratic form which, with integer variables, becomes a difficult problem to solve. In contrast, minimizing MMW is a Min-Max type model of linear nature, that enables solutions to be found in considerably less time, more so when the size of the problem increases.

The optimization procedure employed generates dominants solution, in the sense that they tend to maximize labour stability without increasing the total cost. By solving the problem of minimizing total costs, the most economically efficient solution is obtained. When such a cost is then used as a budget restriction (constraint PB or PS) in models aimed at achieving labour stability, one obtains the optimal solution under these two conflicting interests.

## Conclusions

The models examined above highlight how the problem of scheduling silvicultural tasks can be formulated in different ways, according to the objectives and priorities considered at the moment of planning activities.

In this sense, the objective function of minimizing total costs dominates economic interests. Nevertheless, it is possible to significantly improve labour stability through restricting the range of man-days to be hired, and their maximum fluctuation over time. The best results are achieved combining both restrictions, since the second one only controls the variation between successive months, but does not avoid the fluctuations accumulated throughout the season. This situation is corrected by restricting, in absolute terms, the minimum and maximum workforce to be hired in any period.

The best workforce distribution over the season is obtained by minimizing the workforce variance, although in order to achieve this the model employs a more costly allocation of productivity. The objective function of minimizing MMW also tends to standardize the distribution of man-days, with the advantage that it is a linear model that requires a shorter solution time. Minimizing MMW also tends to generate less labour demand and, consequently, a lower total cost. However, this objective function becomes indifferent to the distribution of man-days through time once it is not possible to reduce the workforce required for a certain month. Therefore, it is not very effective when activity levels vary greatly during the year.

The results of this study illustrate how economic interests oppose those of labour and operative stability. Consequently, it is necessary to compare several formulations in order to achieve a solution that harmonizes the various interests of the company. In particular, dominant solutions provide a favourable option to overcoming conflicting objectives, by allowing an improvement in labour stability without increasing total costs.

In conclusion, through the type of analysis illustrated in this study, it is possible to determine the effect of different objectives and constraints on the solution of the optimization problem. Such analysis enables forest managers to evaluate the additional costs required to satisfy the diverse requirements of the company. Our study illustrates the way in which mathematical programming techniques may be applied to efficiently plan silvicultural tasks at an operative level.

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# Assessing Long-Term Changes in Stand Productivity: a Case Study of Sessile Oak High Forests

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## Abstract

We present a stand-level assessment of long-term productivity changes in four sessile oak (*Quercus petraea*) high forests, using a mensurational growth and yield model. We estimated productivity trend curves as smooth functions of calendar date, in addition to factors such as age, stand density and site productivity. We used a network of 35 permanent plots observed over 60 years, for which age and date were reasonably independent, and silviculture had been accurately recorded. Stand basal area increment varied markedly over the past 60 years. At two of the four sites studied, the normalised trend in basal area increment increased by between 25 and 55%, depending on the underlying model for the age factor. The westernmost site (Normandie) showed a more complex trend (decreasing productivity until 1960, increasing thereafter). Problems remain concerning the generalization of these results to other species and regions, using dominant height as a summary of productivity. The extrapolation of past growth curves over the next 100 years is also discussed.

*Keywords: Quercus petraea, growth trends, global change, forest stand yield, France*

## Introduction

The productivity of European forests has systematically increased over the past 150 years (Spiecker et al. 1996). This now-accepted fact has important consequences at all levels of forest resource management:

1. At the stand level, rotation ages will be dramatically shortened: stands are now growing faster than their counterparts 100 years ago, by between 50% (Dhôte and Hervé 2000) and 100–150% (Pretzsch 1996). In several recent French surveys, young common beech (*Fagus sylvatica*) stands exhibited Site Indices far higher than any stands of the same age

during the 1930s (Contesse 2000). Some authors argue that silviculture may explain part of the growth increase in intensively managed plantations (Cannell et al. 1998) or in semi-natural broadleaf forests (Skovsgaard and Henriksen 1996). However, even lightly managed stands such as oak (*Quercus* spp.) high forests ('wild' genetic material, low thinning intensities, natural regeneration without site preparation or fertilization) have undergone significant growth increases (Becker et al. 1994; Bergès 1998). Increased forest productivity is now a fact with important implications for silviculture: for example, beech and oak optimal rotation age may be shortened by one-third. Two other important issues for foresters are: (i) the impact of enhanced growth on wood properties (specific gravity, log grading); and (ii) the increase in storm damage susceptibility as mean stand height increases.

2. At the forest estate level, shorter rotations have important consequences for regeneration scheduling. Clearfelling or shelterwood cuttings should progressively increase, as commercial maturity is reached sooner. Another management response is to decrease thinning intensity, in order to compensate for faster growth. These strategies need to be evaluated on an economic basis, using new models of growth, wood quality and risk. However, social constraints may also become more acute if regeneration is accelerated: clearfelling is increasingly contested in western Europe, even in the case of natural regeneration.
3. At the regional level and in the socio-economic context, growth changes also influence attitudes to wood supply, the functioning of the forest industry sector, and forest policy (e.g. the potential for allocating forest labour to the most profitable areas, while reserving large areas for recreation). There is also concern about the ecological sustainability of fast-growing forests: the possibility of nutritional deficiency and its consequences such as forest dieback (see also Corbeels et al. 2001; McMurtrie et al. 2001).

In order to help strategic planning in silviculture and resource management, new growth and yield models are needed. Classical mensurational models, based on permanent plot observations, are considered as efficient tools for simulating stand dynamics in response to site quality and silviculture. However, they are also generally viewed as purely empirical, and hence not appropriate for exploring the effects of changing environment. In this paper, our objective is to illustrate an attempt to modify a mensurational model, in order to detect past changes in stand productivity. The modification simply consists in incorporating a trend curve, as a function of date, to account for the historical evolution of productivity. Thus the model is used as a diagnostic tool. We did not consider the problem of predicting future changes, but we shall examine some methodological issues related to this question in our conclusion.

Many of the available results on growth changes were obtained from dendrochronological data (radial growth) or stem analyses (height growth) (Becker et al. 1995; Unthelm 1996). An important unknown, with retrospective methods, is linked to the fact that current tree growth curves are used to estimate what happened in the past. Generally, a sample of trees is chosen from the present population (e.g. dominant trees) and the assumption is made that this sample had the same social rank throughout the stand's life. This assumption, in turn, relies on the classical observation that social rank is strongly conserved in densely stocked high forests such as plantations and even-aged forests (Delvaux 1981). Nevertheless, rank conservation is not absolute: growth trends estimated through retrospective methods may be biased by a signal linked to stand dynamics (tree social promotion). Little information is available to measure this bias, although adequate material exists in permanent plots to quantify it. One can expect that bias is larger for radial than for height growth, due to their different growth patterns.

A second drawback of retrospective studies is that they are based on tree-level increments. The passage to stand level is not always straightforward. In the case of height growth, it can

be argued that dominant height (the average height of the 100 largest trees per ha) is an excellent indicator of stand productivity (e.g. Eichhorn's law, (Eichhorn 1904)) and is almost insensitive to stand density (Lanner 1985). The insensitivity of dominant height growth to spacing is sometimes contested, especially in pine (*Pinus* spp.) plantations (MacFarlane et al. 2000; Meredieu 1998). However, thinning effects on dominant height growth are well-documented for species like Norway spruce (*Picea abies*), common beech (*Fagus sylvatica*) or oaks (*Quercus* spp.), that were intensively studied in Europe: they are hardly detectable, provided that stand density remains not too low. It is reasonable to assume that silvicultural influence on dominant height growth is restricted to early stages of development (plantation density, length of natural regeneration) and does not affect further growth rates. Hence, retrospective analyses of height growth may be used for estimating past environmental conditions, by taking elementary precautions (e.g. using differential equations with respect to height rather than age, excluding data relative to early growth).

In contrast to height growth, radial growth measured on increment cores responds to silviculture: if thinning intensity has changed over the past decades, there is probably a silvicultural signal in tree-ring chronologies taken from managed stands. This is a specific drawback in European forests, the majority of which are managed, and the interpretation of such data with regard to global change is not straightforward.

Thus, in order to assess stand productivity changes, it is also necessary to use continuous stand-level observations, e.g. in permanent plot networks. Two conditions must be met: (i) long-term observations of stand yield must be available, with reasonably stable protocols; (ii) in the observed increments, one must be able to separate the 'natural' trends with stand ageing from the external forcing attributable to environmental changes. For example, Pretzsch (1996) used all Bavarian permanent plots and compared observed volume increments to expected values from Yield Tables. It is difficult to assess whether Yield Tables are appropriate tools for such purposes: the quality of growth changes estimated this way depends strongly on the quality of the Table itself, especially regarding the pattern of stand increment with age. Because Yield Tables were fitted to data from stands that were undergoing growth changes (during the 20th century), it seems preferable to perform new analyses of the raw data and test for date-related trends in the increments. The objective of this paper is to illustrate such a study.

We chose to analyse sessile oak (*Quercus petraea*) high forests in France for several reasons: (i) a network of 35 permanent plots had been installed in the 1930s, at four sites distributed across France (from Normandie in the West to the more continental location of Lorraine in the East); (ii) age and date were fairly independent in this data set, due to the concomitant installation of plots at all stages of development (polestage to mature forest) at the same date; (iii) this allows us to analyse the dynamics of oak stands throughout the stand's life, with a 'wild' and relatively stable genetic population; and (iv) over a period of 60 years, the only silvicultural operations were a series of thinnings of different intensities (the influence of which may be included in stand yield models by simple response curves).

## Material and Methods

### Plots and measurements

The French sessile oak network was created between 1925 and 1934. It is composed of four forests, Bellême (Normandie, 0°31'E-48°23'N), Blois (Loire Valley, 1°16'E-47°34'N), Tronçais (Centre, 2°44'E-46°39'N) and Champenoux (Lorraine, 6°21'E-48°42'N). In each

forest, 10–11 plots (four in Champenoux) were installed simultaneously in five to six compartments at various stages of development (from polestage to mature high forest). In the three youngest stands of each forest, generally two plots were installed, in order to compare heavy vs light thinnings. In mature stands, only one plot was available, in order to provide yield figures during the regeneration phase. Initial stand age in the 1930s ranged from 35 to 200 years. Current age, at any date, ranged from 35 to 230 years. Hence, at any date between 1930 and the present, a large range of ages are present in the data base. The overall simple correlation between date and age is 0.329. Plot area is generally 1 ha, exceptionally 0.5 ha in two young stands, and 2 ha in four mature stands.

All plots are at low altitude and on almost flat terrain. Geological and soil conditions vary between forests (Trencia 1989). Within a forest, there may be some site differences between plots. However, no detailed ecological description was done: this part of the network analysis will be undertaken in the near future, as part of a further study of height growth patterns in the surrounding regions. Unfortunately, Site Index can no longer be used – at least not so simply – to test for site productivity similarity between plots, because height growth has been accelerating over the past decades. In the analysis, differences of site conditions were accounted for by estimating local productivity parameters (at the plot level).

Stands are pure and even-aged. Age evenness was checked at plot installation, by counting rings on stumps (tree individual ages are  $\pm 15$  years about the stand mean). The degree of mixture was analysed throughout the observation period, using successive inventories. Associated broadleaves (mainly common beech) never represent more than 10% of overstorey basal area. Understorey was not systematically monitored: its stocking and species composition differs between forests (very sparse in Blois, sparse-beech-hornbeam in Tronçais, denser in Bellême, dense and very diverse in Champenoux). From previous studies of individual tree growth and stand yield, we speculate that understorey history has little influence on the results presented here. However, this cannot be easily tested.

