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(eds.)

Scientific Tools and Research Needs for Multifunctional Mediterranean Forest Ecosystem Management

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Contents

Foreword

In the Mediterranean, forest, woodlands, rangeland, "garrigues and maquis" (shrubby vegetation), are major components and integral parts of rural areas. For millenaries, these ecosystems have been shaped by man activities. Today their value is related to the production of a broad range of goods and services such as: wood and non-wood products, silvo-pastoral resources, amenities, soil and watershed protection, biodiversity, etc., with some marked geographical differences around the "Mare Nostrum" (Mediterranean Basin). Therefore and even more than in other regions, the management of these ecosystems must: i) clearly address the multifunctionality principles, ii) be integrated at landscape level in the management of rural areas.

Multifunctional management of Mediterranean forests is facing numerous ecological, economic and social challenges. In recent years, scientists have acquired a rather sophisticated knowledge on the components and functioning of forests but have often failed to integrate and synthesize information in a way that can provide adequate decision-making tools. Multifunctional forest management calls for the integration of stand and landscape levels; short and long term; and ecological, economic and social considerations. To successfully practice integration, we need from both management and scientific research, more work focusing on synthesizing existing knowledge and developing appropriate modeling, planning, valuation and finally integration tools in the form of advanced decision support systems.

The seminar "Scientific Tools and Research Needs for Multifunctional Mediterranean Forest Ecosystem Management" organized as an accompanying event of the MEDFOREX annual meeting, took place in Solsona, Spain on the 27th of November 2006. The seminar brought together more than 50 Mediterranean scientists from 15 countries working in different disciplines; forest ecology, forest management and planning, economics, operations research, and information technologies, which are crucial in managing and problem-solving in the Mediterranean forest ecosystems. Twelve papers focusing on cutting-edge theory and state-of-the-art applications were presented in the following topics:

- Mediterranean forest ecosystems: processes and functioning
- Mediterranean forest management decision systems
- Mediterranean forest economics and policy

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Marc Palahí, Yves Birot and Mercedes Rois
July 2007

Executive Summary

The Mediterranean basin constitutes a unique mosaic of terrestrial, freshwater and marine ecosystems, as a result of a distinct regional climate imprinted on a dynamic topography.

Yearly rainfall varies from 100 to more than 2500 mm while the average temperature ranges from 5 to 20 °C, with an intense summer drought period. Mediterranean flora is extremely rich with around 25.000 vascular plant species, widely distributed throughout the diverse ecosystems of the region. Climatic, geomorphic and anthropogenic factors have resulted in a mosaic-type landscape of a variety of forest types that cover an area of 73 mill ha, or about 8.5% of the region's area. However, purely Mediterranean forests and *maquis* cover about 56 mill ha, i. e. 7.5% of the total countries land.

Mediterranean forest ecosystems provide multiple wood and non-wood forest products and services which are crucial for the socio-economic development of rural areas as well as for the welfare of the urban areas of the Mediterranean region. In this context, Mediterranean forests require special attention due to several questions:

(i) They constitute a unique world natural heritage and play a key role in the welfare of urban and rural Mediterranean societies. The goods and services that they produce are very diverse (multi-functionality) and have a great market (many non-wood products) and non-market value (externalities).
(ii) They represent an exceptional richness in terms of biodiversity.
(iii) They are very vulnerable to numerous factors: forest fires, over-exploitation, degradation and desertification.
(iv) Their conservation and management affects the availability of soil and water resources, this last one being a key strategic resource for Mediterranean societies.
(v) Their future (as being in a transitional zone) is seriously endangered by climate change.

This situation requires the development of our knowledge for better conservation and sustainable management of Mediterranean forests.

The seminar "Scientific Tools and Research Needs for Multifunctional Mediterranean Forest Ecosystem Management" brought together Mediterranean scientists from several disciplines – forest ecology, forest management, applied economics, operations research, and information technologies, etc. – to discuss and present new techniques, tools and results that can provide solutions to the complex Mediterranean forest ecosystem management problems. The papers were organized in three main topics to tackle the five questions stated above: *(i) Mediterranean forest ecosystems: processes and functioning, (ii) Mediterranean forest management decision systems, (iii) Mediterranean forest economics and policy.* Such structure provided a unique opportunity to discuss in a multidisciplinary environment key challenges of Mediterranean forestry and forest research. In topic (i) the consequences of climate change on Mediterranean forests were discussed in depth by Matteucci et al. by presenting a long-term monitoring study of a montane-Mediterranean beech forest located at the boundary with the National Park of Abruzzo. In this paper the behaviour of the forest to current climatic conditions and their pronounced interannual variability was described, while the responses to global change were presented on the basis of modelling results. In topic (ii) J. G. Borges reviews the evolution of forest resources management planning in order to address ecological, economic and social sustainability concerns at the landscape level. In addition, several authors (Bravo et al., Calama et al., Sanchez et al.) presented empirical growth and yield models for different Mediterranean forest tree species to be use in multifunctional forest management planning. Finally, González et al. presented a new and innovative fire risk modelling approach to be use in forest management planning and Saura et al. presented a novel technique based on graph theory to address forest landscape structural goals. In topic (iii) Rojas Briales discussed a new policy framework for Mediterranean forests in the context of the recent social changes (from primary to tertiary societies). Furthermore, two papers presented new methods and approaches (Drogue and Venzi, Freeman and Insley) to account for the economic benefits of forest environmental services in forest management. Finally, Daly-Hassen et al. presented a cost-benefit analysis for cork oak in Tunisia.

Mediterranean Forest Ecosystem Functioning

The Response of Forests to Global Change: Measurements and Modelling Simulations in a Mountain Forest of the Mediterranean Region[1]

Giorgio Matteucci[1], Alberto Masci[2], Riccardo Valentini[2] and Giuseppe Scarascia-Mugnozza[3]

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Abstract

Forest ecosystems cover about 30% of land surface and are responsible for one half of global land productivity. On the other hand, in the last 60–80 years, terrestrial ecosystem have been subjected to significant climate changes that, if the current trend will not be slowed down or reversed, will bring them to a different, warmer climate in a relatively short time frame. Hence, it is of paramount importance to study the response of forests to climate and global change.

In this framework, the continuous long-term monitoring of forest responses to environmental conditions can provide useful information on their adaptation to the present climate and its interannual variability, as well as on their functionality. Using simulation models, this information can be combined, in order to have a reasonable idea on how the forest will respond and adapt to those scenarios.

In this paper, we present the results gathered during the long-term monitoring of a montane-Mediterranean beech forest located at the boundary with the National Park of Abruzzo, investigated within the framework of national and European research projects. The behaviour of the forest to current climatic conditions and their pronounced interannual variability will be described, while the responses to global change will be presented on the basis of modelling results.

Keywords: Carbon absorption, montane-Mediterranean forests, eddy correlation, forest ecosystem modelling, long-term ecological research

1 This paper is dedicated to the memory of Roberto Bimbi, 1948–2001

Introduction

In the last 60–80 years, terrestrial ecosystem have been subjected to significant climate changes that, if the current trend will not be slowed down or reversed, will bring them to a different, warmer climate in a relatively short time frame (40–60 years, IPCC 2001). Furthermore, recent findings, obtained through ice-core analysis, have confirmed that, although there have been other periods of atmospheric composition changes in the past, the concentration of greenhouse gases in the present atmosphere is unprecedented for the last 400,000 to 600,000 years (Siegenthaler et al. 2005). The same study has also revealed that a significant part of the climate warming occurring in the glacial/interglacial phases can be attributed to the increase in atmospheric CO_2.

Among the various terrestrial ecosystems, forests cover slightly more than 30% of global land surface (4.1 Gha), contain more than 80% of aboveground terrestrial carbon and nearly 40% of that present in soils. Forest biomes are responsible for one half of global land productivity and it has been calculated that 70% of carbon exchanges between terrestrial biomes and atmosphere is going through them (Waring and Schlesinger 1985). Hence, it is of paramount importance to study the responses of forests to climate and global change. Studies can be performed by exposing forest trees to the future predicted atmospheric conditions (Drake et al. 1997; Scarascia-Mugnozza et al. 2006), but, in this framework, the continuous long-term monitoring of forest responses to environmental conditions can provide useful information on their adaptation to the present climate and its inter-annual variability. Using simulation models, this information can then be combined with the results obtained in the studies where tree species have been subjected to future climatic scenarios, in order to have a reasonable idea on how the forest will respond and adapt to those scenarios (Norby et al. 1999; Woodward and Lomas 2004).

As to ecosystem monitoring, protected areas can play a fundamental role, as there we can actually monitor the ecological consequences of human perturbations on climate in areas that, generally, have been marginally affected by humans (Boyce 1992).

In this paper, we present the results gathered during the long-term monitoring (>10 years, presently ongoing) of a mountain beech forest located at the boundary with the National Park of Abruzzo, that has been investigated within the framework of national and european research projects. Results will be presented on the behaviour of the forest to current climatic conditions. During the years, ecophysiological parameters have been measured and then used to parameterise process-based models which have been utilised to simulate the response of this forest to changed climatic conditions

Site description and methods

In the beech (*Fagus sylvatica* L.) forests of Collelongo (Abruzzo region, Central Italy), a permanent experimental plot (Matteucci et al. 1998) was established in the 1993 (Selva Piana stand, 41º52' N, 13º38'E, 1500 m a.s.l.). It is now run jointly by the University of Tuscia and IBAF-CNR. In the plot, ecological and silvicultural studies are being carried out. The site is also part of the Italian group of the European Monitoring Network on Forest Ecosystems (CON.ECO.FOR., National Forest Service, Italian Ministry of Agricultural Policies).

The Selva Piana stand belongs to a 3,000 ha community forest that is part of a wider forest area, included in the buffer zone of the National Park of Abruzzo. The environmental and structural conditions of the stand are representative of Central Apennine beech forests. In the area of the experimental site, the density is 890 trees ha^{-1}, the basal area is 32.1 m^2 ha^{-1} with a mean diameter at breast height of 18.1 cm (starting sampling from 1 cm diameter trees). Leaf Area Index, measured with both the Plant Canopy Analyser LiCor 2000 and the litterfall method, ranged, over several years, between 4.5 and 6 m^2 m^{-2}. In 2006, mean tree age has been estimated to be around 110 years. The soil, developed on a calcareous bedrock, has a variable depth (40–100 cm) and can be classified as a humic alisol. Site topography is gently sloped with some variation. The meteorological conditions registered at the site over some of the experimental years showed a mean annual temperature of 7 °C (1996–2003) and a mean annual precipitation of 1206 mm (1996–2003). On a longer term (1920–1955), an extrapolation of data measured all over the Abruzzo region gave for the elevation corresponding to the experimental site a mean annual temperature of 6.9 °C and a precipitation of 1127 mm (Cavasicci et al. 1981).

The site is equipped with a 26 m high scaffold tower with walkable floors. At the top of the tower, an additional mast allows to reach the height of 32 m (8–10 m above the canopy). Meteorological sensors are mounted on the tower, along the canopy profile. Air temperature is measured at six levels (0.5, 11, 18, 21, 24 m above the ground), air humidity and wind speed at three levels (respectively 0.5, 2, 24 m and 21, 23, 27.5 m). Incoming total and diffuse photosynthetic active radiation (PAR, Li190, LiCor, USA and SKYE, UK) and net radiation (REBS, USA) are measured above the top of the canopy. Precipitation is measured above (27.4 m) and below the canopy (1.5 m) by rain gauges. Soil temperature is measured at two depths (0.05 and 0.20 m). In the soil, also the heat flux is monitored by flux plates (REBS, USA). During winter, an ultrasound sensor is used to monitor snow depth. All the meteorological data are collected by a Campbell CR10 data-logger (Campbell, USA). All the equipments are powered by six 70W solar panels installed in a nearby clearing (200 m from the tower). The solar panels charge four 12V/140A batteries placed in a box underneath the panels. From the box, a 200 m long 24 V cable brings the power to the energy control system placed in a hut near the tower. The system distributes electricity at all the equipments at 12V. In the hut, four additional 12V/140A batteries are connected to the control system. These batteries are used as emergency buffer. When their charging state is too low, a gasoline generator is automatically switched on to recharge them.