Silviculture was quantified by stand density. We used an index computed after Reineke (1933), *Rdi*: if  $N$  is the number of stems per hectare and  $d_g$  (in cm) is the quadratic mean diameter, then

$$Rdi = \frac{N \cdot d_g^\alpha}{\beta} \quad \alpha = 1.701 \quad \text{and} \quad \beta = 171582$$

The overall range of *Rdi* in the data set is between 0.45 and 1.05, except for stands during shelterwood cuttings (where *Rdi* rapidly approached 0). Of course, the heavier the thinnings, the lower the density indices. Globally, there is a slight trend for *Rdi* to decrease with stand age, with a simple correlation of  $-0.322$ .

Data collected in these plots are two-fold: periodic complete inventories of tree girth at breast height and periodic measurement of total height (and volumes) on samples.

Inventories were performed on the following population: below age 60, trees were not physically numbered, and all trees larger than 1 cm (diameter at breast height) were measured for girth (or diameter) at breast height, whatever the species. Above age 60, tree numbering was done only on overstorey trees (oaks and other species), and hence, following this operation, only numbered trees were measured. Around the date of tree numbering, a specific method was designed to accommodate for the changing population: trees not numbered were considered as a thinning.

From these raw data, all stand statistics are computed (number of trees per ha, basal area etc...). Basal area increment is the variable analysed in this paper. This increment is 'raw',

i.e. it includes trees dying during the corresponding period: net increments were much more erratic, due to irregular mortality rates (silviculture also influences this, and affects basal area increment regularity). Accounting for dying trees is straightforward when tree lists are used. When only histograms per diameter classes were available, we used a method for estimating which diameter classes the dead trees belonged to (Dhôte and Hervé 2000).

Total height was measured on samples, not necessarily at each inventory date, but systematically when a thinning was performed. Until 1950, these samples were large-sized (up to 250 trees per plot) and composed of all thinned trees (measures after felling). Later on, samples of approximately 30 trees per plot were selected, so as to spread uniformly over the whole diameter range and the plot area. These trees were measured standing, using optical devices. Hyperbolic height-diameter curves were fitted to each sample, using previous analyses by Dhôte and Hercé (1994): in this 3-parameter equation, two parameters were fixed for the whole data set, and one was estimated for each sample. When no sample was available, the free parameter was interpolated. Dominant height was estimated using dominant diameter and height-diameter curves. There are a number of errors affecting dominant height estimated in this way (sampling, measurement and modelling errors). The accuracy of height (and furthermore height increment) was considered too low for the present analysis.

For the analysis, the sample size was: four forests, 35 plots, 340 inventories (plot x date), 305 growth periods (78 in Bellême, 102 in Blois, 46 in Champenoux, 79 in Tronçais).

## Analysis

The objective was to analyse the variations of stand basal area increment  $iG$ , and specifically to test whether there has been any long-term trend affecting this variable. However, basal area increment is also affected by many factors:

- stand age (or stage of development);
- site quality;
- silviculture (described here by stand density  $Rdi$ );
- short-term climatic fluctuations (drought, frost damages, wind disturbance, that we wanted to filter out); and
- other biotic or abiotic damage.

A common way to decompose these sources of variability is to use a multiplicative model of the following form:

$$iG = F_1(H_0).F_2(Rdi) + \varepsilon \quad [1]$$

where  $F_1$  is a potential stand increment, depending on age and site quality (both summarized in dominant height  $H_0$ ),  $F_2$  is a modifier for silvicultural effects (here using  $Rdi$ ),  $\varepsilon$  a normal error accounting for all other factors.

This model was adapted for the present study:

- we preferred not to use dominant height as a predictor for age and site influences; indeed, since height is probably influenced by long-term changes, part of the trend we were trying to estimate might have been absorbed by height; instead, we used a function of stand age and estimated plot-specific 'site quality' parameters;
- a trend was introduced, as a multiplicative correction to model [1]; this trend was a smooth function of date;

- in addition to the trend, periodic fluctuations were also estimated; and
- we observed that residual variance was not homogeneous with age, being larger in the young stages, when the independent variable was large; therefore  $iG$  was log-transformed, and we assumed that the error was normal and additive on  $\text{Ln}(iG)$ .

These assumptions led to the following model: the functions in model [2] were chosen by graphical means, in order to mimic the observed trends (see below):

$$\text{Ln}(iG_{\text{plot, period}}) = \theta_{\text{plot}} + \theta_{\text{period}} + G_1(\text{age}) + G_2(\text{Rdi}) + G_3(\text{date}) + \varepsilon \quad [2]$$

The question of model parameterization was difficult. The data set had a hierarchical structure (forests, plots inside forests). Since forests are quite distant from each other, subjected to contrasting climatic conditions and based on different soil types, it was possible that they behaved differently with regard to any of the three functions in model [2]. To make the problem statistically solvable, the following assumptions were adopted:

- the shape of the age curves  $G_1$  may differ between forests, but is the same for all plots inside a forest; so we had to test for forest-specific parameters in this model component;
- the response curve to stand density  $G_2$  is a characteristic of the species behaviour, and hence is global for the whole data set; and
- no *a priori* assumptions could be made for date trends, and we adapted the model to the observed patterns.

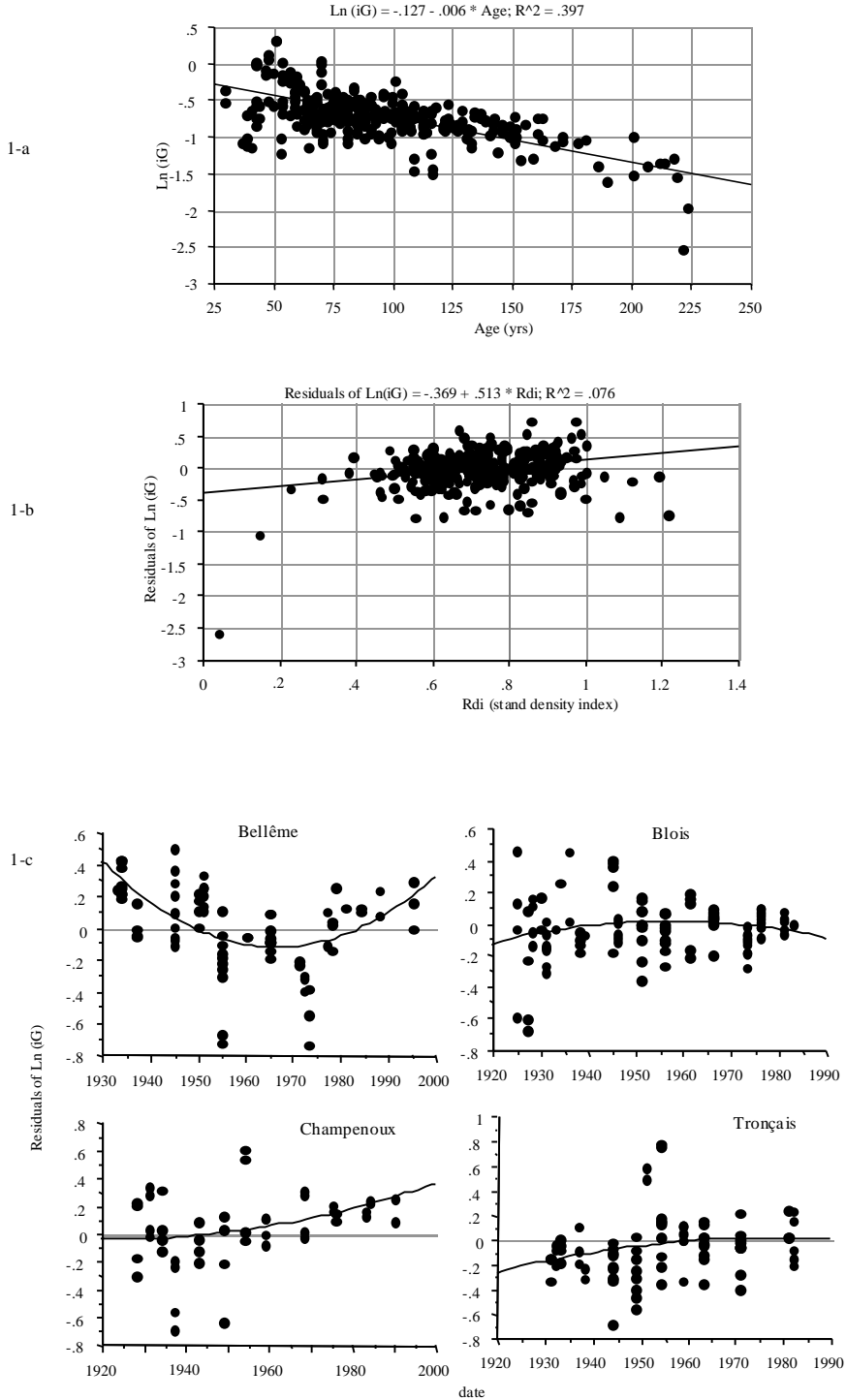
Only linear models were considered for functions  $G_{1 \text{ to } 3}$ . It was fitted using the stepwise regression procedure (descending mode) of software Statview 4.5™. Forest, plot and period specific parameters were estimated using dummy variables. Effects having a probability of t-test lower than 0.05 were removed from the model: for example, a plot-specific parameter was estimated only when the plot increment differed significantly from the ‘mean model’.

## Results

The first three steps of data analysis are summarized in Figure 1. Age is the first variable introduced in the regression: log-transformed stand basal area increment decreases linearly with age (Figure 1-a). At this stage no forest-specific parameters were introduced.

Figure 1-b shows the residuals of the first step, plotted against Reineke Density Index  $Rdi$ . Inside the [0.5; 1] interval, there is a slight trend for stand increment to increase. However, a linear model is not appropriate at both extremes of  $Rdi$ : also, returning to raw data from a linear log-transformed model would give an exponential response curve, and we know that this is not appropriate. The best combination we found was to use  $\text{Ln}(Rdi)$  (with a positive parameter) and  $Rdi$  (with a negative parameter). Returning to the untransformed expression, this produces a response curve of the form:  $Rdi^a \cdot \exp(-b \cdot Rdi)$ , with  $a$  and  $b$  positive parameters. Since estimates of  $a$  (respectively  $b$ ) were in the [1.3; 1.5] (respectively [1.6; 2.0]) intervals, this produces response curves with a flat maximum around  $Rdi = 0.7$ .

Figure 1-c shows the residuals after the first two steps. These residuals are segregated by forest, and three different patterns are visible: (1) in Bellême, a parabolic shape, productivity being higher than the mean at the beginning and end of the observation period; (2) in Blois, no visible structure, and a quite large variance during the first decades; and (3) in Tronçais and Champenoux, a monotonic trend.



**Figure 1.** Three first steps of the analysis of log-transformed stand basal area increment (Ln(iG)). 1-a. Linear regression with age as independent variable. 1-b. Residuals of former step plotted against Reineke Density Index *Rdi* (see text for definition). 1-c. Residuals of former steps plotted against date for each forest.

These forest-specific behaviours could reflect our assumptions made at previous steps: for example, removing age-effects with only one (global) parameter leads to a biased model if forests have distinct age parameters. This possible bias was tested, by using forest-specific age parameters. Such parameterizations were never able to remove the specific forest trends presented in Figure 1-c. Date-trends were introduced in the model as linear functions. To account for the particular case of Bellême, a segmented linear model was considered, with two linear segments intersecting in 1960.