The site is part of the EUROFLUX network, that studies the long-term exchanges of carbon and water vapour of European forest ecosystems (Aubinet et al. 2000). At the site, fluxes of carbon at the ecosystem level have been measured since 1993 with the eddy covariance technique (Valentini et al. 1996). Details on the eddy covariance technique can be found elsewhere (Baldocchi et al. 1988). Although it has some limitations (an homogenous canopy and atmospheric unstable conditions are needed), the technique is able to integrate the carbon and water gas exchanges over an extended area (0.1–1 km^2) and provides carbon sequestration estimates without being destructive; it can be operated on a continuous basis (24 h a day) for one or more years. The experimental set-up follows the EUROFLUX methodology, thoroughly described by Aubinet et al. (2000). The instrumentation consists of a fast response infrared gas analyser (LI-6262, LI-COR, USA) and a three dimensional sonic anemometer (SOLENT 1012R2, Gill Instruments, UK). Air is drawn through the analyser by a pump (VDO M48x25/l, Antriebstechnik GmbH, Germany) installed downstream of the analyser. Being inside a homogenous area covered by beech forests, the site provides a large fetch. Fetch dimensions are, approximately, 2.5 km × 2 km (NNE-SSW × WNW-ESE).

During the years, the structure and ecophysiology of the forest stand have been studied in details (photosynthesis, soil respiration, growth, nutrient conditions, soil water content, etc.). A description of the methods and data is beyond the scope of this paper. Details and data have been published elsewhere (Matteucci et al.1998).

The results derived through the ecophysiological and structural studies have been elaborated to parameterise process-based models that were utilised to simulate the response of this forest to changed climatic conditions. For this purpose, we used two process-based models, CANOAK (Baldocchi 1997) and GROMIT (Medlyn et al. 1999). Both models have been tested and validated for the beech forest under study.

The climate scenarios used for the simulations were obtained through the IPCC Data Distribution Centre (http://ipcc-ddc.cru.uea.ac.uk/cru_data/cru_index.html). We decided to examine the effects of climate change at a time when [CO_2] is expected, by the year 2070–2100, to be 700 µmol mol^{-1}, that is 2× the present-day atmospheric concentration, using the outputs of the Hadley Centre Global Circulation Model HadCM2. HadCM2 has a spatial resolution of 2.5° × 3.75° (latitude by longitude). Long-term climate data were obtained by the Potsdam Institute for Climate Impact Research (PIK), that refined the HadCM2 grid to 0.5° × 0.5°. PIK generated climate data for the period 1831–2100 for a number of sites, applying an exponential increase of [CO_2] from the year 1990 to the year 2100.

Results and discussion

Forest and climate

Due to the large geographical and topographical variability of the Italian landscape, the typical Mediterranean climatic bi-seasonality with dry and hot summers and moist and cool autumns and winters can be locally modified. This is particularly true for sites located in the mountain ranges. In the period of full operation of the experimental site in the Collelongo beech forest, mean annual temperature showed a certain variability (6–8 °C), while total annual precipitation ranged from 982 mm (1997) to 1340 mm (2000), with a ±15% oscillation around the short-term mean of 1206 mm. Interannual differences were more important for summer temperature and precipitation (June, July and August). From 1996 to 2003, there was an increasing trend in mean summer temperature, going from 13.4 °C (1996), to 18.4 °C (2003) with the maximum in the exceptionally hot summer of 2003, while summer precipitation was limited and showed differences up to 300% (range 56–234 mm, average 129±22 mm), due the less (1997) or more frequent (2002) occurrence of convective afternoon rain showers. Interestingly, the increasing trend of summer temperature within 1996–2003 was apparently not linked to precipitation, as in the warmest year the summer precipitation (180 mm) was greater than the period mean.

We all know that the effects of forests on climate is relevant, particularly within canopies and in their immediate surroundings, but also at regional scale (Waring and Schlesinger 1985; Piussi 1994). The instrumentation installed at the site makes possible to monitor with precision which are the effects of the beech canopy on micrometeorological parameters. In this respect, several features can be presented and discussed, but it is of interest to show which are the trends in air and soil temperatures during snowmelt and canopy development, two moments that are particularly relevant for deciduous forests growing in mountainous sites. In Figures 1 and 2 the monthly trends of above-canopy air temperature and soil temperatures at 5 and 30 cm depths are shown for the months of April and May of the year 1998.

In April (Figure 1), the daily temperature amplitude was significant, ranging between 8 and 15 °C. Absolute minimum temperature reached –10 °C, while absolute maxima were already at spring level, arriving almost at 15 °C. Mean monthly temperature was 2.5 °C. Due to the presence of the snow cover, soil temperature at both depths was almost constant in the first half of the month and no signs of a daily trend were present. It is worth noting that around day 105, when the snow melted, soil became more reactive to thermal fluctuations, starting to follow the daily trend in air temperature, particularly at –5 cm depth, while the deeper one seemed to follow daily mean temperature, being more insulated from sudden temperature changes, such as those present at the end of the month.

Temperature trends for the month when growing season starts (May) are presented in Figure 2.

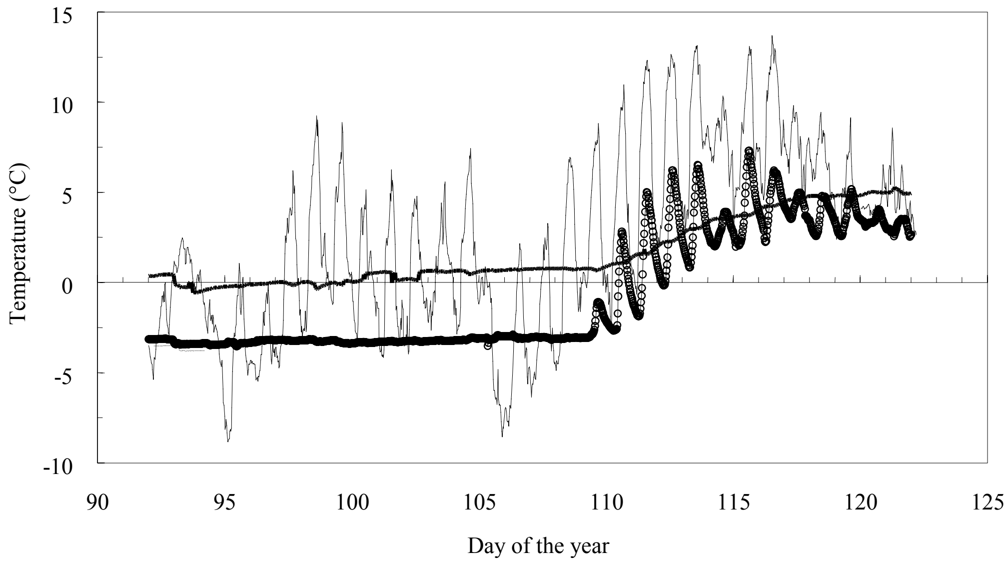

Figure 1. Monthly trends for temperatures measured in April 1998 in the Collelongo beech forest. Data are measured every 10 seconds and averaged every half-hour. Air temperature above the canopy (thin line), soil temperature at –5 cm (open circles) and at –30 cm (thick line).

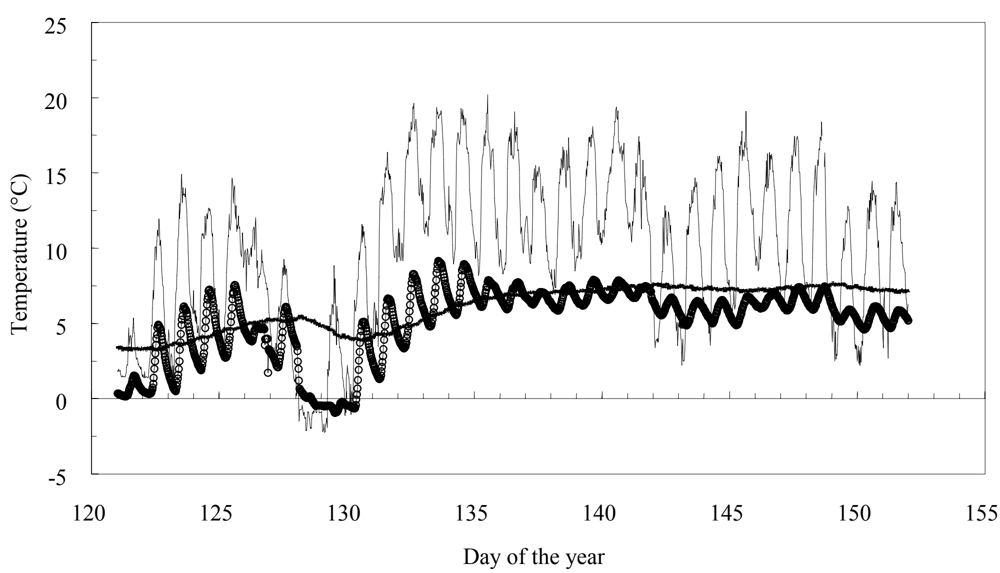

Figure 2. Monthly courses for temperatures measured in May 1998 in the Collelongo beech forest. Data are measured every 10 seconds and averaged every half-hour. Air temperature above the canopy (thin line), soil temperature at –5 cm (open circles) and at –30 cm (thick line).

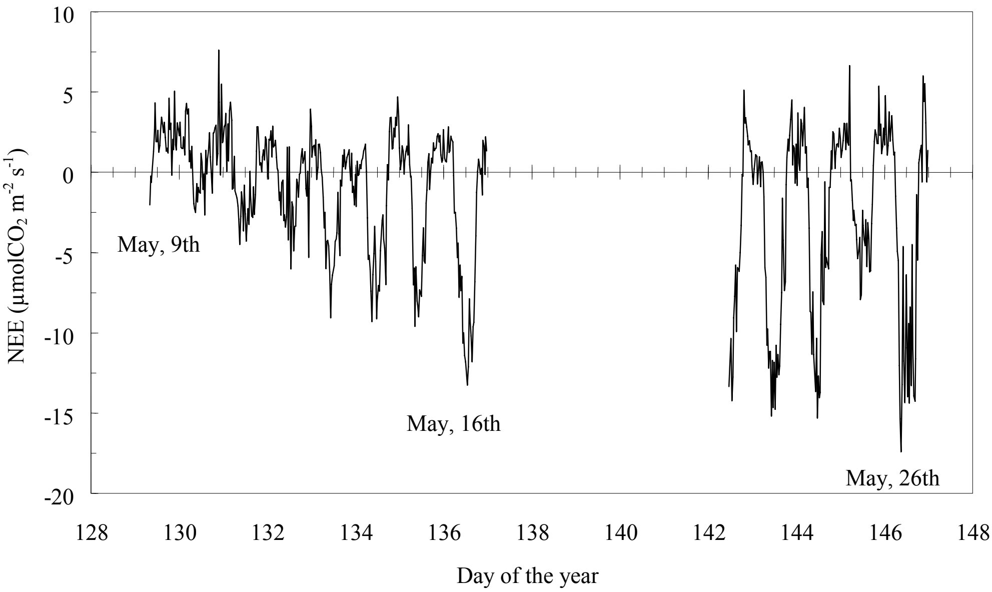

Figure 3. Daily courses of net ecosystem exchange (NEE) measured by eddy correlation above the beech forest of Collelongo in May 1998. Data are measured continuously and averaged every half hour. The sign follows the micrometeorological convention, being negative when CO_2 is going from the atmosphere to the canopy and positive vice versa.