The fourth step of the analysis was to introduce two series of dummy variables in the model:

- a series of 34 plot-indicators, in order to estimate productivity levels for plots being significantly more or less productive than the average; and
- a series of 13 calendar period-indicators (5 year intervals), in order to estimate temporary higher or lower increments.

The last step of the analysis was to introduce all effects in the regression, and test for the need to consider forest-specific age parameters. The final model reads:

$$\begin{aligned}
 \text{Ln}(iG) = & \alpha + \sum_{\text{forests}} \varphi_f \cdot I_f + \beta_0 \cdot \frac{\text{age}}{100} + \sum_{\text{forests}} \beta_f \cdot \frac{\text{age}}{100} + \gamma \cdot \text{Ln}(Rdi) + \delta \cdot Rdi \\
 & + \chi_0 \cdot \text{date}' + \sum_{\text{forests}} \chi_f \cdot I_f \cdot \text{date}' + \psi \cdot I_{\text{Bellême}} \cdot \text{date}'' \\
 & + \sum_{\text{periods}} \omega_{\text{per}} \cdot I_{\text{per}} + \sum_{\text{plots}}^{34} \zeta_{\text{pl}} \cdot I_{\text{pl}}
 \end{aligned} \tag{3}$$

where  $I_f$  is 1 in forest  $f$ , else 0 (same for plots & periods)

$$\text{date}' = \frac{\text{date} - 1960}{30} \text{ and } \text{date}'' = \text{date}' \text{ if } \text{date} < 1960, \text{ else } 0$$

In this model,  $\beta_0 \cdot \frac{\text{age}}{100}$  is the average age-pattern; if any of the parameters  $\beta_f$  is found different from 0, then the corresponding forest has an age-pattern significantly different from the average. The same parameterization is used for date-trends, in order to test for specific forest-trends.

Statistics of fit of model [3] are given in Table 1. The model explains 73% of the variance of Log(basal area increment) and the F-test of regression is 48.6. The most important parameters were estimated with a high precision (probability of t-test less than  $10^{-4}$ ). No forest-specific parameters were found to be significant for the age-effect. The date-trends varied markedly between forests, confirming the trends shown in Figure 1: Bellême had a complex pattern (decreasing productivity between 1930 and 1960, then increasing), Tronçais and Champenoux had a continuously increasing productivity, Blois increased less.

## Discussion and Conclusions

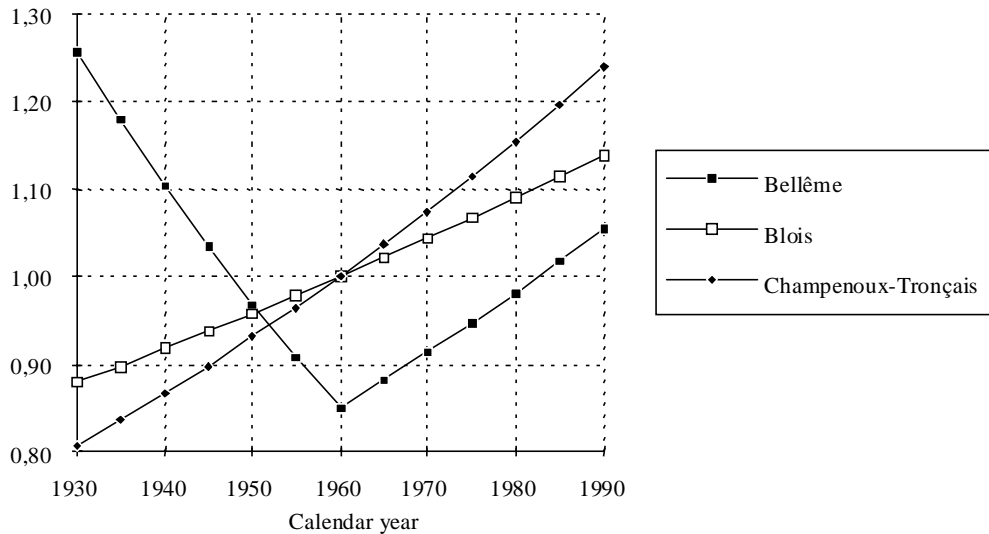
### Stand-level data confirm large productivity changes and suggest regional differences

Figure 2 shows the date-trends for the four forests (Tronçais and Champenoux grouped), when we return to the original variables. These curves represent the multiplier to be applied



**Table 1.** Statistics of fit for model [3]. Name of variable and parameter, parameter estimate, probability of t- test for each effect.

Independent variables with an associated parameter significantly different from 0			
Variable	Parameter name	Parameter estimate	Prob (t)
Intercept	$\alpha$	1.885	<0.0001
age'	$\beta_0$	-0.582	<0.0001
Ln(Rdi)	$\gamma$	1.494	<0.0001
Rdi	d	-2.067	<0.0001
I-Bellême	$\varphi_{\text{Bellême}}$	-0.162	<0.0001
date'	$\chi_0$	0.215	<0.0001
I-Blois.date'	$\chi_{\text{Blois}}$	-0.086	0.0456
I-Bellême.date''	$\psi$	-0.606	<0.0001
Period name	Period parameter		
I-1930-34	$\omega_{30-34}$	0.113	0.0058
I-1950-54	$\omega_{50-54}$	0.197	<0.0001
I-1970-74	$\omega_{70-74}$	-0.183	<0.0001
Plot Name	Plot parameter		
I-Hermoussel1	$\zeta_{\text{Herm1}}$	-0.150	0.0453
I-Sablonnières rouges	$\zeta_{\text{SR}}$	0.217	0.0152
I-Sablonnières1	$\zeta_{\text{Sablo1}}$	-0.193	0.0028
I-Sablonnières4	$\zeta_{\text{Sablo1}}$	-0.169	0.0159
I-Charmaie	$\zeta_{\text{Charm}}$	0.276	0.0395
I-Bois Brochet1	$\zeta_{\text{BB1}}$	-0.143	0.0400
I-Clé des Fossés	$\zeta_{\text{CdF}}$	0.160	0.0293

 Index of relative basal area increment :  
 reference 1 = Tronçais in 1960.

**Figure 2.** Qualitative behaviour of model [3]. Trend curves of stand basal area increment, plotted against date (from 1930 to 1990). These curves represent the multiplier to be applied to a general model, in order to feature date-related changes of French sessile oak productivity.

to an 'intrinsic' model (function of age, site quality and silviculture), in order to account for the historical change of productivity in our data set. The reference value 1 was chosen for Tronçais in 1960.

The behaviour at Tronçais and Champenoux is the simplest, and also the most consistent with former results obtained from dendrochronological data. In these two forests, stand basal area increment has increased by 54% between 1930 and 1990. Blois is also increasing, at a lesser rate (+29%): the reason why this trend was not obvious in Figure 1-c is that 3 out of 10 plots in this forest had productivity levels quite different from the mean (in Table 1: plots Sablonnières 1 and 4, Charmaie). The dendrochronological study of Becker et al. (1994), concerning oak forests near Champenoux, gave a radial growth increase of +47% between 1930 and 1987, which is close to our figure.

These figures of growth change were obtained with model [3]. What level of confidence can we have in these results? Model [3] may seem quite complex. However, complexity arises mainly from the high number of dummy variables; the basic model has just 3 continuous independent variables: age, date and *Rdi*. As we mentioned above, these 3 variables are not completely independent: there are slight correlations between them (simple correlation coefficients are about 0.3). This collinearity is due either to the temporal sampling plan or to the evolution of thinnings during the stands' life. The question is then: can this collinearity create the date-trend, or at least bias its amplitude or sign?

At intermediate steps of the analysis, alternative models were considered. The most critical modelling choice, as far as growth change is concerned, is the method of parameterizing age-effects. Changing the model from a global age-parameter to specific forest age-parameters modifies both the age and date parameters, due to the slight correlation between age and calendar date (steeper age-effects are generally associated with steeper date-trends). In Tronçais, for example, the estimate of growth change from 1930 to 1990 varied from 24% to 54%, according to the underlying age model.

Thus, our estimate of growth change is sensitive to the structure of the age-model. Furthermore, among the variety of tested models, we adopted model [3] giving the larger estimate of growth change (54% in Tronçais). We chose this model mainly for two reasons: (i) the parameterization allows one to decide statistically whether forests should be distinguished for both age and date effects; and (ii) the model structure uses all the relevant information contained in the data base (forests, plots, silviculture, age, date).

The most surprising fact, in our analysis, is the complex history of Bellême, where the trend was not monotonic. Productivity has sharply decreased between 1930 and 1960, and thereafter has increased at the same rate as other sites. We do not think that this is an artifact of the model: even with different equations or with specific forest age-patterns, the result remained highly significant. This pattern is difficult to understand in terms of the effects on growth of CO<sub>2</sub> atmospheric concentration, nitrogen deposition and climatic warming (Cannell et al. 1998).

Little information is available in the literature, concerning regional differences in the long-term trends of forest growth. In the compilation by Spiecker et al. (1996), growth changes were found by almost all contributing teams; the amplitude of change was variable, but there were also a number of methodological differences that prevent a direct comparison between sources (e.g. large inventory databases vs. permanent plots, height vs. diameter growth, continuous yield observation vs. retrospective studies).

In conclusion, growth change estimates are sensitive to the quality of the age model and to the robustness of the sample in the (age, date) plan. It would be of interest to obtain new data on regional variations for growth change, by using reasonably standardized methods of sampling, measurement and modelling. What makes the problem difficult is that there may be regional or local (site) variability for both the age-pattern of increments and the date-related trends. To confirm our result, it would be useful now to collect new data sets specifically

addressing this problem: the sampling strategy will rely on: (i) strict control of ecological conditions; (ii) considering different regions and replicating the design for different species; (iii) separating age and date; and (iv) combining basal area observations in permanent plots and dominant height reconstruction using stem analyses.

### **How can we adapt growth & yield models to a changing environment?**

Among forest scientists, it is common to contrast growth and yield models and process-models of ecosystem functioning, especially regarding their suitability for forecasting future stand dynamics under a changing environment (Mohren and Burkhardt 1994). Usually, the former are considered empirical, i.e. constrained to reproducing historical observations from field trials, whereas the latter are considered more general and more adapted to simulating changing ecological situations.

However, growth and yield models are not simply reproductions of the ‘real world’, nor a set of regression equations. They involve a theoretical framework that is frequently overlooked, even – sometimes – by forest biometricians: (i) the concept of Site Index, for pure, even-aged forests, is a very efficient simplification of site-related growth variability; (ii) curves relating volume growth to stand density indices (Assmann 1970) are also efficient, and present analogies with growth responses to Leaf Area Index in process-based models; and (iii) the age-dependence of growth rates, or more generally the effect of developmental stage – height being commonly used as a driving variable in growth and yield models (Arney 1984) – is physiologically meaningful (Valentine 1985; Mäkelä 1986; Dhôte 1996).

On the other side, process-based models are not simply general, theoretical representations of ecosystem dynamics. They involve a large number of parameters, for which estimates are obtained by field or laboratory experiments. In this regard, the quality of predictions provided by process-based models is dependent on the quality of parameterization, and hence on the underlying data base. In addition, these models are usually quite complex, due to a large number of processes represented. This makes them more versatile, but also more subject to artifacts that may be difficult to evaluate.

Mohren and Burkhardt (1994) did not consider that process-based models would ever replace growth and yield models in traditional forest management applications. Simplicity of design is a major advantage of the latter, not only for practical purposes, but also for model fitting and validation. However, a fundamental hypothesis of growth and yield models is now clearly violated, the invariance of site conditions: Site Index, for example, must be revisited. The major drawback of growth and yield models is not that they are contingent on data (all models are, except purely theoretical ones), but that they are based on a phenomenological theory that does not explicitly feature the underlying growth factors (C, N and other nutrients, water).