Daily temperature amplitude range was similar to the former month, but minimum and maximum temperature were higher. In May, mean monthly temperature was 9.8 °C compared to 2.5 °C of April: at this site, during 4 years, the difference between May and April proved to be always the largest month-by-month change. Nevertheless, it is still possible to find freezing temperature in this month. Soil temperature at –5 cm was still following the daily air temperature trend, while at –30 cm, temperature was increasing according to the temperature sum. It is interesting to observe what happened around the mid of the month (day 130–145). In this period, due to the rapid leaf unfolding, the Leaf Area Index (LAI) passed from 1.68 m^2 m^{-2} in day 128 to 4.6 m^2 m^{-2} in day 143 (Cutini et al. 1998) and daily temperature amplitudes at –5 cm became day by day less significant when compared to those of the second half of April and the first half of May. This was occurring albeit the daily air temperature amplitudes remained similar as previous days.

Canopy fluxes

The type of equipments installed at the site allows to measure processes that go far beyond meteorology and micrometeorology. In fact, at the Collelongo beech forest, forest-atmosphere mass and energy exchanges are measured with the eddy correlation technique (see Methods section). Fluxes of carbon, water vapour and sensible heat are measured at the site since 1993 (Matteucci et al. 1998) and continuously since 1996, within the European project EUROFLUX (Valentini et al. 2000). Due to the relevance of the role of forest ecosystems as carbon sink/source, linked to the implementation of the Kyoto protocol (IGBP 1998), here we will concentrate on carbon fluxes (Net Ecosystem Exchange, NEE),

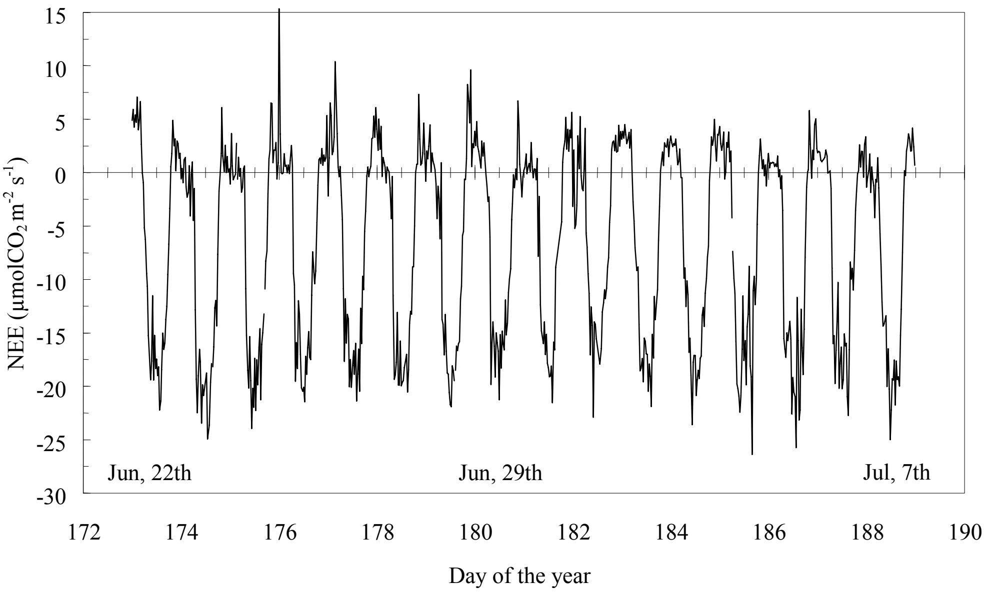

Figure 4. Daily courses of net ecosystem exchange (NEE) measured by eddy correlation above the beech forest of Collelongo between June and July 1998. Data are measured continuously and averaged every half hour. The sign follows the micrometeorological convention, being negative when CO_2 is going from the atmosphere to the canopy and positive vice versa.

presenting measurements performed in three contrasting periods of the year 1998: at the onset and closing of the growing season and when the canopy is in its full competence. In fact, these periods can reasonably exemplify forest responses to climate during the annual course.

In figure 3, the period between 9 and 26 May is presented. In 1998, the deciduous beech forest opened its leaves between the first and the second week of May and it is interesting to see how canopy fluxes measurements were able to catch this phase of the growing season.

Measurement of instantaneous NEE started to present the typical daily trend from May 11, with maximum values of –4 µmolCO_2 m^{-2} s^{-1}. From that day, maximum midday NEE kept on increasing, reaching –13 µmolCO_2 m^{-2} s^{-1} on May 16 and –15 µmolCO_2 m^{-2} s^{-1} towards the end of the month. The daily trend showed the classical differences, with CO_2 emission towards the atmosphere at night (positive values) and CO_2 absorption by the canopy during the day (negative values). The first day in which the forest was a carbon sink on a daily basis was on May 11, when it gained 0.2 gC m^{-2} day^{-1}. After this day, the canopy showed a fast response to climate and carbon absorption was already 3.7 gC m^{-2} day^{-1} on May 16. Indeed, on May 26 LAI was already at its maximum of 5.4 m^2 m^{-2} and the forest absorbed 4.2 gC m^{-2} day^{-1}.

The reaching of the maximum LAI was not contemporary with the maximum photosynthetic competence of the canopy that was reached between mid June and early July. Net Ecosystem Exchange for this period is shown in Figure 4. During the period between 22/06/98 and 7/07/98, the weather was almost always clear, with bright and hot days.

During all days, instantaneous NEE reached maximum values ranging from –20 and –25 µmolCO_2 m^{-2} s^{-1} around midday. Interestingly, every day, canopy fluxes showed a relatively large time window in which they were around the maximum (10:00–16:00). On average, during this period, the forest absorbed carbon between 6:00 and 19:00-19:30. Although the

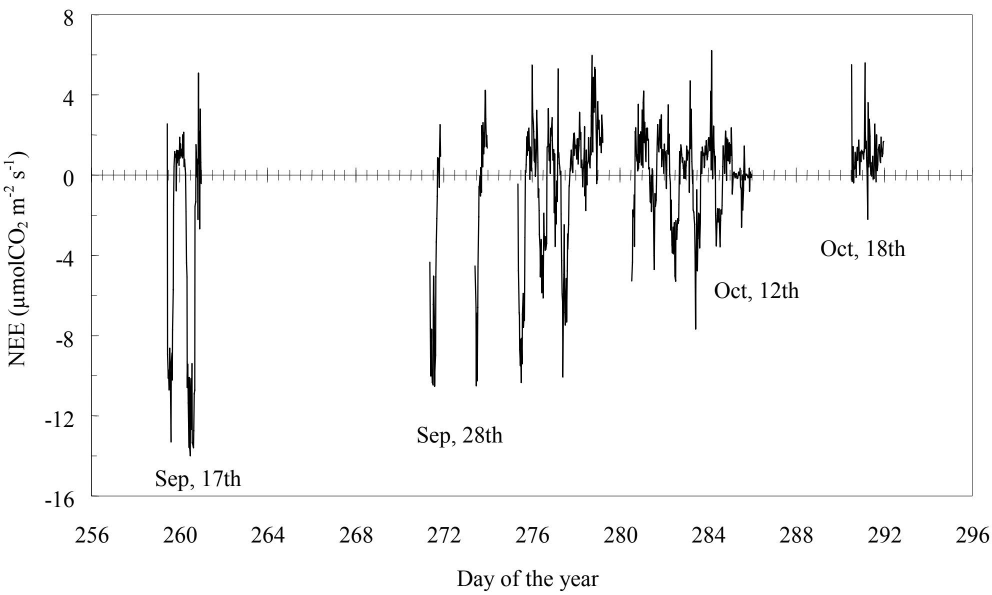

Figure 5. Daily courses of net ecosystem exchange (NEE) measured by eddy correlation above the beech forest of Collelongo between September and October 1998. Data are measured continuously and averaged every half hour. The sign follows the micrometeorological convention, being negative when CO_2 is going from the atmosphere to the canopy and positive vice versa.

LAI in this period was comparable to that in late May – early June, the maximum NEE was by 20 to 30% higher due to fact the leaves reached their maximum photosynthetic competence and the temperature were more favourable. Compared to the usual climatic conditions, night temperatures were particularly elevated in July and the ecosystem was characterised by relevant night respiration instantaneous fluxes, ranging between 3 and 5 $\mu molCO_2\ m^{-2}\ s^{-1}$. In the period shown in Figure 4, the average daily carbon gain was 6.8 gC $m^{-2}\ day^{-1}$, with maximum of 7.9 and minimum of 5 gC $m^{-2}\ day^{-1}$.

Another important period in the growing season of a deciduous forest, is the phase of leaf senescence. Eddy correlation data measured in this period during 1998 are shown in Figure 5. Around mid September, the canopy was still photosynthetically active, with NEE daily peaks of –14 $\mu molCO_2\ m^{-2}\ s^{-1}$.

Toward the end of the month, daily maximum NEE decreased to –10 $\mu molCO_2\ m^{-2}\ s^{-1}$ and reached –5 $\mu molCO_2\ m^{-2}\ s^{-1}$ during the first 10 days of October. In this period, the daily trend were very variable, loosing the typical shape of full-season days (see Figure 4). In 1998, the last day in which the forest was a slight carbon sink was October 10 (0.3 gC $m^{-2}\ day^{-1}$). It is worth noting that the canopy fluxes at the closing of the growing season presented a larger day-by-day variability that those at the onset of the season (Figure 3). This is partly due to the more variable climatic conditions in September and October, but also to the fact that leaf shedding is more variable that leaf development, with differences existing also at the various canopy depths.

As seen in Figure 4, the period bridging the end of June and the first half of July, is the phase with optimal and maximum carbon absorption by the beech forest. These results confirm what already found and reported for the same forest in 1993–94 (13), 1996 and 1997. Interestingly, the period of maximum NEE is also the phase in which leaves have completely developed, both as dimension and nutrient content.

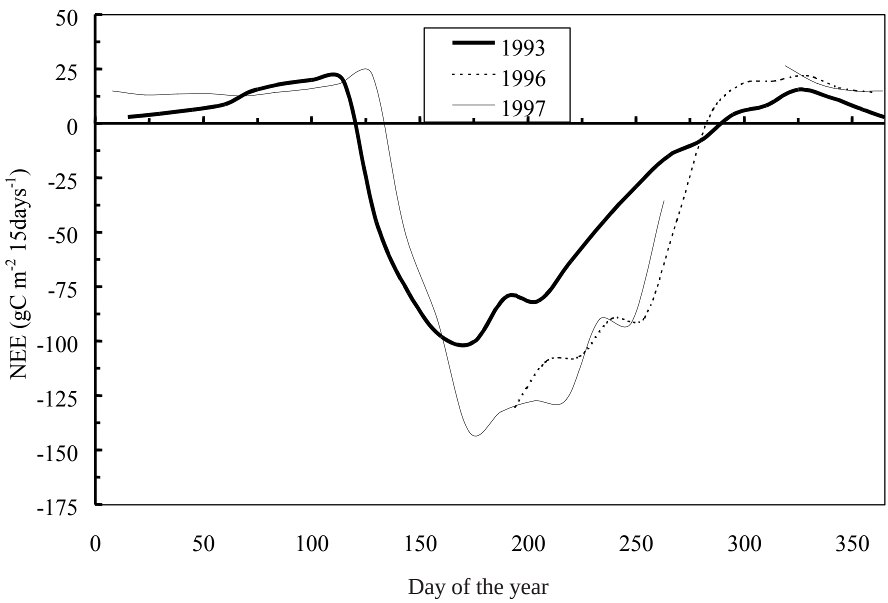

Figure 6. Seasonal course of net ecosystem exchange (NEE) measured by eddy correlation above the beech forest of Collelongo in 1993, 1996 and 1997. Data are measured continuously and summed in 15 days periods. The sign follows the micrometeorological convention, being negative when carbon is going from the atmosphere to the canopy and positive vice versa.

From data such as that presented in Figures 3–5, when measured for the whole year, it is possible to calculate annual NEE trend and carbon budgets (Matteucci et al. 1998; Aubinet et al. 2000; Valentini et al. 2000). When the aim is to scale up in time (more years) and space (region, globe), interannual flux variability comes into play. In fact, it is not sufficient to determine if a forest is a carbon sink or source on a particular year, but it is necessary to verify such features in the long term and which factors are causing sink/source behaviour.

Data for three years for the Collelongo forest are presented in Figure 6.

Though the leaf unfolding was anticipated in 1993, the carbon absorbed during the growing season (May-September) was significantly higher in 1996 and 1997 compared to 1993. For the three years, the period with maximum carbon sink was the same, while significant differences were present for the onset and closing of the carbon gain period. This period started earlier and ended later in 1993. The differences in summer carbon absorption between 1993 and 1996–97 can be partly explained by a 20% lower LAI in the former year and partly by the presence in 1993 of an important water stress (nearly three months without rain) (Valentini et al. 1996). It is worth noting that, in all the three years, the forest was a significant carbon sink, the absorbed carbon ranging from 4.7 tC ha^{-1} a^{-1} (1993, Valentini et al. 1996) to 6.6 tC ha^{-1} a^{-1} (1996-97, Valentini et al. 2000).

Forest and climate change: modelling results

Results obtained from the structural and ecophysiological studies performed at the site (Matteucci et al. 1998) have been used to parameterise process-based models. Climate data collected at the site were fed to models and model runs have been performed. The results coming from eddy correlation and growth measurements have been used as validation data to check the performance of models for the Collelongo beech forest (Medlyn et al. 1999). Models have then been run with a modified climate, changed according to the climate scenarios predicted by Global Circulation Models (see Methods for details). Climate change was considered to occur after 1990.

Before entering in the description of some modelling results, a description of the climate scenarios for Italy, in terms of temperature and precipitation, is reported.

According to predictions for 10 stations, distributed latitudinally and altitudinally all over Italy, with a larger coverage in Central-South Italy, in the period 2040–2060 mean temperature should increase by 4.3 °C and precipitation by 10%, compared to 1850–1870 (which are not far from current conditions). The predicted temperature increase is expected to have more significant impacts on mountainous areas, where its change relative to current values will be much more relevant (73% compared to 35% of lower elevations). Although predictions for precipitation are much more uncertain than those for temperature, the predicted slight increase of rainfall (10%) might not compensate for the augmented evaporative demand driven by increased temperature, possibly causing harmful effects on the vegetation of the driest areas (Mouillot et al. 2002). It must be stressed that the predicted temperature could have a strong impact on the forest tree species, which altitudinal distribution is also linked to temperature conditions (both average and extreme temperatures). In this respect, we have calculated that an average 4.3 °C change in temperature, should shift significantly upwards (or northwards) the actual vegetation zones (Bakkenes et al. 2002). The impact should be more relevant in coastal and high elevation zones, where vegetation is already subjected to environmental extremes. Similar impacts and shifts have been reported to possibly occur, following climate changes, in East Asia (Omasa 1998) and Japan (Tanaka et al. 1998).

The model GROMIT was used to simulate gross primary production (GPP), bud break date and growing season length at the Collelongo beech forest. Mean bud break date varied around day 137 for the period 1830–1990, with a range of variability of 12–15 days. In this respect, the predictions were in accordance with the real bud-break for the nineties (Matteucci et al. 1998). From 1990 onwards, due to the increasing trend of temperature, bud break date, with a similar variability range, kept on anticipating, to arrive at an average bud break date at day 112–115 for the period 2080–2100. Growing season length showed a similar pattern, increasing after year 1990. However, as the GROMIT model is using a balance of photosynthesis/respiration to start leaf senescence, the increasing trend was not so evident as that of bud break, possibly because higher temperature in late summer caused the photosynthesis/respiration ratio to decrease. In response to a longer growing season and particularly to [CO_2] progressive augmentation from year 1990, GPP of the Collelongo beech forest showed an increasing trend. Relatively to the average GPP for the 1990–1999 decade, in 2010–2019, GPP increased by 18%, in 2050–2059 by 46% and in 2090–2099 by 62%. The standard deviation of GPP within each decade was between 1 and 2 tC ha^{-1} a^{-1}. According to the above described results, climate and atmospheric changes would exert a relevant role for the Collelongo beech forest, presented here as a “test” site.

At a finer scale, fluxes of carbon and water vapour were simulated for a full growing season (May, 1st – Oct, 15th) with the CANOAK model, comparing results of 1996–1998 climate and [CO_2] with conditions in 2080 (Table 1).

Under current conditions, the beech forest of Collelongo shows a GPP of 11.17 tC ha^{-1} yr^{-1}, with a carbon gain (NEE), during the growing season of 5.73 tC ha^{-1} yr^{-1}. The proportion of net fluxes and respiration is almost 1:1 and evapotranspiration (ETR) amounts to 346 mm. Similarly to what reported for other species (31), when only temperature was changed there was a decrease of GPP (9–19%) and an increase in respiration (6–11%). Globally, the two changes caused a sensible lowering of NEE (22–48%). The concurrent slight increase in ETR resulted in a decreased water use efficiency (GPP/ETR) by 12% and 21%, respectively with

Table 1. Simulated gross productivity (GPP), evapotranspiration (ETR), net ecosystem exchange (NEE) and total respiration (R) of the beech of Collelongo, under four scenarios of climate and atmospheric change. Simulations are from the model CANOAK and have been calculated for one growing season. The sign of NEE follows the micrometeorological convention, being negative when the ecosystem absorbs carbon. GPP, NEE and R are in tC ha^{-1} $seas^{-1}$; ETR in mm $seas^{-1}$.

Variable	Current conditions	$[CO_2]$=700	Eff. %	T+2°C	Eff. %	T+4°C	Eff. %	$[CO_2]$=700 T+2 °C	Eff. %
GPP	11.17	15.9	42	10.21	-9	9.05	-19	14.66	31
ETR	346	267	-23	360	4	357	3	284	-18
NEE	-5.73	-10.43	82	-4.48	-22	-3.00	-48	-8.92	56
R	5.43	5.45	0.3	5.74	6	6.05	11	5.74	6

2 °C or 4 °C temperature increase. When simulations were performed in an elevated CO_2 atmosphere, GPP and NEE increased significantly. When the scenario included both temperature and CO_2 increase, the stimulation of productivity by CO_2 was slightly lowered both by the increase in respiration (6%) and by a direct effect on GPP, probably due to higher than optimal temperature for photosynthesis. Anyhow, the system proved to still absorb more carbon when compared to the current climatic conditions. Elevated CO_2 had also a direct effect on canopy evapotranspiration, reducing it by about 20%. This resulted in an increased water use efficiency by 85% when only a change of $[CO_2]$ was considered, and by 60% when also temperature was increased by two degrees. This is an important result if we take into account that the site, although in the mountain, is located in the Mediterranean region, where precipitation can be a limiting factor in summer. For the simulations under the increased temperature scenarios (with and without the changing of atmospheric CO_2 concentrations), it must be underlined that the model has not a phenological cycle included, hence the growing season was kept constant during the simulations. In this respect, it was reported, for beech in France and oak in Southern England, that the length of the growing season may have a very important role for forest carbon balance (Randle et al. 1999).

Conclusions

We have seen that the results gathered in the long term monitoring of a forest ecosystem located in the buffer zone of a protected areas can be useful to understand forest behaviour and response to current climate.

The use of models, parameterised after studies and data from the site, can provide, when coupled to the available climate scenarios of global change, information on the possible response of the studied deciduous beech forest to global changes.

In this respect, it would valuable to put together the information on climate, forest functioning and ecosystem dynamics in protected areas, in a way to form a database to evaluate climate effect in current and predicted changed conditions.

Combined research approach will surely provide the most valuable information that are needed to predict how terrestrial ecosystems will react to the already occurring global change (Cernusca et al. 1998) and protected areas should be a preferred location for such type of research, as those are the places where it is possible to monitor the ecological consequence of human perturbations on landscape and climate (Hannah et al. 2002). The amount of money that could be necessary to implement a combined research approach in protected areas would be a small fraction of the money currently spent on environmentally harmful subsidies (James et al. 1999).

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Mediterranean Forest Management Decision Systems

New Tools for Designing Landscapes: Models and Decision Systems

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Abstract

Forest resources management planning must address ecological, economic and social sustainability concerns at the landscape level. New tools are thus needed that may help design sustainable landscapes. A brief history of forestry planning models and decision systems is presented. Advanced tools for designing landscapes are discussed. Emphasis is on the comparison between heuristic and exact approaches to modelling spatial concerns. Both the unit restriction model and the area restriction model are addressed. Models must be used within decision systems for practical applications. Architectural frameworks for decision systems development may be key for effective application. Results of recent research are presented. Some applications are discussed to illustrate further research needs.

Keywords: forest management; landscape-level planning; heuristics; mathematical programming

Introduction

Forests have emerged in public perception as a source of multiple goods and services. Socioeconomic development and demographic trends have led to an increasing awareness of the importance of biodiversity, aesthetic values, recreation uses and also of the externalities that derive from the use of forests for commercial purposes. In the developed world, forest land is viewed often as a consumer good and not as a source of traditional products such as timber: people derive satisfaction and a sense of security from its mere existence. These trends define a different framework for forest management and policy. The study of the inter-relations and trade-offs between economic activities assumes more relevance, and spatial considerations are often critical.

In the past, research in forest management modelling has emphasized the improvement of strategic management models. The time dimension, a determinant characteristic of forestry production systems, has been allowed to play a major role in the development of harvest schedules. However, the implementation of forest plans also requires spatially feasible prescriptions. Financial efficiency usually dictates the concentration of activities of harvesting and infrastructure (e.g. road network) development. Conversely, socioeconomic and environmental goals often suggest its dispersion in time and space. The spatial arrangement of harvests has become a critical management concern. Accordingly, spatial resolution has become an important issue in forest management modeling (Borges et al. 1999).

Heuristic techniques to address harvest scheduling based on the decomposition of a linear programming (LP) master problem such as the ones developed by Hoganson and Rose (1984) and Gunn and Rai (1987) have the ability to recognize considerable detail within a strategic planning framework. Nevertheless, in addition to the ability to recognize individual stands in the output of a harvest scheduling model, managers must take into account the interactions of management decisions between neighboring stands. The spatial conditions generated by a model (e.g. amount and type of forest edge and interior space) become an additional forest output (Borges et al. 1999)). This output may be further be critical to address for example landscape resistance to forest fires (Gonzalez et al. 2005).

The incorporation of spatial relationships into forest management scheduling models and spatial optimization has thus been as increasingly important research topic. Thompson et al. (1973) pioneered the formulation of adjacency constraints in a forest management model. Since then, the integration of spatial considerations into forest management scheduling models has received considerable attention. Spatial optimisation is seen as key to addressing current multi-objective forest management concerns. Two major approaches have been used. The first includes exact methods such as mathematical programming that produce an optimal solution (Constantino et al. in press). The second includes heuristics, which seek to approximate an optimal solution at a reasonable computational cost, without being able to guarantee optimality or even feasibility (Borges et al. 2002). This paper will provide a brief review of both approaches. Models must be used within decision systems for practical applications. This paper will further refer to the technology needed to support model use.