In order to predict future stand dynamics under changing environment, growth and yield models will have to change radically. Two aspects of their design seem critical (see also Dewar 2001):

- the state variables (height, basal area, density indices...) might become more diverse: for example, the use of Leaf Area Index should be examined (does LAI bring information beyond that of stand density indices?); also, explicit inputs of the main environmental resources (CO<sub>2</sub> concentration, soil N and water reserve) will probably be necessary; it is certainly counterproductive to get into sophisticated details of basic physiological processes associated with these resources, but general response curves for height and basal area growth would be very useful; and
- the structure of growth equations should be improved, by incorporating concepts of process-based models: for example, transposing the principles of carbon budgets at the

relevant scales (stand, year) could help designing more robust equations of volume or basal area increment, especially their dependence on developmental stage.

From these examples, it is clear that growth and yield models cannot evolve without incorporating ecological knowledge. Conversely, knowledge gained by forest biometricians may also be useful to process-modellers and ecological experimentalists:

- experiments in controlled conditions, e.g. on CO<sub>2</sub> greenhouses, are usually performed on small trees and need to be extrapolated to 'real ecosystems'; an interesting test would be to compare such greenhouse results with the historical analysis of past growth trends; and
- the simplicity of growth analyses makes it possible to collect extensive data bases, covering a large range of regions with different climatic and soil conditions; besides its usefulness to foresters, regional information on ecosystem dynamics in the past will certainly suggest some ideas on the cause of growth changes.

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# The Sustainable Management of Temperate Plantation Forests: from Mechanistic Models to Decision-Support Tools

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## Abstract

Sustainable forest management seeks to ensure that the behaviour of managed forest ecosystems is environmentally and socio-economically acceptable. In this synthesis paper I assess the actual and potential contribution of modellers to the development of sustainable forest management, from scientific understanding of the underlying processes to practical decision-support tools. On the scientific side, detailed forest ecosystem models have been developed to understand and predict management and climate impacts on forest behaviour, based on a comprehensive description of plant-soil and carbon-nutrient-water interactions. However, two key processes for sustainable forest management – plant growth allocation and soil nutrient cycling – continue to challenge modellers. Hydraulic homeostasis has recently emerged as a guiding principle for modelling growth allocation. Combining this with root-shoot functional balance leads to a simple scheme incorporating both hydraulic and nutritional constraints on allocation for use in simplified, process-based growth models. While this scheme predicts realistic growth and yield trends with stand age, the individual roles played by allocation and stomatal conductance require further study. Hydraulic homeostasis alone cannot separate these. Soil nutrient cycling models differ in how they describe the regulation of microbial growth and diversity, two key processes for sustainable management. Existing models of the former probably suffice (at least for N cycling), but uncertainty in the latter presents a major limitation for predicting long-term ecosystem behaviour; more experimental work is required here. Models should also incorporate non-microbial immobilization processes. On the practical side, mensuration-based growth and yield models have been incorporated into decision-support tools designed to find management practices satisfying mainly economic objectives (e.g. maximum net present value), subject to various constraints (e.g. even wood production). Information from process-based models has yet to make a significant contribution. I discuss three options for enhancing the information flow from process-based models to decision-support tools: (i) use of process-

based growth indices to improve conventional growth and yield models; (ii) simplification and direct incorporation of process-based growth models; and (iii) use of mass balance analysis to generate robust constraints between, for example, wood yield, ecosystem C storage and site nutrient loss. This last option provides a particularly promising way forward, whereby the environmental criteria of sustainability might be incorporated as extra constraints into established decision-support tools.

*Keywords: forest, management, model, plantation, sustainability*

## **Introduction**

### **How can models contribute to sustainable forest management?**

In its broadest sense, sustainable forest management is management that leads to environmentally and socio-economically acceptable forest behaviour. Whatever the criteria involved, they may be applied on a variety of timescales. For example, we might be concerned about whether, under current management practices, the water table/water quality/soil fertility/wood production/carbon storage/net present value/amenity value of a managed area of forest will decline to unacceptable levels over the next five years, or over the next five rotations. The word ‘sustainable’, however, implies taking the long-term view.

The first step, then, should be to define the sustainability criteria – the long-term objectives of forest management – and this, of course, lies outside the domain of modelling itself. Given these criteria, models can contribute directly to the assessment of sustainable forest management, by providing both qualitative understanding and quantitative predictions of the impact of various management practices on forest ecosystem behaviour over different timescales.

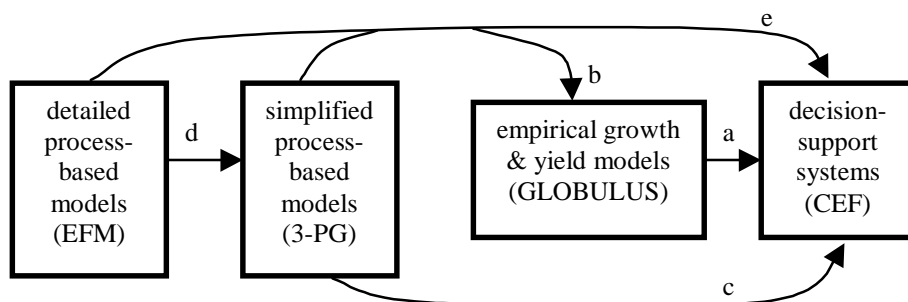
Several examples of models, and their application to sustainable forest management problems, are presented in this Proceedings. Broadly speaking, the models range in a quasi-continuous fashion between two extremes. At one extreme are research-oriented models (e.g. the Edinburgh Forest Model, Cannell and Thornley 2001), whose principal aim is to gain scientific understanding of the complex interplay between the many plant and soil processes involved. These models are generally characterised by a large number (sometimes hundreds) of parameters, many of which are difficult to estimate directly (but not all of which are necessarily critical to know accurately). At the other extreme are empirical, mensuration-based, growth and yield models and decision-support systems which currently provide practical tools for forest managers (e.g. GLOBULUS-CEF, Tomé et al. 2001). Lying between these two extremes are simplified, biologically-based growth models (e.g. 3-PG, Landsberg et al. 2001) which aim to capture the essential features of the underlying processes while using only readily available data as input.

We might picture the overall contribution of models to sustainable forest management as a flow of information from detailed process-based models, to simpler biologically-based models and empirical growth and yield models, to decision-support systems (Figure 1).

### **What are the main challenges for modellers ?**

Broadly speaking, modellers are presented with three main challenges if their models are to contribute to the development of sustainable forest management practices.





**Figure 1.** Possible flows of information between different kinds of model and decision-support systems: a = empirical yield predictions, b = growth indices, c = process-based yield predictions, d = model simplification, e = mass balance constraints (for details, see ‘Knowledge Transfer’ section of the main text). Examples of models and decision-support systems discussed in this Proceedings are indicated in parentheses: EFM (Edinburgh Forest Model, Cannell and Thornley 2001), 3-PG (Landsberg et al. 2001), GLOBULUS (Tomé et al. 2001) and the CEF forest management system (Tomé et al. 2001).

At the research end, there are two main scientific challenges. Firstly, because stemwood volume production is (and is likely to remain) of central interest to forest managers, an adequate model description of stemwood growth over a rotation is required, taking into account the main hydraulic, nutritional, energetic and other (e.g. mechanical) constraints on carbon assimilation and growth allocation. Of these, growth allocation is arguably the key challenge.

Second, assessment of the long-term sustainability of soil fertility, and of water and air quality, requires an adequate representation of the processes underlying soil nutrient cycling over timescales that include several rotations. Of central interest here are the processes regulating the availability of soil inorganic nutrients. These processes include mineralisation, immobilisation, inorganic inputs, leaching and gaseous emissions, and uptake by trees and other vegetation.

Together, these two scientific challenges demand no less than the construction of complete forest ecosystem models that couple the carbon, nutrient and water cycles in both the plant and the soil.

At the applied end, a third challenge is to incorporate the results from detailed, process-based models into practical tools for forest managers. How best can the scientific knowledge from relatively complex ecosystem models be transferred to decision makers (Figure 1)?

It is not the aim here (nor was it that of our workshop) to attempt a comprehensive review of forest models, but rather to highlight recent progress in developing and applying models to problems of sustainable forest management. I assess how modellers are currently meeting the three key challenges noted above – growth allocation, soil nutrient cycling, and knowledge transfer – and suggest ways in which their contribution to sustainable forest management might be developed in the future.

## Modelling Growth Allocation

Various approaches to modelling growth allocation are exemplified in this Proceedings. These range from mechanistic descriptions of the transport and utilization of labile C and N growth substrates in trees (EFM, Cannell and Thornley 2001), to empirical schemes linking

below-ground allocation to indices of soil fertility (3-PG, Landsberg et al. 2001) or water stress (GRAECO, Loustau et al. 2001), to the use of constant allocation fractions (G'DAY, McMurtrie et al. 2001b). These and other approaches were reviewed by Cannell and Dewar (1994). In this section I discuss a more recent approach that has emerged out of attempts to understand the age-related decline in stemwood growth.

### Age-related decline in tree growth

This phenomenon, well-documented in foresters' yield tables, is important in the context of sustainable management because it is a key determinant of forest rotation length. Nevertheless, the phenomenon remains inadequately understood at the process level. The classical explanation, that maintenance respiration consumes an ever-increasing proportion of photosynthates as trees grow larger (Kira and Shidei 1967), appears not to stand up to experimental scrutiny (Ryan and Waring 1992; Ryan et al. 1995). Alternative hypotheses have been proposed, invoking hydraulic (Ryan and Yoder 1997) and/or nutritional constraints (Murty et al. 1996) on photosynthesis rates per unit foliage area, and on growth allocation to foliage area, during stand development.

### Pipe-model hypothesis

The latter hypothesis, that the age-related growth decline is associated with a decline in foliage allocation, was first examined within the context of a growth model by Mäkelä (1986), using the pipe-model theory (Shinozaki et al. 1964) according to which the sapwood area: foliage area ratio,  $A_{sw}/A_f$ , is a constant. Mäkelä (1986) showed how maintenance of a constant ratio  $A_{sw}/A_f$  leads to a decline in tree growth rate with height, due to the ever-increasing carbon demand by (non-productive) sapwood growth as the pipe length increases.

However, observed variations in  $A_{sw}/A_f$  in trees led Whitehead et al. (1984) to examine in more detail the theoretical behaviour of  $A_{sw}/A_f$  expected from water balance considerations (see Cannell and Dewar 1994). They considered the steady-state balance between water transport within trees and the rate of foliage transpiration. On a leaf-specific basis this may be written as

$$(\psi_s - \psi_f)K_L = g_s D \quad (1a)$$

where:  $\psi_s$  and  $\psi_f$  are the soil and foliage water potentials,  $K_L$  is the leaf-specific hydraulic conductivity between soil and foliage,  $g_s$  is the leaf stomatal conductance, and  $D$  is the air water vapour pressure deficit. In the simplest analysis  $K_L \propto A_{sw}/hA_f$  where  $A_{sw}$  is the sapwood cross-sectional area,  $h$  is tree height (or, more correctly, the hydraulic path length of the xylem), and  $A_f$  is leaf area. Equation 1a then implies:

$$\frac{A_{sw}}{A_f} \propto \frac{g_s D h}{\psi_s - \psi_f} \quad (1b)$$

so that, in contrast to the pipe model, one might expect  $A_{sw}/A_f$  to increase with tree height. A more complete analysis would include the contribution to  $K_L$  from the hydraulic conductivity between the soil and the roots (Magnani et al. 2000). An increase in  $A_{sw}/A_f$  with tree height

would lead to a more rapid height-related decline in tree growth than for the pipe model (all else being equal), with potential consequences for the length of forest management cycles.