Models

There are two basic mathematical models to recognize the spatial context of forest management planning decisions (Murray 1999). Both models assume that the forest area has been classified into stands. The Unit Restriction Model (URM) assumes that all stands are almost as large as the maximum harvest area so that any two adjacent units cannot be harvested in the same time period. The Area Restriction Model (ARM) assumes that stands' area is variable and may be much smaller than the maximum harvest area, so that cutting two adjacent units in the same time period is feasible even if there are adjacency constraints.

The integer (IP) or mixed integer (MILP) linear programming framework has been considered to solve both the URM and the ARM models (e.g. Kirby et al. 1980; Covington et al. 1988; Torres Rojo and Brodie 1990; Hof and Joyce 1993; Snyder and Revelle 1997; Martins et al. 2005; McDill and Braze 2000; McDill et al. 2002; Crowe et al. 2003; Goycoolea et al. 2005). In order to circumvent computational constraints to the use of exact methods, other authors addressed spatial modeling with heuristics such as Monte Carlo Integer Programming (e.g. O'Hara et al. 1989; Nelson et al. 1991), simulated annealing (e.g. Lockwood and Moore 1993; Dahlin and Sallnas 1993), tabu search (e.g. Bettinger et al.

1998; Caro et al. 2003), with dynamic programming based approaches (e.g. Hoganson and Borges 1998; Borges et al. 1999) or with hybrid heuristic approaches (e.g. Pukkala and Kangas 1993; Boston and Bettinger 2002; Falcão and Borges 2002).

Both exact and heuristic approaches have thus been used to solve the URM. Yet generally spatial optimization has to address objectives other than complying with a maximum harvest area constraint. Contrarily to the URM, the ARM may help address a wide range of spatial objectives. Stand design produces the landscape spatial elements upon which decisions are made and thus impacts the spatial conditions targeted by optimization (Barrett 1997; Borges and Hoganson 1999; Heinonen 2007). The ARM may thus further be interesting to increase management flexibility as it aggregates dynamically stands into harvest blocks. Several authors reported heuristic solutions to the ARM (e.g. Lockwood and Moore 1993; Boston and Bettinger 2002; Falcão and Borges 2002; Heinonen 2007). Nevertheless, computational constraints have been until recently an obstacle to effective use of exact methods to address the ARM.

Two main integer programming formulations have been proposed to overcome the computational obstacle (Constantino in press). The first approach encompassed an exponential number of variables (e.g. Martins et al. 1999 and 2005; Goycoolea et al. 2005). The second approach encompassed an exponential number of constraints (e.g. Martins et al. 1999; McDill et al. 2002; Crowe et al. 2003). Constantino (in press) has recently presented a new integer programming formulation for the ARM that has a polynomial number of variables and constraints. The differences between the potential for applicability of heuristic and exact approaches to solve large ARM instances are becoming less important.

Progress is being made on the development of both heuristic and exact approaches to address spatial optimization in large-scale forest management scheduling problems. In the case of meta-heuristics such as simulated annealing, genetic algorithms and tabu search a major concern to be addressed is the adjustment of parameters and penalties that are problem-specific. Pukkala and Heinonen (2006) proposed recently an approach to optimize heuristic search. In the case of exact methods, the research focus is on other formulations to target a wide range of spatial conditions of interest. In both cases, decomposition of the forest management master problem appears to be a promising option (e.g. Borges et al. 1999; Martins et al. 2005; Wei and Hoganson 2006; Heinonen 2007).

Decision systems

In order for models to be used effectively, they must be programmed, integrated and used within computer-based decision systems. Stimulated by developments in business administration and industry, these systems have been improving the quality and transparency of decision-making in natural resource management (Reynolds et al. 2005). They provide support to solve decision problems by integrating database management systems with simulation, analytical and operational research models, graphic display, tabular reporting capabilities, and the expert knowledge of scientists, managers, and decision makers (Reynolds et al. 2005). Decision systems may be classified into to major groups: artificial intelligence approaches and decision support systems.

Artificial Intelligence approaches such as Expert Systems (Zahedi 1993; Turban and Aronson 2004) are particularly useful for addressing interpretation, prediction, diagnosis, planning, monitoring, and control problems. Both stand and landscape management and conservation have been supported by these systems. For example, the Ecosystem Management Decision Support (EMDS) system (Reynolds 2001) has evolved an integrated expert system approach to multifunctional forest management.

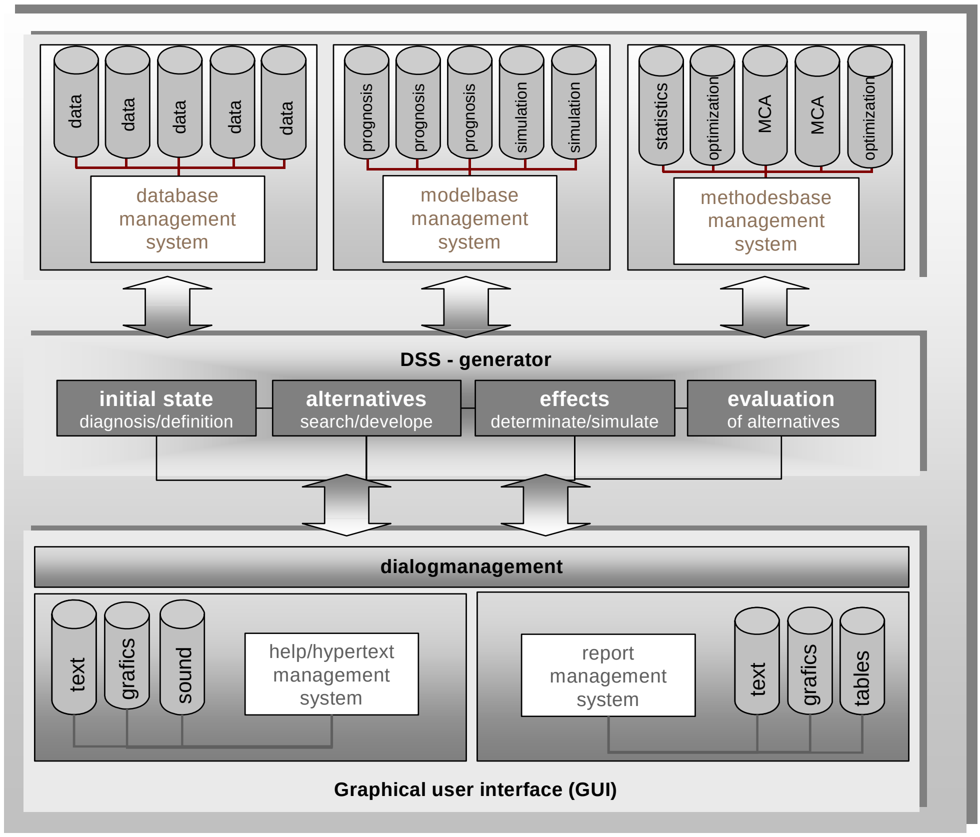

Figure 1. Architecture of a decision support system (from Reynolds et al. 2005).

The general architecture of a regular Decision Support System (DSS) encompasses database and modelbase/methodsbase management systems (DBMS and MBMS, respectively) and a graphical user interface (GUI). Data is organized and made available by the DBMS to models in the MBMS that process and convert it into information and recommendations to the decision-maker. The GUI supports the communication between the system and the decision-maker. Models for spatial optimization such as exact and the heuristic approaches discussed in this review integrate the MBMS within the DSS (Figure 1).

Reynolds et al. (2007) reviewed some of the more important and most recent developments of DSS in forest management, including examples from North America, Europe, and Asia. Reynolds et al. (2007) further reviewed some of the more significant methodological approaches such as artificial neural networks, knowledge-based systems, and multi-criteria decision models. According to the authors, a basic conclusion that emerges from the review is that the availability of DSS in forest management has enabled more effective analysis of options and implications that alternative management approaches have for all components of forest ecosystems. This effectiveness is largely a consequence of modeling approaches such as the ones discussed in this review.

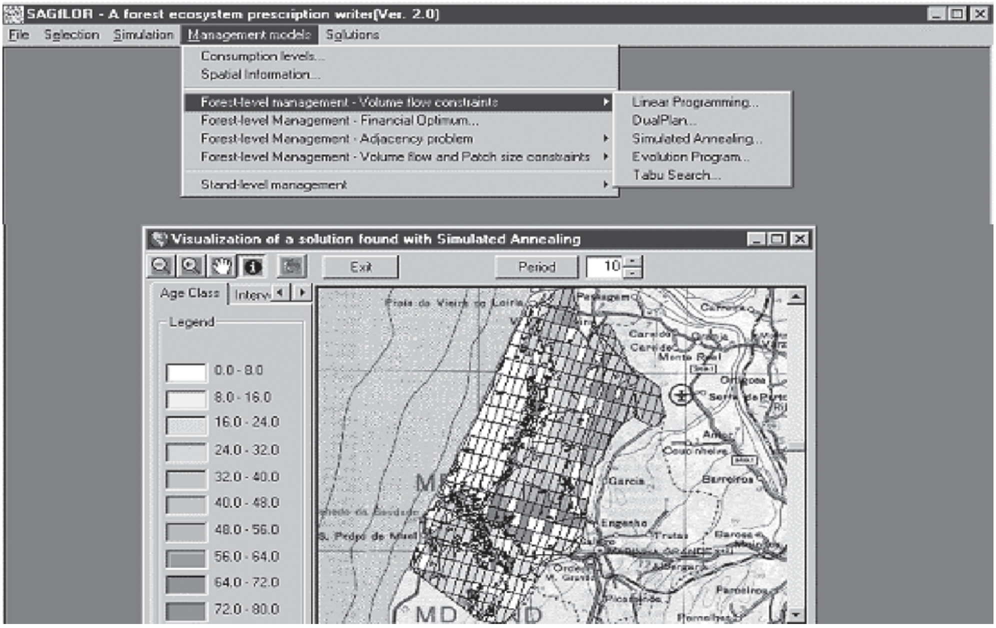

Figure 2. Interfaces to select management models from the MBMS and for visualization of solutions by the Decision Support System SADfLOR.

Figure 3. Visualization of a forest stand using the Decision Support System MONTE.

Discussion

Spatial optimization is a key tool for effective landscape design. Both exact and heuristic approaches have been used extensively to address sustainability concerns in forest management. Applications in Portugal demonstrate the potential of these approaches when integrated in adequate decision systems to enhance Mediterranean forest management planning.

For example, the DSS SADfLOR completed in Portugal in 1999 (Borges et al. 2003) has been used successfully in the context of several outreach efforts e.g. management planning for National Forests, community forests, industrial forests and non-industrial private forests (Figure 1). It has also been developed to assist with strategic management planning for the major pulp and paper industry groups (Portucel, Silvicaima and Celbi Stora Enso) in Portugal (Borges and Falcão 1998; 1999 and 2000) and Mediterranean forest ecosystem management (e.g. Ribeiro et al. 2004; Falcão and Borges 2005). A client-server architecture was developed in the context of projects aimed at developing an integrated planning system for the pulp and paper industry (Grupo Portucel Soporcel) (Ribeiro et al. 2005). Recently, the programming of process-based models added capabilities to address climate change scenarios in forest management. 3D visualization capabilities have also been developed that may enhance solution reporting (Falcão et al. 2006). Current research and development efforts focus on evolving of better capabilities to support group decision making and sustainability assessments (Martins and Borges in press).

In Spain, another forestry decision support system – Monte (Pukkala 2003) – was also developed based on advanced forest simulation tools and heuristic optimization techniques. The simulation system includes models to predict the risk of forest fires (González et al. 2006), the habitat suitability of some bird species as well as the carbon balance of a given management schedule. Complex spatial Mediterranean forest planning problems (integrating landscape ecological considerations or the risk of forest fires) can be addressed by MONTE (see Palahí et al. 2004; González et al. 2005).

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Reflection on the Current Status and Future Challenges of Forest Management Planning System in Turkey

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Abstract

Turkey has just started to embrace ecosystem based multiple use forest management philosophy focusing on biodiversity and participation. As such, sustainable management of forest ecosystems creates a great challenge to forestry sector in Turkey due mainly to various ecosystems, harsh environment, heterogeneous socio-cultural set up and unstable economy. This paper, therefore, presents the current forest management philosophy, regulations, socio-cultural and organizational structure, and implementation of management activities across the country. The forest management system was restructured to accommodate new developments such as biodiversity conservation, participation of stakeholders, multiple forest values, GIS based spatial database and modeling with operations research techniques with new forest management guidelines and ecosystem based planning philosophy. The paper concludes that the current management system, which is about to change, is of neo-classic European style with sole wood production in focus and calls for urgent change in planning concept towards ecosystem based multiple use forest planning.

Keywords: forest management planning; biodiversity conservation; multiple-use forest management; Turkey

1. Introduction

Forest management focuses on the holistic integration of various forest values such as recreation, water quality, soil protection, and biodiversity in addition to the conventional wood production. Participation, biodiversity conservation and multiple-use of forest ecosystems based on the accommodation of ecological, economic and socio-cultural components without jeopardizing the ecosystem health and integrity are key factors in sustainable management of forest landscape.

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The traditional forest management system of Turkey, essentially initiated in the 1960s by the help of European counterparts, focuses primarily on wood production. With nearly 21.2 million ha of forested areas, Turkey has several distinct biogeographic regions, each having their own endemic species and natural ecosystems, providing major flyways for millions of migratory birds and embodying almost three quarters of the total plant species in Europe; as such its management is of crucial importance to the rest of the world. Almost 99.9% of the forest land base belongs to the state who has managed all forests across the country since 1937. The planning process is centralized in Turkey: forest management plans are primarily prepared by the planning department at the headquarters of the General Directorate of Forestry. Both state and private companies have the authority for timber cruising and management planning. Even-aged management (62%), uneven-aged management (5%) and coppice management (33%) have all been practiced across the country while the latter has just been halted recently. Age class regulation method is practiced for a given rotation age to set and achieve the regulated forest as a target for both even-aged and coppice forests while size class distribution is controlled for even-aged forests. Management plans are prepared at forest planning unit level where a district forester is responsible for plan implementation.

Recently though, the focus of forest management planning has changed towards ecosystem management (Grumbine 1994; Baskent and Jordan 1995; Baskent and Yolasığmaz 1999) meaning a multiple use management concept. As such, various forest values such as recreation, water production, soil protection and biodiversity conservation are just being participatory integrated into the plans using advanced information technologies. Establishing an effective multiple use forest management system based on participation and biodiversity conservation is a great challenge for the Turkish forestry sector. This paper, therefore, presents the current management philosophy, regulations, socio-cultural and organizational structure and implementation of management activities across the country. The forest management system is evaluated based on the concept of sustainable forest management, while major pitfalls of the system are documented and some avenues for sound management are presented. Primary challenges relate to the effectiveness of conservation program, availability of coherent biodiversity data, inadequacy of institutional capacity; awareness, training and common understanding of biodiversity and protected area concept; coordination among the related institutions and stakeholders, and willingness and enthusiasm of authorities to accept and implement the concept.

2. The current status of forest management system

2.1 Forest resources and ownership

According to the latest national forestry inventory, forests occupy about 21 million ha of landbase, corresponding to 27% of the country area. Productive forests cover only about 48.2% (10.0 million ha) of the total forest area and the remaining 51.8% (10.7 million ha) carries only degraded or severely degraded unproductive forests. Conifer forests dominate the high forests, occupying 32% of the total forest area, which corresponds to 80% of the high forests. Deciduous or high hardwood forests occupy only 8% of the total forest area, which corresponds to 20% of the high forest (Konukçu 2001). About 80.2% of the total forest area is primarily managed for timber production and 15.8% is allocated to conservation areas including forest recreation sites and protection forests. Small part of the total forest area (4.0%) is allocated to nature protection and biodiversity conservation including national parks, nature parks, nature conservation areas, nature monuments, seed stands, gene conservation forests, cloned seed orchards and protected areas (Baskent et al. 2005).

The standing tree stem volume in Turkey is about 1.2 billion m^3, and the total volume increment is about 34,270,000 m^3/year. The average annual increment is about 1.96 m^3/ha, while it is 3.42 m^3/ha for productive coniferous high forests and 0.28 m^3 for degraded coniferous high forests. For the deciduous forests the figures are 3.66 m^3/ha, 5.16 m^3/ha and 0.51 m^3/ha respectively. Average annual increment for the high forests is 2.02 m^3/ha, for the productive coppice forests 2.39 m^3/ha and for the degraded coppice forests 0.20 m^3/ha. These figures indicate that the present increment and productivity of the Turkish forests are considerably low and below their potential.

Of the 21 million ha of forests, 99.9% is owned and managed by the state. Since 1937, state management has been dominating in Turkish forestry. The state forest organization (General Directorate of Forestry, GDF) operates under the Ministry of Environment and Forestry and is, however, unable to *properly* manage the forest areas due to a number of land ownership related issues. First, since legal forest boundaries have not been completed as of yet for proper ownership and land use titles, many forest areas are still under dispute due to social conflict. Second, landbase is not properly allocated to effective land use categories again creating technical problems and conflicts among land use sectors. Third, traditional land use rights heavily claimed and illegally exercised by the local people create further problems in preparing and implementing management plans. Management activities are usually hindered within the areas adjacent to local residential areas or villagers due to social conflicts. Last, while the constitution protects the forested areas to remain forests, ironically approximately 2.6 million ha of forest areas have been lost since 1950. Thus, the unresolved ownership problem acerbates the plan implementation.

2.2 Social aspect of management

As far as current forest management planning process is concerned, there is no room for the community to involve in defining objectives and determining the actions. To a certain degree, however, the regional (topographic, ecological, social and economic) differences are considered and some forest values are integrated in the plans. Due mainly to technical inability, unwillingness and orthodox forest management administration system, various other forest values in Turkey are not currently part of forest management planning system although researches do suggest that the Turkish Forest Service should change the guidelines towards a holistic management of the forest ecosystems. Regardless, timber production oriented management plans of the country provide around 400 million dollars economic income to the society.

Forest resources of the country provide vital socio-economic contributions especially for local communities, which comprise around 9 million people living in approximately 19,000 forest villages. Almost all energy needs of these communities are provided as fuelwood, which is given them at highly subsidized prices from the state forests production (almost 80% of the total fuelwood production, which is around 10 million m^3). Roundwood needs of forests villagers are also provided at subsidized prices from the state forests, which are around 380,000 m^3 annually. The total value of such subsidies provided by the GDF to forest dwellers in terms of fuelwood and roundwood is estimated around 100 million US Dollars in 1997 (Erkuran 2001; Konukçu 2001).

Forest villagers play an important role in the degradation of forest resources in Turkey. They are mostly dependent on forest resources to meet their basic needs for cutting trees and grasses within forests, grazing open forest areas and collecting illegally non-wood forest products. As the forest villages are generally scattered, it is quite difficult and costly to transfer various social services to them. Due mainly to the low level of shared family income, lack of technology in the agricultural and livestock activities, low rate of produces by the villagers and insufficient knowledge about forest management, the forest villagers are unable to generate necessary incomes to satisfy their basic needs. The people therefore look for a short term gain such as firewood and fodder supply from the forest resources and driven by occasional political gains (Türker et al. 2001; Barli et al. 2005). The forest villages are therefore in a position to cause the degradation of forest resources creating unregulated forest structures. While NGOs exist in the country, they are ineffective as they have not enough experiences and the capacity regarding forestry issues at large. However, starting with the Rio World Summit in 1992 and following some international forestry processes or initiatives, community participation was seriously revisited by the Turkish forest service. As a result, forestry town meetings, workshops and seminars have been held to launch participation. The management department is keen to accommodate the views of local people and prepare participatory multiple use forest management plans.

2.3 Management planning process

Strategic and tactical plans, combined forest management plans (around 1500) are primarily prepared by the planning department of General Directorate of Forestry and are renewed for every 10 years based on management guidelines. Few plans are, however, prepared by private or consulting firms. Management objectives (generally in the form of wood production) across the country are set and implemented according to the forest management guidelines.

The process starts with a timber cruising to determine forest cover types, standing growing stock and increment. Stand types and their sizes are characterized with 1/15,000 infrared aerial photos and thus forest cover type maps are created. Stands are defined by only species mix, crown closure and development stages. Circular sample plots are distributed over the forest using 300 by 300 meters intervals. Various plot sizes such as 400 m^2, 600 m^2, and 800 m^2 are used based on crown closure of stands. The ground survey data are then compiled and the final cover type maps are generated, the yield and the increment values, thus total growing stock and the increment, calculated based on stand types. Site index is determined according to age and dominant height of stands as appropriate trees are found. However, the production capacity of approximately 10 million ha of degraded forests is unknown, creating serious problems to implement appropriate silvicultura1 activities.

Rotation lengths are generally determined based on management objectives along with biological characteristics of commercial trees and production capacity of forest sites. When forest planning was first initiated in 1963, the rotation lengths for all commercial trees were set by GDF as 80–120 years for Calabrian pine, 120–140 years for Scotch pine and Anatolian pine, 140–220 years for Cedar and oak species. After 1978 the rotation lengths were revisited and changed based on new developments in forestry (Daşdemir 1998). Starting from year 2006, however, the rotation lengths for all commercial tree species have been determined based on the recommendation from 27 regional directorates of forestry. Similarly, planning periods are determined based on the silvicultural needs of the species. For the forests with fast growing trees such as Calabrian pine 10 year-planning period is used, whereas forests with other species such as other pines and hardwood species 20 year-length is used.

Based on the forest inventory, the current age/size class structure is determined. Empirical yield tables constructed for all commercial trees are used to develop optimal age/size class structure. Using classical modified area or size regulation method (even-aged and uneven-aged) a harvest schedule is determined for only one period, leaving other periods unplanned until the rotation period. The management planning process finalizes on the phase of monitoring plan implementation. The assessment of forest management plan implementation is carried out intending to help, guide and educate the district rangers or responsible forest engineers. No legal consequences are described for them in case of mismanagement according to management planning guidelines.

All management plans are prepared based on planning guidelines that are legally binding documents in Turkish Forest Service. While the guidelines specify a certain management approach, they permit only model development and implementation with its own guidelines other than classical forest management planning approach. The guidelines are prepared and approved by the head of the Forest Service and become valid for all forests across the country with very little room for the foresters to take regional decisions. However, the guidelines are now about to change, in order to reflect new developments and regional differences to a certain degree.

2.4 Management planning approaches

Over the history, a number of planning approaches have been used in Turkey. Classical area regulation approach (even-aged management) has been widely used across the country for even-aged forests (95%) comprising mainly light demanding trees such as pine, cedar and oak. Hufnagel's size class method has been used for the uneven-aged forests (5%) comprising mainly shade tolerant species such as fir. However, few model approaches were also used. In 1982, a classical simulation technique using clear cut with highly mechanized silvicultural prescriptions was used in Calebrian pine forests. In 1990, a Turkish-German collaborative model management planning approach was introduced based on successful regeneration effort using various silvicultural prescriptions on a stand level. So far, 107 management plans were prepared and implemented in Zonguldak, Sinop, Bolu, Kastamonu and Trabzon Regional Directorate of Forests. The approach uses longer rotations and regenerating periods, l:l0,000 scale cover type map, concept of continuous forest cover and stand/sub-compartment description. Besides its successful regeneration efforts, the model left the foresters with a great problem of determining the spatial layout of harvesting activities (i.e. size, location and timing of actions on the ground) and undetermined future structure of forests (Baskent 1999).