### Hydraulic homeostasis

The conclusion from Equation 1b that  $A_{sw}/A_f$  increases with tree height is not quite so straightforward, because this depends on the associated responses of  $g_s$  and  $\psi_f$ . However, it is observed for certain species (especially trees) that the diurnal minimum foliage water potential ( $\psi_f$ ) is rather constant over a range of soil moisture conditions and plant development stages (Oren et al. 1999; Magnani et al. 2000). Hydraulic homeostasis in trees appears to be regulated on diurnal timescales by stomatal conductance (Schäfer et al. 2000). Such behaviour has been interpreted in terms of plant avoidance of catastrophic xylem cavitation (Tyree and Sperry 1988; Sperry and Pockman 1993).

Whatever its functional significance, hydraulic homeostasis simplifies the analysis of water balance relationships such as Equation 1b (Magnani et al. 2000). With constant  $\psi_f$  (and fixed  $D$  and  $\psi_s$ ), Equation 1b reduces to:

$$\frac{A_{sw}}{A_f} \propto g_s h \quad (1c)$$

Thus, as tree height increases, the tree must either increase its sapwood area: foliage area ratio (Magnani et al. 2000), or decrease its stomatal conductance (Yoder et al. 1994), or both (Schäfer et al. 2000). Height-related changes in sapwood permeability (an implicit factor in  $K_L$ ) may also occur (Pothier et al. 1989a; 1989b). The larger the increase in  $A_{sw}/A_f$  with height, the smaller the associated decrease in  $g_s$ . Either response would lead to a decline in growth rate, through reductions in total leaf area and photosynthesis per unit leaf area, respectively.

For example, note that the pipe model hypothesis (constant  $A_{sw}/A_f$ ) is not inconsistent with water balance (Equation 1b) and hydraulic homeostasis (Equation 1c), provided that it is associated with a strong decline in stomatal conductance with height (i.e.  $g_s \propto 1/h$ ). At the other extreme, if  $g_s$  were independent of height then Equation 1c would imply that  $A_{sw}/A_f \propto h$ . Thus, hydraulic homeostasis permits a continuous family of dependences of  $A_{sw}/A_f$  on tree height, depending on height-related changes in stomatal conductance (and sapwood permeability).

The implication here is that the age-related growth decline in trees (and its management consequences) cannot be quantified using hydraulic homeostasis alone; one needs an additional relationship between either  $g_s$  or  $A_{sw}/A_f$  and tree height. Changes in sapwood permeability with height also need to be taken into account. These are future challenges for modellers.

### Combining hydraulic and nutritional constraints on growth allocation

Mäkelä (1986) combined the pipe-model constraint between sapwood and foliage allocation with a functional balance constraint between the assimilation of carbon and nitrogen (Davidson 1969), thus constraining foliage-root allocation. The combined model describes both hydraulic and nutritional constraints on growth allocation to foliage, sapwood and roots, although Mäkelä (1986) did not take into account the associated decline in stomatal conductance implied by water balance considerations ( $g_s \propto 1/h$ , Equation 1c).

As noted above, an alternative to the pipe model would be to assume that  $g_s$  is independent of height, leading to  $A_{sw}/A_f \propto h$  (Equation 1c). One could then combine this latter relationship (rather than the pipe model) with functional balance. This has been done by the present author in the SUSTAIN model of C-N-H<sub>2</sub>O interactions in trees. Table 1 summarises the key assumptions of SUSTAIN (in its simplest version for constant environmental conditions); full mathematical details will be given elsewhere (Dewar, in preparation). Note that the expression used for the water-use efficiency ( $WUE$ , Table 1) implies a constant (height-independent) ratio of intercellular to leaf surface CO<sub>2</sub> concentration, reflecting the underlying assumption that  $g_s$  is independent of height.

**Table 1.** Processes and their description in the SUSTAIN model, in the version for constant environmental conditions.

Process	Description in the model
Absorbed radiation ( $APAR$ )	Beer's law function of foliage biomass
Gross primary productivity ( $GPP$ )	$GPP = LUE \times APAR$ ; $LUE$ = light-use efficiency (a saturating function of $[CO_2]$ ).
Respiration	Fixed fraction of $GPP$ .
Plant N uptake ( $U$ )	$U = \sigma_N \times R$ ; $\sigma_N$ = specific root activity, $R$ = fine root biomass.
C allocation to foliage, sapwood and fine roots	Constrained by hydraulic homeostasis and functional balance: $(\psi_s - \psi_f)kA_{sw}/h = E_c = GPP/WUE$ , $U = v \times GPP$ . $\psi_s$ = soil water potential, $\psi_f$ = leaf water potential (constant), $k$ = sapwood specific hydraulic conductivity, $A_{sw}$ = sapwood cross-sectional area, $h$ = sapwood height (sapwood biomass $\propto A_{sw} \times h$ ), $E_c$ = canopy transpiration, $WUE$ = water-use efficiency $\propto [CO_2]/D$ ( $D$ = air saturation deficit), $v$ = mean plant N:C ratio (constant).
Height growth	Proportional to new foliage growth.
Senescence	Constant rates per biomass for foliage, sapwood and fine roots.

**Table 2.** Sensitivity analysis of the SUSTAIN model under various constant environments. Base case: root specific N uptake  $\sigma_N = 0.02$  kg N (kg fine root C)<sup>-1</sup> yr<sup>-1</sup>, atmospheric CO<sub>2</sub> concentration = 350 ppm, air saturation deficit  $D = 1$  kPa. For each of scenarios 1–3, one of these variables is doubled while the other two are fixed at their base values.  $t_c$  = canopy closure time (when LAI has a maximum),  $LAI_{max}$  = maximum leaf area index (at canopy closure),  $h_{10}$  = tree height at 10 yrs (site index),  $T_{rot}$  = rotation period (when MAI has a maximum),  $MAI_{max}$  = maximum MAI.

Scenario	$t_c$ (yr)	$LAI_{max}$ (m <sup>2</sup> m <sup>-2</sup> )	$h_{10}$ (m)	$T_{rot}$ (yr)	$MAI_{max}$ (m <sup>3</sup> ha <sup>-1</sup> yr <sup>-1</sup> )
Base case	9	4.1	11	37	11
1. $\sigma_N \times 2$	6	5.4	17	27	19
2. $D \times 2$	8	2.9	8	31	10
3. $CO_2 \times 2$	8	5.9	16	44	15

Table 2 illustrates the predicted sensitivity of some common forestry variables to environmental conditions. The positive growth response to a doubling of the specific root activity ( $\sigma_N$ , a measure of soil inorganic N availability) reflects an increase in foliage/root allocation due to functional balance; there is no change in the light-use efficiency, *LUE*, which is assumed to be independent of soil fertility (Table 1).

The negative growth response to a doubling of air saturation deficit (*D*) reflects a decrease in foliage/sapwood allocation due to hydraulic homeostasis; *LUE* is assumed to be independent of *D* (like  $\sigma_N$ ).

Increasing atmospheric [ $\text{CO}_2$ ] leads to increases in both light- and water-use efficiencies (the effect on *WUE* being dominant). As a result, foliage/sapwood allocation is increased through hydraulic homeostasis, but foliage/root allocation is decreased through functional balance. The net effect (for the parameter values used in Table 2) is a higher foliage allocation (relative to the base case) up to canopy closure and a lower foliage allocation thereafter; the latter response, however, is offset by the  $\text{CO}_2$ -stimulated increase in *LUE*, leading to an overall positive growth response.

These simulations illustrate the way in which simple hydraulic and nutritional constraints on growth allocation may be combined. Due to its simplicity, the growth responses of the model can be interpreted relatively easily in terms of the underlying processes.

The above scheme could be extended to explore the consequences of different assumptions for the  $g_s$ -height relationship, intermediate between the extremes represented by SUSTAIN and the pipe model. For example, does the trade-off between height-related changes in  $A_{sw}/A_f$  and  $g_s$  (Equation 1c) mean that the decline in tree growth with stand age is relatively insensitive to what one assumes about their individual height dependences?

SUSTAIN is also being developed to incorporate temporal variation in environmental conditions, and to couple the tree growth model to a dynamic model of C, N and  $\text{H}_2\text{O}$  balance in the soil, following an approach developed previously (Dewar 1997).

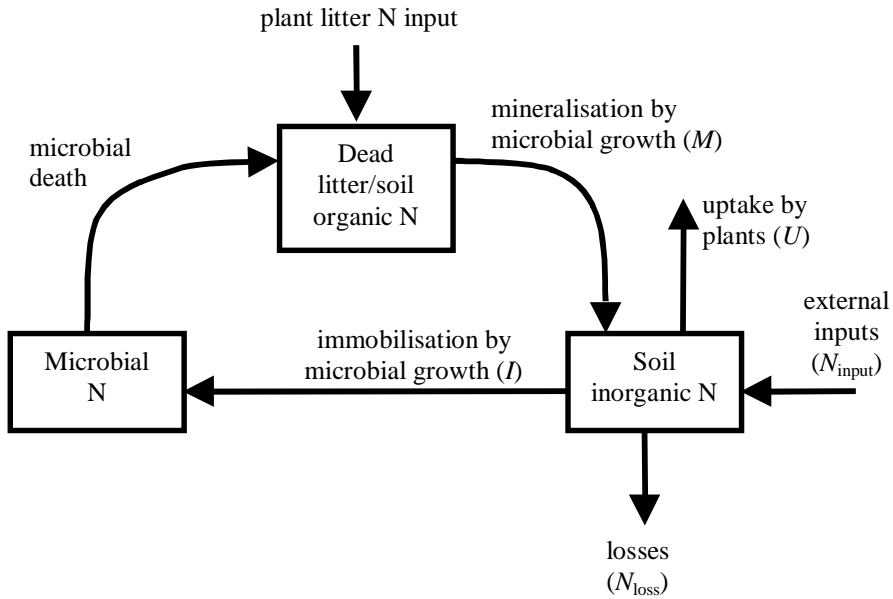
## Modelling Soil Nutrient Cycling

The models of soil nutrient cycling described in this Proceedings differ in their characterisation of litter and soil organic matter (SOM), i.e. the number of pools represented, and their physico-chemical properties. GRAECO (Loustau et al. 2001) contains a single soil layer. ASPECTS (Rasse et al. 2001) has 2 pools: litter and SOM. G'DAY (McMurtrie et al. 2001b) uses the CENTURY decomposition model (Parton et al. 1993) with 4 litter pools (structural and metabolic, both above- and below ground), 3 SOM pools (active, slow and passive) and a soil inorganic N pool. MIT (Corbeels et al. 2001) has a CENTURY-like structure with an additional pool for microbial biomass. EFM (Cannell and Thornley 2001) also includes a soluble C pool and separates soil inorganic N into ammonium and nitrate.

In this section, however, I have chosen to compare some of these models in terms of their characterisation of microbial growth. This choice is motivated by the fact that microbial growth is a key process for 3 major aspects of sustainable forest management: forest productivity, water quality and air quality.