Starting from 1998, multiple use forest management planning approach has been used in the country. While its framework and guidelines have not been fully accepted, the approach is generally recognized as the future planning concept. The approach is simple and focuses on the initial determination of forest values and netting down the forest areas accordingly. A map showing the forest values is created using a number of criteria and the subsequent management plan is prepared based on this map. It is important that multiple forest objectives can be considered in this approach again based on the potential values and services of the forest under consideration. Economic input, however, is not incorporated into management plans. No decision making techniques are used either.

2.5 Organizational structure of management planning

The forest management department under the administration of Turkish Forest Service is responsible for the preparation and assessment of forest management plans for all forest areas regardless of ownership. The department maintains 30–40 state forest management teams (FMT); each consists of one chief forest engineer and nearly two forest engineers. The department also maintains about 15 forest management specialists responsible for assessment and control of timber cruising and the preparation of forest management plans. After 1987, however, some forest management plans have been licensed out for private forest consultants to prepare them under the state developed management guidelines. Given both state and private contracts to prepare management plans, about two million ha of forest areas are generally re-surveyed and their management plans are prepared per year with a strategic target to renew the management plans for all country's forests over a 10 year period basis.

The state forest administration is geographically and hierarchically organized across the country. At the top, the forest areas are divided into 27 forest regions, where each region is administered and managed by a regional directorate of forestry. Each region is further divided into state forest enterprises (forest industries) all totaling to 228. The size of forest enterprises ranges from 10,000 ha to 100,000 ha over the country with an average size of 83,000 ha, 3–15 times larger than that of a forest district planning unit. Each forest industry is again partitioned into forest districts where a forest engineer is appointed to manage the forest resources. It is at this end that management plan is required to develop and execute. All together there exist around 1,339 management units.

A management planning unit is defined as a geographically contiguous area of forest with certain administrative and political borders, topography, and synchronized technical works. Its size ranges from 1,000 ha to 40,000 ha averaging around 16,000 ha in Turkey. Ideally, it is stated to be around 5,000 ha. The interesting issue here is that each district must have at least a management plan. Within a management unit, where a management plan is prepared, the forest can be grouped into a number of sub-planning units based on forest composition, topography, management system, forest stratification, and rotation length with consideration of forest sites. Regardless of spatial locations, the stands are grouped into sub-management units with sizes greater than a rotation period. Stands are further geographically grouped into compartments based on distinct geographic features such as roads and rivers with less than 50–70 ha of area. The compartments are used in locating, recording and controlling any management activities on the ground.

3. Trends and future challenges of forest management

A hierarchically structured, state oriented, centralized forest administration system, a classical area regulation approach, a single management objective of maximum timber production, a standardized management planning style and a wood production based inventory system are all major characteristics of the current forest management system of the country. However, multiple values of the country's forests, where about 80% is sensitive to soil erosion, are not vehemently integrated into management planning. The forest cadastre is not finalized and thus the ownership of the forested areas is unknown, creating great concern for forest management planning and implementation process. Such uncertainty also creates great concern to the forest dependent communities with about 9 million people. Long term forecasting of forest dynamics under management interventions is not exercised, failing to create alternative management options to make better decisions using decision support systems such as operational research techniques. Ongoing social pressure, illicit use of forest resources, frequent displacement of forest officers and inappropriate silvicultural treatments are some other problems. In addition, the low per capita income rate of 300 dollars of forest dependent communities creates great concern to the forestry sector.

Given the problems, however, the forest management system of Turkey is now changing into an ecosystem based multiple use concept, focusing on participation and biodiversity conservation. As a result few international projects such as Turkey-Germany consortium supported *"model forest management planning system focusing on stand-based management practices"*, GEF (Global Environmental Facility) supported *"biodiversity integrated forest management planning system"* (applied in İğneada Lagoons, Köprülü Kanyon National Park, and Camili Bioreserve areas) and BTC (Baku-Tbilisi-Ceyhan Pipeline Co.) supported *"ecosystem based participatory multi-purpose forest management planning"* (implemented in Yalnızçam forests of Ardahan) have been conducted over the last decade. While each project has its own planning approach, they all ensured that protected areas such as national parks, nature parks, nature conservation areas, nature monuments, seed stands, gene conservation forests and cloned seed orchards have been set aside for conservation of nature as well as biodiversity. With these few model case studies, the classical area/size regulation method applied across the country since 1962 has changed into a comprehensive participatory and ecosystem based multiple use forest management concept based on ecological sustainability.

The future challenge of forest management systems revolves around multiple use concept, participation, biodiversity conservation, institutional capacity building, modeling and spatial database managed with GIS and remote sensing (Baskent et. al. 2007). In fact, there are a number of technical, legal and institutional challenges confronting multiple use forest management (MUFM) during its implementation. The concept refers to the integration of forest values such as water production, erosion control, recreation, non-wood forest products and biodiversity conservation into management on a participatory basis. To implement the MUFM concept on the ground across the country a number of challenges including scientific, technical, legal and institutional challenges are required.

Scientific challenges refer to the design of conceptual framework of MUFM process. First of all, basic components are to be identified and relationship among them documented before implementing the concept on the ground. Second, scientific robustness of the concept should be tested and awareness as well as knowledge building should be exercised among the practitioner to comprehend well the concept before implementation. Third, a comprehensive method is required to characterize and quantify forest values to functionally associate to forest structure. Here, a sound growth-and-yield projection model for multiple forest values as exercised for wood production is necessary to project the development of stands characteristics over time. As well, the MUFM requires additional forest inventory such as biodiversity, forest values, economic capacity, site productivity and forest health and vitality. Here, information technologies such as GIS and remote sensing are indispensable to create and manage spatial database. Forth, effective participation of stakeholders is necessary to stratify forest landscape for various uses and then set and achieve management objectives and conservation targets. Participation is required to resolve, at least, the potential conflicts among stakeholders before plan implementation. Lastly, modeling exercises with appropriate decision making (operations research techniques) tools such as lineer/goal programming, simulated annealing, genetic algorithm and cellular automata should be utilized to develop alternative management schedules and select the best among them.

Technical challenges relate to the understanding of MUFM concept and availability of resources such as hardware-software and the expert technical people. The biodiversity protected areas and the associated special management plans (i.e., grazing plan, recreation plan, and ecotourism plan) within a forest management plan can be prepared and implemented by the field officers in accordance with the chief forester who has the final authority in the field. Here, both the planners and the implementing officers should coordinate each other during the plan preparation and implementation. The capacity of the technical people as well as the infrastructure for information technology is needed.

Legal challenge refers to the enhancement of current forest management guidelines. Given the existing laws and regulations related to forestry and nature conservation in Turkey, there is no truly identified authority of interest groups and stakeholders responsible for biodiversity conservation. To secure the biodiversity integrated forest planning, new forest planning regulations incorporating views and suggestions of stakeholder should be urgently completed and issued. Also, the new regulations should permit the practices of ecosystem based multiple-use forest planning concept. The new regulations should allow the resolution of ownership problems of forest lands so that the land stratification would be based on and become effective.

For the *institutional challenge*, strong cooperation is needed within the departments of forest service and between the other related institutions. Forest Service of Turkey is structured with central and rural organizations to prepare and implement all forest management plans across the country (Baskent et al. 2007). Common organizational problems include lack of technical staffs in the field. Second, maintenance of enthusiastic field officers with biodiversity conservation background is extremely rare, thus continuous training and education program is necessary to overcome the problem. Since biodiversity is recognized in a hierarchical structure starting from genes, species, ecosystems and process, specialties of different disciplines and interdisciplinary approaches are necessary for effective biodiversity management. A biodiversity management authority unit is necessary to have the continuity in gathering, monitoring, synthesizing and formatting biodiversity data that they can be used to integrate into the forest management planning process.

Given the defined future challenges, the following recommendations would improve the management planning process in Turkey.

1. The socio-economic structure of the forestry-dependent communities should be enhanced to relax the continuous pressure on forests by allowing effective participation in planning process, creating awareness through town meetings and workshops, allowing their involvement in collecting and marketing non-wood forest products and providing incentives with low rated financial credits for utilizing modern production methods.
2. Ownership problem should immediately be resolved using information technologies such as GIS and remote sensing. Forest sites should be determined and classified as the basis for land titles. Historical rights-of-use of the local people having legal ownership documents should be respected. The regulations should not be frequently changed in resolving the land ownership problem to cause the confusion.
3. Wider forest management objectives (products and services) should be incorporated into plans based on the demands of people. Dynamic rotation ages and periodic forest utilization areas should therefore be determined according to these multiple objectives.
4. Forest management plans should be monitored and assessed regularly. Advanced information technology such as GIS, database management systems and remote sensing, should be used to automate and enhance the management planning process.
5. Regional field foresters should not be subjected to *frequent* lateral changes, which cause major problems during the implementation phase. Current personnel policy should be revised to fill the vacancies in forest districts immediately.

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Modeling Forest Dynamics with an Empirical Approach to Support Stand Management: The Case Study of Mediterranean *Pinus pinaster* in Central Spain

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Abstract

Adequate knowledge on forest stand dynamics is crucial for forest ecosystem management. Four different aspects should be covered in studying stand dynamics from a modeling point of view: the processes involved, the data requirement, the modeling approach and the output required. In this paper, the strategy used to develop a stand model for Mediterranean Maritime pine (*Pinus pinaster* Ait.s sp. *Mesogeensis*) in Central Spain is described. Different processes have been included in an empirical model: regeneration, growth, mortality and coarse woody debris dynamic. Our model is compatible with the National Forest Inventory design and is adequate for use in practical forestry in Mediterranean areas where foresters apply and extensive silviculture.

Keywords: regeneration; silviculture; inventory; growth; woody debris

Introduction

Pinus pinaster Ait. (Maritime pine) is widely distributed in south-western Europe (France, Italy, Spain and Portugal) and north-western Africa (Algeria, Morocco and Tunisia). Its distribution is not continuous due to geographic isolation (mountain ranges, sea, etc) and ancient human activities in the Mediterranean Basin. Currently, Maritime pine is the most widespread conifer species in Spain (nearly 1,200,000 ha) occupying a great variety of sites. In Spain, Maritime pine populations show a high level of genetic diversity as well as an important genotype-by-environment interaction that favors local adaptation to ecological conditions (Alía et al. 1997).

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The most common forest dynamic models can be classified as either empirical or process-based. Empirical models are more effective for assessing alternative forest management regimes, while process models can explore tree and stand responses to untested field conditions because they simulate fundamental eco-physiological processes. Empirical models can be classified by the resolution of predictions: whole-stand models, size-class models and individual-tree models. The individual tree models attempt to represent the stands as mosaics with cells represented by individual trees whose development can be determined, or not, by variables dependent on the distance between them (Munro 1974; Pretzsch et al. 2002) In the case of those models dependent on the distance, competition indexes are used where it is necessary to know the position of each one of the trees which compose the stand while in those independent of the distance this level of detail is unnecessary. In spite of what we might intuitively think, up to now no clear superiority has been detected of the indexes dependent on the distance over the independent ones (Tomé and Burkhart 1989; Biging and Dobbertin 1995). The data obtained from individual tree models can be translated into stand results by means of a simple aggregation. However, transferring the stand results to individual tree results is more complicated since complex distribution rules must be developed (See, among others, Somers and Nepal 1994; Ritchie and Hann; 1997a, 1997b) and with many determining factors (relative tree sizes, crown factor, etc.).