### The key role of microbial growth

Figure 2 shows the main pools and fluxes of nitrogen (N) in the soil. The rate of change of soil inorganic N ( $N_{\text{inorg}}$ ) is the balance between gross mineralisation (*M*) and immobilisation



**Figure 2.** The principal pools and fluxes of nitrogen in the litter/soil. The key pool for sustainable forest management is soil inorganic N, which regulates both plant N uptake ( $U$ ) (hence productivity) and N losses ( $N_{\text{loss}}$ ) (hence water and air quality). The key soil process regulating soil inorganic N is microbial growth, which drives mineralisation ( $M$ ) and immobilisation ( $I$ ), and thus the internal cycling of N between soil inorganic N, microbial biomass and dead soil organic matter. Two additional fluxes – abiotic incorporation of inorganic N into existing soil organic matter, and conversion of inorganic to organic N by mycorrhizae without biomass production – are not included but may also be important (Aber et al. 1998).

( $I$ ) through microbial growth, external N inputs by fertiliser application, atmospheric deposition and fixation ( $N_{\text{input}}$ ), losses through leaching and gaseous emissions ( $N_{\text{loss}}$ ) and plant uptake ( $U$ ):

$$\frac{dN_{\text{inorg}}}{dt} = M - I + N_{\text{input}} - N_{\text{loss}} - U \quad (2a)$$

The fluxes of direct interest to sustainable management are  $U$  (affecting forest productivity – see Table 2, scenario 1) and  $N_{\text{loss}}$  (affecting water and air quality). Both of these fluxes depend to a large extent on the soil inorganic N pool, which is therefore the central pool of interest. The key soil process regulating soil inorganic N is microbial growth, which drives mineralisation and immobilisation, and thus the internal N cycling between soil inorganic N, microbial N and dead soil organic N (Figure 2).

It would seem advantageous, therefore, to model microbial biomass and growth explicitly, in order that the processes of mineralisation and immobilisation can be adequately described. This approach was first elaborated by Parnas (1975), who developed a simple model of SOM

decomposition, based on the assumption that the rate of decomposition of any substrate is proportional to the growth rate of its decomposers.

To illustrate this approach and its relevance to sustainable forest management, let us assume for simplicity that  $N_{\text{loss}}$  and  $U$  are directly proportional to the soil inorganic N pool:  $N_{\text{loss}} = \lambda_{\text{loss}} N_{\text{inorg}}$  and  $U = \lambda_u N_{\text{inorg}}$  ( $\lambda_u$  might depend on fine root biomass,  $R$  – see Table 1). Also, we may write  $M = v_{\text{SOM}} G_m / Y_m$  and  $I = v_m G_m$ , where  $G_m$  is the microbial growth rate,  $Y_m$  is the microbial growth efficiency, and  $v_{\text{SOM}}$  and  $v_m$  are the N:C ratios of soil organic matter and microbial biomass, respectively. Equation 2a then becomes:

$$\frac{dN_{\text{inorg}}}{dt} = \left( \frac{v_{\text{SOM}}}{Y_m} - v_m \right) G_m + N_{\text{input}} - (\lambda_{\text{loss}} + \lambda_u) N_{\text{inorg}} \quad (2b)$$

Because  $N_{\text{inorg}}$  is a relatively small, fast-turnover pool compared with the other pools in the soil, it may be considered to be in an approximate, ‘moving’ steady state which tracks the slower evolution of the rest of the system. Then, at any one time the value of  $N_{\text{inorg}}$  is given by solving  $dN_{\text{inorg}}/dt = 0$  using Equation 2b. Doing this, the corresponding rates of plant N uptake and N loss can be written as:

$$U \approx \frac{\lambda_u}{\lambda_{\text{loss}} + \lambda_u} \left\{ \left( \frac{v_{\text{SOM}}}{Y_m} - v_m \right) G_m + N_{\text{input}} \right\} \quad (2c)$$

and

$$N_{\text{loss}} \approx \frac{\lambda_{\text{loss}}}{\lambda_{\text{loss}} + \lambda_u} \left\{ \left( \frac{v_{\text{SOM}}}{Y_m} - v_m \right) G_m + N_{\text{input}} \right\} \quad (2d)$$

where:  $\approx$  indicates the moving steady-state approximation. Equations 2c and 2d show explicitly the key dependence of plant N uptake and N loss on microbial growth rate ( $G_m$ ).

### Modelling microbial growth

In view of these remarks, it is useful to compare models in their characterisation of the key process of microbial growth ( $G_m$ ).

In CENTURY, the active pool represents both live microbial biomass and dead microbial products; live microbial biomass is not explicitly represented. The decomposition rate of each litter and SOM pool (including the active pool) is proportional to its C content (with soil moisture and temperature modifiers). Microbial growth is represented as the production of new active SOM, and is a linear combination of the C contents of the remaining six C pools. This approach, which is also used in the soil sub-model of G'DAY (McMurtrie et al. 2001b), assumes that microbial growth is limited by C substrate alone, and not by soil inorganic N availability or microbial biomass.

The MIT model (Corbeels et al. 2001) represents microbial biomass explicitly. N-limitation of microbial growth is introduced *via* a decomposition rate modifier which depends (in the notation of Figure 2) on the N flux ratio  $(M+N_{\text{input}}-N_{\text{loss}}-U)/I$  representing the supply of inorganic N relative to microbial N demand. As in CENTURY, microbial biomass itself is assumed not to limit microbial growth. The dependence of microbial growth on a flux ratio is a useful empirical device, although from a mechanistic point of view it may be more appropriate to relate microbial growth to the soil inorganic N pool.

Thornley and Verberne (1989) modelled microbial growth rate ( $G_m$ ) as a function of the soil organic C pool ( $C_{\text{SOM}}$ ), the soil inorganic N pool ( $N_{\text{inorg}}$ ), and microbial biomass ( $B$ ) (their Equation 16). They assumed that  $G_m$  is a Michaelis-Menten function of  $C_{\text{SOM}}$  and  $N_{\text{inorg}}$ , and is proportional to  $B^q$ , where  $q = 1$  and  $2/3$  represent autocatalytic growth and surface-limited growth, respectively. They also assumed that  $B$  has a ceiling value that is proportional to  $C_{\text{SOM}}$ , the rationale being that soil microbes must physically attach themselves to their substrates in order to grow, and the number of attachment sites increases with  $C_{\text{SOM}}$ . This model provides a simple yet flexible method of combining the main factors regulating microbial growth.

In a further refinement, Thornley (1998) introduced a soluble C pool, representing labile C metabolites such as glucose, other sugars and organic acids. Microbial growth is then a function of this soluble C pool rather than of SOM carbon. This approach, which is also used in the Edinburgh Forest Model (Cannell and Thornley 2001), allows other processes such as root exudation and C leaching to be modelled. Thornley (1998, p.78) remarked, however, that in many respects the resulting increase in complexity, in comparison with the earlier model of Thornley and Verberne (1989), made little difference to the behaviour of the system, and that 'the soil appears to be relatively passive in its responses compared with the very plastic responses of the plant to the environment'. This conclusion suggests that a relatively simple model of microbial growth (e.g. Thornley and Verberne 1989) suffices to capture the essential features of soil N cycling.

However, non-microbial immobilization processes may also be important in temperate forest ecosystems (Aber et al. 1998). These include abiotic incorporation of inorganic N into existing SOM, and conversion of inorganic to organic N by mycorrhizae without microbial biomass production. The long-term effect of these processes on soil N cycling in forest ecosystems remains to be explored by modellers.

## Microbial diversity

Two additional key variables in Equations 2c and 2d are the N:C ratios of microbial biomass and dead soil organic matter (respectively,  $v_m$  and  $v_{\text{SOM}}$ ).

The CENTURY model (Parton et al. 1993) assumes an empirical relationship between the N:C ratio of newly-formed SOM and the soil inorganic N pool ( $N_{\text{inorg}}$ ), reflecting underlying shifts in microbial species composition with soil N availability. However, the parameters of this relationship are poorly known, and sensitivity analysis of the G'DAY model shows that this uncertainty presents a major limitation for predicting long-term trends in the carbon sink strength of forests (McMurtrie et al. 2001a).

The empirical link between SOM N:C ratios and  $N_{\text{inorg}}$  might be explored by modellers at a more mechanistic level. Following the approach of Parnas (1975), for example, one might represent several microbial pools – one for each substrate – with different (but fixed) N:C ratios and with growth rates which differ in their dependence on  $N_{\text{inorg}}$ .

In summary, current soil nutrient cycling models serve to highlight the uncertainty concerning the regulation of microbial diversity, and to explore the long-term implications of that uncertainty. Modellers should also explore the role of non-microbial immobilization processes (Aber et al. 1998). Ultimately, progress on these fronts will require further experimental studies.



## Knowledge Transfer

Tomé et al. (2001) and Pinto et al. (2001) each describe a decision-support tool designed to find the management regime (e.g. cutting cycle, labour scheduling plan) that achieves a certain objective (e.g. maximum net present value, minimum total cost) subject to various constraints (e.g. even wood production, even workforce distribution). A common feature of these systems is that they are set up to solve constrained optimization problems. How can process-based models contribute here?

Tomé et al. showed how a conventional growth and yield model (GLOBULUS) for *Eucalyptus globulus* has been incorporated into the CEF decision-support system for forest management in Portugal (Figure 1, flow 'a'). Due to its modular nature, the GLOBULUS-CEF system may provide a useful framework into which knowledge from process-based models may be incorporated. In this regard, however, a potentially serious limitation of detailed, process-based models is their high data input requirement (see also Discussion by Dhôte and Hervé 2001).

In this section I examine four ways in which this problem is currently being addressed (Figure 1, flows 'b–e'). The first three of these (growth indices, process-based yield predictions, model simplification) aim to predict growth and yield from process-based models using readily available data only. The fourth approach (mass balance constraints) aims to provide general insights and principles for sustainable forest management that are to some extent independent of the model parameters.

### Growth indices (Figure 1, flow 'b')

In this approach, detailed, process-based models are used in conjunction with readily available input data, to derive summary growth indices that capture temporal and spatial variations in climate and soil properties, for example. The growth indices are then used to modify the parameters of empirical growth and yield models, with the aim of improving their predictive power.

For example, using the detailed, process-based model BIOMASS (McMurtrie et al. 1990), Snowdon et al. (1999) calculated annual photosynthetic carbon fixation ( $P$ ) for *Pinus radiata* D. Don grown in the Australian Capital Territory (ACT). They assessed the use of  $P$  as a growth index to improve the Schumacher yield model (Schumacher 1939). Significant improvement in the descriptive power of the Schumacher model was obtained when the maximum yield parameter and time variable in that model were modified as functions of  $P$ . Because climatic data across the ACT are highly correlated,  $P$  calculated at a single location was able to capture the annual variability for the entire region. Spatial variability was taken into account through indices of site fertility, geological substrate and soil water-holding capacity, allowing further improvements to be made.

### Process-based yield prediction and model simplification (Figure 1, flows 'c' and 'd')

Process-based models may be used to predict yield directly (Figure 1, flow 'c') as an alternative to empirical yield models (Figure 1, flow 'a'). In order to reduce the input data requirements, some model simplification is necessary (Figure 1, flow 'd'), with only the essential features of the underlying processes being retained. Again, the aim is to improve on the predictive power of existing empirical yield models. Of course, all models (including so-called detailed ones) are simplifications of reality.

One advantage of using simplified, process-based models (Figure 1, flow 'c') over the use of growth indices within conventional yield models (Figure 1, flow 'b') is that, by construction, process-based models satisfy the constraint of mass conservation. This feature constitutes one of the crucial differences between process-based models and empirical growth and yield models. The constraint of mass conservation is especially important for the assessment of sustainable forest management, where the criteria include consideration not only of wood yield, but also of carbon storage, nutrient loss and other ecosystem properties whose prediction demands a rigorous account of carbon, nutrient and water balance (see below). Growth indices are not sufficient for this purpose.

The 3-PG model is an example of a simple, process-based yield predictor, which leads to improved estimates of growth and yield compared with conventional models (Landsberg et al. 2001). Such models may be derived or assessed from a more mechanistic basis using more detailed models. For example, detailed canopy radiation interception models can be used to assess simpler 'big-leaf' approximations such as Beer's law. The simplifying concepts of light and carbon use efficiencies have also been assessed in this way (Dewar et al. 1998).

The 3-PG model takes into account plant carbon and water balance only. GRAECO (Loustau et al. 2001) and ASPECTS (Rasse et al. 2001) include carbon and water balances in the plant and soil. When considering the inclusion of nutrient balance, one may question whether model simplification continues to be feasible, or whether the complexity of C-N-H<sub>2</sub>O cycling in trees and soil (as represented in the Edinburgh Forest Model, for example) is an essential feature that cannot be avoided. Analysis of models such as SUSTAIN and G'DAY suggest that it may be possible to simplify C-N-H<sub>2</sub>O interactions in the plant and soil while retaining a level of biological realism that is adequate for many applications.

### Mass balance constraints (Figure 1, flow 'e')

As noted above, a key feature of process-based models is that, by construction, they satisfy mass conservation. Mass balance imposes a powerful constraint on sustainable management, and is especially useful in analysis of the long-term, quasi-equilibrium behaviour of the system. For this reason, sustainable behaviour is often defined by modellers as the behaviour in the steady state (Dewar and McMurtrie 1996; Thornley and Cannell 2000; Cannell and Thornley 2001; McMurtrie et al. 2001b). What, then, can steady-state mass balance tell us?

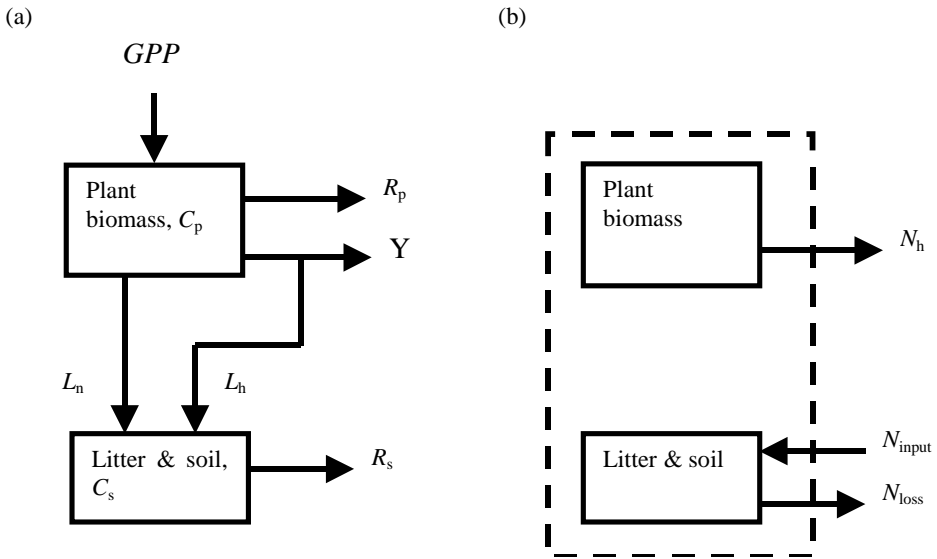
Let us imagine the hypothetical situation where a certain forest management regime is applied indefinitely over successive rotations to a given area of land. The system might also be subjected to periodic disturbance by storms and herbivory, for example, but I will exclude long-term climate change here (see Loustau et al. 2001; Rasse et al. 2001).

After a transient that depends on the initial state, the system will eventually tend to fluctuate around a time-averaged steady state. In this steady state, the amounts of C, N and H<sub>2</sub>O in the various tree, litter and soil compartments, when averaged over a management or disturbance cycle, remain constant over time. By definition, the steady-state time-averaged fluxes of C, N and H<sub>2</sub>O into each compartment are equal to the corresponding fluxes out. The steady state (i.e. sustainable) properties will depend on the management regime applied.

The steady state might seem of limited practical interest, given that it may take ecosystems centuries to reach equilibrium (McMurtrie et al. 2001b). Nevertheless, steady-state analysis is useful for several reasons. Firstly, steady-state analysis forces us to take a long-term view of ecosystem behaviour (implicit in the word 'sustainable'). Secondly, steady-state analysis helps us to understand the intrinsic nature of the link between ecosystem behaviour and management practices, independent of transient effects. Thirdly, the steady-state balance of fluxes yields simple but powerful constraints on ecosystem behaviour, leading to useful

insights into the relationship between sustainable yield, ecosystem carbon storage and ecosystem N loss rates (see below). Fourthly, on certain timescales it is possible to divide the ecosystem into ‘fast’ and ‘slow’ pools (as in Equations 2b–c), whereby the steady-state approximation can be applied to the fast-turnover pools, even when the ecosystem as a whole has not reached a steady state (McMurtrie et al. 1992). Lastly, the steady state may serve as a practical guide to where the system is heading at any one time.

Figure 3 shows the major C and N pools and fluxes in a forest plantation ecosystem. In the steady state, the balance of plant C fluxes is (Figure 3a):



**Figure 3.** (a) Major carbon pools ( $\text{kg C m}^{-2}$ ) and fluxes ( $\text{kg C m}^{-2} \text{ yr}^{-1}$ ) in a forest plantation.  $C_p$  = plant C,  $C_s$  = litter and soil C,  $GPP$  = gross primary productivity,  $R_p$  = plant respiration,  $Y$  = C removed in harvested products,  $L_n$  = natural plant litter production,  $L_h$  = slash input,  $R_s$  = soil respiration. (b) External fluxes of nitrogen ( $\text{kg N m}^{-2} \text{ yr}^{-1}$ ) into and out of a forest plantation (outer box).  $N_h$  = N removed in harvested products.  $N_{\text{input}}$  = N deposition, fixation and fertilisation.  $N_{\text{loss}}$  = N leaching and gaseous emissions. In the steady state, C and N fluxes into each box balance the corresponding fluxes out, with implications for the relationship between  $Y$ , ecosystem carbon storage ( $C_p + C_s$ ) and  $N_{\text{loss}}$  (see text).

$$\langle GPP \rangle = \langle Y \rangle + \langle R_p \rangle + \langle L_n \rangle + \langle L_h \rangle \quad (3a)$$

where:  $GPP$  is gross primary production (*i.e.* photosynthesis),  $Y$  is the removal rate of C in forest products,  $R_p$  is plant respiration,  $L_n$  is the rate of natural litter production (biomass turnover + plant mortality),  $L_h$  is the rate of slash input to the soil from thinning and harvest debris, and  $\langle \dots \rangle$  denotes the time-average over a management or disturbance cycle. In Equation 3a,  $\langle Y \rangle$  represents the steady-state (*i.e.* sustainable) yield.

The steady-state balance of soil C fluxes is:

$$\langle L_n \rangle + \langle L_h \rangle = \langle R_s \rangle \quad (3b)$$

where:  $R_s$  is soil respiration. Finally, the steady-state balance of ecosystem N fluxes is (Figure 3b):

$$\langle N_{\text{input}} \rangle = \langle N_h \rangle + \langle N_{\text{loss}} \rangle \quad (3c)$$

where:  $N_{\text{input}}$  is external inputs from N deposition, fixation and fertiliser application,  $N_h$  is the removal rate of N in harvested forest products, and  $N_{\text{loss}}$  is the rate of N loss through leaching and gaseous emissions.

These relationships, being based on mass balance alone, are useful because they provide constraints on the C and N fluxes in the steady state that do not depend on the details of the underlying mechanisms involved. The following simple example illustrates how the steady-state balance of fluxes can provide useful constraints on the sustainable yield and its relationship to other quantities in the steady state.

Let  $C_p$  and  $C_s$  denote the plant and soil carbon contents. Also let  $k_h$  denote the fraction of plant biomass harvested each year, and  $\beta$  the fraction of harvested biomass that is then removed as forest products (the remaining fraction being left on site as slash debris). Then the rate of C removal in forest products is  $Y = \beta k_h C_p$  and the rate of slash input to the soil is  $L_h = (1-\beta)k_h C_p$ . We now make 3 simplifying assumptions: (i) natural litter production and soil respiration follow first-order kinetics,  $L_n = k_n C_p$  and  $R_s = k_s C_s$ , where  $k_n$  and  $k_s$  are rate constants; (ii) plant respiration is proportional to gross primary productivity,  $R_p = (1-\phi)GPP$ , where  $\phi$  is the ratio of net to gross primary productivity; and (iii) gross primary productivity is a saturating function of plant biomass,  $GPP = G_{\text{max}} C_p / (C_p + C_0)$ , where  $G_{\text{max}}$  is the maximum  $GPP$  (corresponding to 100% absorption of incident radiation) and  $C_0$  is the plant biomass at which  $GPP$  is 50% of  $G_{\text{max}}$ . This last assumption is a crude numerical approximation, but it captures the essential feature for the conclusion that follows.

Substituting these assumptions into Equations 3a and 3b, and solving for the steady-state carbon contents  $C_p$  and  $C_s$ , it is straightforward to show that the sustainable ecosystem carbon storage ( $EC = C_p + C_s$ ) and the sustainable yield ( $Y = \beta k_h C_p$ ) are given by:

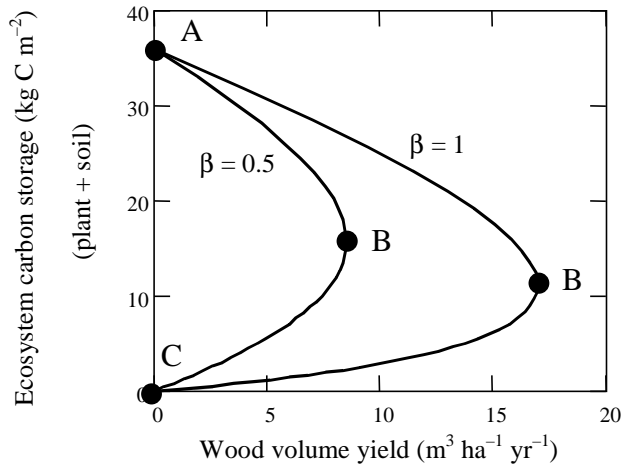
$$\langle EC \rangle = \left\{ 1 + \frac{k_n + (1-\beta)k_h}{k_s} \right\} \left( \frac{\phi G_m}{k_n + k_h} - C_0 \right) \quad (4a)$$

and

$$\langle Y \rangle = \beta k_h \left( \frac{\phi G_m}{k_n + k_h} - C_0 \right) \quad (4b)$$

Figure 4 shows the predicted relationship between  $\langle EC \rangle$  and  $\langle Y \rangle$  as the biomass harvest fraction  $k_h$  is increased from 0 to 40%  $\text{yr}^{-1}$ , for two values of the harvest removal fraction

( $\beta = 50\%$  and  $100\%$ ). The figure illustrates that there is no simple inverse relationship between carbon stored and wood yield, in accordance with the simulations of the Edinburgh Forest Model presented by Cannell and Thornley (2001). Rather, starting from an unmanaged forest ( $k_h = 0$ ) at which carbon storage is maximal and wood yield is zero (Figure 4, point A), the relationship is inverse only up to a critical harvest fraction (given by  $k_h = 12\% \text{ yr}^{-1}$  in the example shown in Figure 4, point B) at which the maximum possible wood yield is obtained. Beyond this point, carbon storage and wood yield decline together due to the reduction in photosynthesis by heavy thinning, reaching zero at another critical value of  $k_h$  (equal to  $40\% \text{ yr}^{-1}$  in the example shown in Figure 4, point C) which represents the limit of sustainability. Comparison of Figure 4 with Figure 5 of Cannell and Thornley (2001) suggests that point B is analogous to the optimal wood harvesting rate of  $10\text{--}20\% \text{ yr}^{-1}$  obtained using the Edinburgh Forest Model.



**Figure 4.** Relationship between steady-state ecosystem carbon storage ( $EC$ ) and wood volume yield ( $Y$ ) (Eqns 4a-b), as the harvest fraction ( $k_h$ ) increases from 0 to  $40\% \text{ yr}^{-1}$ , for two values of the removal fraction ( $\beta$ ). Other parameter values:  $G_{\max} = 3 \text{ kg C m}^{-2} \text{ yr}^{-1}$ ,  $C_0 = 3 \text{ kg C m}^{-2}$ ,  $\phi = 0.5$ ,  $k_n = 0.1 \text{ yr}^{-1}$ ,  $k_s = 0.05 \text{ yr}^{-1}$ , wood density =  $270 \text{ kg C m}^{-3}$ . Point A: unmanaged forest ( $k_h = 0$ ) with maximum carbon storage and zero wood yield. Point B: forest managed for maximum yield; further analysis shows this to occur when the ratio  $k_h/k_n$  is equal to  $(1/\alpha) - 1$  where  $\alpha = k_n C_0 / \phi G_{\max}$  (giving  $k_h = 12\% \text{ yr}^{-1}$  here). Point C: unsustainable forest for which carbon storage and wood yield are both zero; this occurs when  $k_h/k_n$  is equal to  $(1/\alpha) - 1$  (giving  $k_h = 40\% \text{ yr}^{-1}$  here). The relationship between  $EC$  and  $Y$  depends on  $\beta$ , but the values of  $k_h$  at points A, B and C are independent of  $\beta$ . The present Figure may be compared with Figure 5 of Cannell and Thornley (2001).

The steady-state N loss rate through leaching and gaseous emissions can be calculated from the N flux balance (Equation 3c). Denoting the average N:C ratio of forest products by  $v_h$ , we have  $\langle N_h \rangle = v_h \langle Y \rangle$ , so that Equation 3c implies that:

$$\langle N_{\text{loss}} \rangle = \langle N_{\text{input}} \rangle - v_h \langle Y \rangle \tag{4c}$$

Thus, for a fixed N input rate, harvesting for maximum sustainable yield (as at point B in Figure 4) also minimises the sustainable N losses (in accordance with Cannell and Thornley 2001). For example, with average atmospheric N inputs of  $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ , a N:C ratio of forest products (wood) of  $v_h = 0.001 \text{ kg N kg}^{-1} \text{ C}$ , a harvest removal fraction of  $\beta = 100\%$ , and other parameters as in Figure 4, Equations 4b–c predict that the minimum rate of N loss (obtained with  $k_h = 12\% \text{ yr}^{-1}$ ) represents 54% of the total N output (*i.e.*  $5.4 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ), the remaining 46% being removed in harvested products. With  $\beta = 50\%$ , the minimum N loss increases to 77% as more plant N is transferred to the soil in slash debris.

The conclusions from this simple model are in broad agreement with simulations of the Edinburgh Forest Model (Cannell and Thornley 2001), essentially because these conclusions reflect mass balance constraints (which all process-based models satisfy by construction) and the saturating dependence of *GPP* on plant biomass (which all reasonable process-based models mimic). Thus, steady-state mass balance analysis can contribute general insights into sustainable forest management that are to some extent independent of the model parameters.

Of course, mass balance is a general constraint that also applies to non-steady state conditions. For example, McMurtrie and Dewar (1997) used a mass balance model (N-BAL), in conjunction with the Schumacher equation for stem volume growth, to evaluate the balance between N removals (due to harvesting and fire) and N inputs (both natural and as added fertilizer) in managed stands of karri (*Eucalyptus diversicolor*) in southwestern Australia. Their analysis led to a criterion for estimating the depletion (or accretion) of site N over a single forest rotation (effectively, a non-steady state analogue of Equation 4c).

Relatively robust relationships between wood yield, carbon storage and N loss, derived from mass balance, might be incorporated as additional constraints within established decision-making systems, such as that developed by CEF (Figure 1, flow 'e'). Thus, net present value could be maximised subject to constraints that include the environmental criteria of sustainability, not just even wood flow.

## Summary and Recommendations

I have tried to give some indication of the actual and potential contribution of modellers to the development of sustainable forest management. The challenges are both scientific and practical in nature.

On the scientific side, the last 10 years has seen significant progress in the development of detailed forest ecosystem models based on a comprehensive description of plant-soil and carbon-nutrient-water interactions. By exploring 'what-if' scenarios, these models have helped us to better understand how management practices might affect such aspects of ecosystem behaviour as tree growth, soil fertility and carbon storage. Through sensitivity analyses, they have also helped to identify key uncertainties for further study. Two key processes for sustainable forest management – plant growth allocation and soil nutrient cycling – continue to challenge modellers.

Hydraulic homeostasis has recently emerged as a guiding principle for modelling growth allocation. Combining this with root-shoot functional balance leads to a simple scheme incorporating both hydraulic and nutritional constraints on allocation for use in simplified, process-based growth models. While this allocation scheme predicts realistic growth and yield trends with stand age, the individual roles played by allocation and stomatal conductance require further study. Hydraulic homeostasis alone cannot separate these.

Soil nutrient cycling models differ in how they describe the regulation of microbial growth and diversity – two key processes for sustainable management. Existing models of the former probably suffice (at least for N cycling), but uncertainty in the latter presents a major limitation for predicting long-term ecosystem behaviour; further experimental work is required here. The role of non-microbial immobilization processes should also be explored in models.

On the practical side, mensuration-based growth and yield models have been incorporated into decision-support tools designed to find management practices that satisfy mainly economic objectives (e.g. maximum net present value), subject to various constraints (e.g. even wood production). Information from process-based models has yet to make a significant contribution.

Three options for enhancing the information flow from process-based models to decision-support tools are: (i) use of process-based growth indices to improve conventional growth and yield models; (ii) simplification and direct incorporation of process-based growth models; and (iii) use of mass balance analysis to generate robust constraints between (for example) wood yield, ecosystem C storage and site N loss.

Perhaps the immediate challenge for process-based modellers is to convince foresters and other users that they (modellers) actually have something to contribute. In the short-term this might best be achieved through option (iii) above, which is attractive because the environmental criteria of sustainability could be incorporated as extra constraints into established decision-support tools. Option (i) could also be developed in order to improve conventional growth and yield predictions, to be replaced in the longer-term by option (ii). The modular nature of the CEF forest management system (Tomé et al. 2001) makes it a prime candidate in which to pursue these options.

## Acknowledgements

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# Programme

## International Workshop Models for the Sustainable Management of Temperate Plantation Forests

7–9 September 2000 • Bordeaux, France

### Thursday 7<sup>th</sup> September 2000

09:15–09:30 Introduction to the workshop. *Jean-Michel Carnus (INRA Pierroton, France).*

09:30–10:10 **Introductory keynote address:** Plant-Soil Interactions: key processes for understanding sustainable management. *Peter Millard (Macaulay Land Use Research Institute, Aberdeen, UK).*

Session 1 – Carbon and water balance (Chairperson: Jean-Michel Carnus)

10:10–10:50 **Keynote address:** Maximising wood yield, carbon storage and efficient use of N. *Melvin Cannell (Centre for Ecology and Hydrology, Edinburgh, UK).*

11:10–11:40 Simulated effects of tree harvest on C fluxes in 20<sup>th</sup> and 21<sup>st</sup> century temperate forests. *Daniel Rasse (University of Liège, Belgium).*

11:40–12:10 The 3-PG model: matters arising from evaluation against plantation data from many countries. *Joe Landsberg (Landsberg Consulting, Canberra, Australia).*

12:10–12:40 Impact of a 2xCO<sub>2</sub> climate scenario on the water and carbon balances and primary productivity of maritime pine. *Denis Loustau (INRA, Pierroton, France).*

Session 2 – Nutrient balance (Chairperson: Roderick Dewar)

14:30–15:10 **Keynote address:** Plant-soil models as tools to explain changes in forest productivity and site fertility over successive rotations. *Ross McMurtrie (University of New South Wales, Sydney, Australia).*

15:10–15:40 A model approach to assess the sustainability of the soil nitrogen balance of short rotation eucalyptus in Western Australia. *Marc Corbeels (CSIRO Forestry and Forest Products, Wembley, Australia).*

- 16:00–16:30 Nitrogen uptake in young Norway spruce on a fertile site in southern Sweden – effects of fertilization and herbicide treatments. *Urban Nilsson (Swedish University of Agricultural Sciences, Alnarp, Sweden).*
- 16:30–17:00 Linking production, leaf area, nitrogen use, and soil nitrogen availability in loblolly pine plantations. *Lee Allen (North Carolina State University, Raleigh, USA).*

### Friday 8<sup>th</sup> September 2000

#### Session 3 – Decision support, biomechanics (Chairperson: Denis Loustau)

- 09:30–10:10 **Keynote address:** The use of management-oriented growth and yield models to assess and model forest wood sustainability: a case study for eucalyptus plantations in Portugal. *Margarida Tomé (Instituto Superior de Agronomia, Lisboa, Portugal).*
- 10:10–10:40 Improving the methodology for stand-level assessment of long-term productivity shift: an example in sessile oak high forests. *Jean-François Dhôte (INRA-ENGREF, Nancy, France).*
- 11:00–11:30 A regeneration model for planted Norway spruce in southern Sweden. *Fredrik Nordborg (Swedish University of Agricultural Sciences, Alnarp, Sweden).*
- 11:30–12:00 Evaluation of different optimisation models to schedule silvicultural operations (in French). *Paulina Pinto (Departamento de Ingeniería Forestal, Pontificia Universidad Católica de Chile, Santiago, Chile).*
- 12:00–12:30 Simulation of tree growth including mechanics: a new perspective into the investigation of stand stability. *Thierry Fourcaud (Laboratoire de Rhéologie du Bois de Bordeaux, Pierroton, France).*
- 14:30 Visit to Pierroton experimental site.

### Saturday 9<sup>th</sup> September 2000

#### Session 4 – Synthesis (Chairperson: Michel Arbez)

- 09:30–10:00 **Keynote address:** Workshop synthesis, future prospects. *Roderick Dewar (INRA, Bordeaux, France).*
- 10:00–10:30 Open discussion (led by Margarida Tomé and Roderick Dewar): Workshop outcomes, future projects and collaborations envisaged.
- 11:00–12:30 Open discussion (continued).

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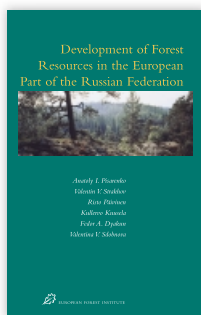
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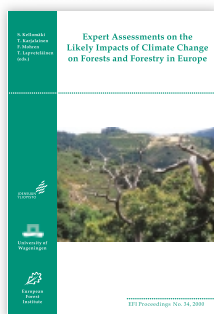
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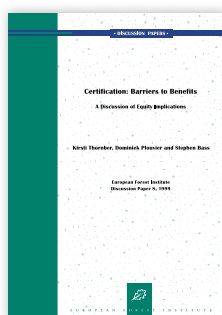
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