Different processes should be included in a forest model: regeneration, growth, mortality, coarse woody debris dynamic, etc. Regeneration is a key process in forest stand development. With appropriate management, this phase constitutes a basic tool of Mediterranean extensive silviculture, as the use of natural regeneration promotes biodiversity and the sustainability of this particular species. It is therefore important to be able to assess and predict stand regeneration, in relation to certain ecological and silvicultural parameters. Growth and yield models contain detailed information about stand development and are therefore important tools for evaluating expected responses to proposed silvicultural treatments. The appropriate structure of a growth model is determined by specific questions that the model is expected to address and by the resolution of the available data. An individual-tree model has been developed. This model contains different sub-models (diameter growth, height growth, mortality, site index, ingrowth, taper equations, etc.). Within all the events which forest dynamics may be broken into, mortality is the least understood of all. Regular mortality is usually controlled by thinning and dead trees are removed in order to avoid diseases. However, biological and structural diversity enhancements imply that dead trees, at least to some extent, must remain in the forest. Development of mortality equations provides a useful tool for determining the probability of dead tree occurrence in stands. Additionally, a two-step snags and logs abundance model has been developed to understand the process of snag and log abundance. In this case, we use an empirical approach for developing the model to support forest management at stand level. Different statistical methods have been used to develop the whole model. Logistic, linear and non-linear and principal components analyses have been used in different sub-models.

Dead wood plays an important role in ecological processes in forest ecosystems. Although decaying logs and snags are recognized as an important component of forest dynamics linked to biodiversity (Carey 1983; Hamilton and Brickell 1983; Bravo-Oviedo et al. 2005), there is a lack of knowledge of dead wood dynamics in Mediterranean forests, where externalities such as biodiversity maintenance has a great importance. In managed forests under a sustainable yield paradigm, dead trees have been minimized to avoid pest problems and hazards. Thus, trees killed by insects, diseases and fire are commonly harvested immediately, if economics and accessibility permit. Currently, the increasing importance of biodiversity and the carbon pool have led managers to maintain and promote dead wood in managed forests. Forest and wildlife managers have suggested that 5 to 10 snags per hectare are adequate (Eseen et al. 1993) for maintaining the biodiversity.

The data base used to develop the model and data requirement for using the model are key factors because it has been demonstrated that the data have a considerable influence upon model projections (Hann and Zumrawi 1991).

The objective of this paper is to present the general structure, and its state of development, of an individual-tree model for stands of *Pinus pinaster* in the Southern Iberian System of Spain.

Model structure

The model developed consists of various interrelated sub-models:

- Initialization and complementary models
  - Initialization sub-model
  - Site quality sub-model
  - Ingrowth sub-model
  - Stem taper sub-model
- Regeneration model
- Growth model
  - Diameter growth projection sub-model
  - Height growth projection sub-model
- Mortality model
- Coarse woody debris model

As well as these sub-models for developing the combined model and to be able to use the data, it was necessary to elaborate a series of auxiliary relationships (thickness of bark equations, etc.).

Initialization and complementary models

Initialization sub-model. This sub-model allows us to estimate precise variables for the correct development of the model but which are not normally measured by forest managers. In the case where measurements are available it is preferable to use these rather than to make an estimate by means of this sub-model. The fundamental equations of this sub-model allow us to predict the total height of the tree, the height of the base of the crown, the maximum breadth of the crown and the height at which the maximum breath is to be found. The equations described (Eq. 1–4) are as follows:

$$HT = 13 + (a_0 + a_1 \cdot Ho - a_2 \cdot Dg) \cdot e^{\left(\frac{-a_3}{\sqrt{DBH}}\right)} \tag{1}$$

$$HCB = \frac{HLCW}{1 + \exp^{\left[a_0 + a_1 \cdot \left(\frac{SBA}{HT}\right) + a_2 \cdot SBA + a_3 \cdot \ln(SBA) + a_4 \cdot BAL\right]}} \tag{2}$$

$$HLCW = \frac{HT}{1 + \exp^{\left[a_0 + a_1 \cdot HT + a_2 \cdot BAL + a_3 \cdot SBA\right]}} \tag{3}$$

$$LCW = \left[a_0 + a_1 \cdot DBH + a_2 \cdot DBH^2\right] \cdot \left[CR^{\left(a_3 + a_4 \cdot CL + a_5 \cdot \frac{DBH}{HT}\right)}\right] \tag{4}$$

Where HT is the total height, Ho is the dominant height, Dg is the quadratic mean diameter, DBH is the diameter at breast height, HCB the height at the base of the crown, BAL is the sum of the basal area of the trees larger than the subject tree, Sba is the basal area of the plot, HLCW is the height at which the maximum width of the crown is reached, LCW is the maximum width of the crown, CR is the crown ratio and CL is the length of the crown.

Site quality sub-model. Site index curves developed by Bravo-Oviedo et al. (2004) with the data for the permanent plots of the CIFOR-INIA have been used. Bravo-Oviedo et al. (2004) used the Bailey and Clutter (1974) model in an algebraic difference form (Equation 5), in which the reference age is 80 years, reaching a site index of between 9 and 24 meters.

$$H_2 = \exp\left[a_0 + \left(\ln(H_1) - a_0\right)\cdot\left(\frac{T_2}{T_1}\right)^{-a_1}\right] \tag{5}$$

Where H_2 is the dominant height at age T_2, H_1 is the dominant height at the age of T_1 and a_0 and a_1 are the parameters.

Natural mortality sub-model. For the two species, a logistical model was developed for predicting the survival probability of each tree. The logistical model has the following general structure:

$$P = \left(1 + e^{-(\alpha + \sum b_i X_i)}\right)^{-1} \tag{6}$$

Where is the constant term, is a linear combination of parameters b_i and independent variables X_i

Independent variables were the relation between the normal diameter of the tree and the quadratic mean diameter of the stand and the interaction between the seasonal quality and the site index. A more detailed description of this sub-model can be found in Bravo-Oviedo et al. (2006).

Ingrowth sub-model. This sub-model consists of two differentiated parts, since at the first stage we predict whether ingrowth exists in a particular plot by means of a logistic regression (of equal structure to those of eq.7), and afterwards the magnitude of the ingrowth in the plots where this takes place is predicted by means of a linear function. The independent variables used in this sub-model are the quadratic mean diameter and basal area. A detailed description of this model can be found in Bravo et al. (2007).

Stem taper sub-model. This sub-model consists of a taper equation which allows us to know, by integration, the volume for different types of products. The general structure of the model is the following:

$$d = \left(1 + \beta_0 \exp(\beta_1 * h)\left[\beta_2 * D * (1-h)^{\beta_3 + \beta_4\left(\frac{HT}{D}\right) + \beta_5(1-h)}\right]\right) \tag{7}$$

Where d_i is the diameter at different heights, h_i is the height where d_i is measured, HT is the total height and D diameter at breast height.

Auxiliary relations. As well as the static equations described which predict the total height, the height at the crown base, the maximum width of the crown or the height at the said maximum crown width, it is necessary to create other auxiliary equations in order to allow us to have the base for the backdating process available. When the current diameter measurements are available over bark and past growth inner bark, it is necessary to create an equation which relates the diameters with and without bark. This relation is adjusted simultaneously so that its use can be reversible. Thus, the equation is:

$$DOB = a_0 \cdot DIB^{a_1}$$

and in consequence its simultaneous:

$$DIB = \left(\frac{1}{a_0}\right)^{\left(\frac{1}{a_1}\right)} \cdot DOB^{\left(\frac{1}{a_1}\right)}$$

Where DOB is the diameter at breast height over bark and DIB is the diameter at breast height inner bark

Regeneration model

The regeneration model consists of two components (Rodríguez-García et al. 2007): a logistic model (analogous to that described later when we talk about the coarse Woody debris model) which allows us to predict the success of the regeneration and a linear model which allows us to quantify the regeneration present in each stand and its height growth.

Growth model

Diameter growth projection sub-model. The diameter growth model is based on a classical model previously demonstrated in other species (Hann and Larsen 1991; Zumrawi and Hann 1993; Bravo et al. 2001). The general structure of this sub-model is the following (eq. 8):

$$\Delta DBH = e^{\left[a_0 + a_1 \cdot \ln(DBH+1) + a_2 \cdot DBH^2 + a_3 \cdot \left(\frac{CR+k_1}{1+k_1}\right) + a_4 \cdot \ln(SI) + a_5 \cdot \left(\frac{BAL^2}{\ln(SBA+k_2)}\right) + a_6 \cdot \sqrt{SBA}\right]} \quad (8)$$

Where ΔDBH is the diameter increase, DBH is the diameter at breast height, Cr is the crown ratio, SBA is the stand basal area, SI is the Site Index and k_1 and k_2 are constant parameters in each equation.

Height growth prediction submodel: The developed height growth model is based on different approaches from PROGNOSIS model (Stage, 1973), a polinomic equation in its logarithmic form. The basic structure of this height growth submodel is formulated as follows (eq. 9)

$$\ln(\Delta \mathrm{HT}) = \mathrm{a}_0 + \mathrm{a}_1 \cdot \ln(\Delta \mathrm{DBH}) + \mathrm{a}_2 \cdot \ln(\mathrm{DBH}) + \mathrm{a}_3 \cdot \ln(\mathrm{HT}) + \mathrm{a}_4 \cdot \mathrm{CR} \quad (9)$$

Where ΔHT is height growth, ΔDBH is diameter growth, DBH is diameter at breast height, HT is total height and CR is crown ratio.

Coarse woody debris model

A two-step regression approach was used to model the presence of CWD. In the first step, a logistic model to predict the probability of the presence of CWD in a specific plot was fitted and in a second step, a linear model made it possible to quantify the snags in basal area (m^2/ha) terms and the logs in volume (m^3/ha) terms.

Before obtaining the best logistic model of CWD, a previous study of the variables was carried out. Principal Component Analysis, Discriminant Analysis and the Score variable selection in logistic regression (probability of 0.05) were able to prove the main characteristics of the variables and the main correlations. A similar set of independent variables, transformations and combinations were used for the two stands in the logistic process. Input parameters for this model were the variables that describe the development of the stand, site conditions variables, climate characteristics and forest management conditions. Species, soil texture, organic soil matter, pH, soil type, stoniness, slope and thinning in the past 15 years were considered categorical variables. The information obtained from applying the score and stepwise variable selection method was combined with an understanding of the biological mean of selected parameters.

The logistic model is as shown in Eq. 10 here P is the probability of the event modelled, in this case the presence of coarse woody debris, that is bound between 1 (presence) and 0 (absence), α the intercept term, $\Sigma b_i X_i$, the linear combination of parameters b_i and independent variables , and e is the base of natural logarithm.

$$P = \left(1 + e^{-(\alpha + \sum b_i X_i)}\right)^{-1} \tag{10}$$

Independent variables in the resulting logistic regression equations were used if significantly different from zero (P ≤0.05). The goodness-of-fit was studied by means of the Hosmer and Lemeshow (1989) test and the Akaike information criterion (Zhang et al. 1997). A LOGISTIC procedure from SAS 8.1 statistical program was used in the process (SAS Institute Inc 2001). The ROC curves of each model were used to compare the accuracy of different logistic regression models.

The linear component of this model allows us to quantify snag and log abundance (in basal area and volume terms, respectively) in those stands in which, on the basis of the logistic component, it is determined that there are snags or logs. The linear component tested is as follows (Eq. 11):

$$y = a_0 + \sum a_1 X_i \tag{11}$$

where y is BA_{snag} (basal area of snags in m^2/ha) or Vol_{logs} (volume of logs in m^3/ha). Tested are QMD, the quadratic mean diameter (m), Ho, the dominant height (m), N the number of trees of the stand (tress/ha), BA, the basal area of the stand (m^2/ha), BA_msp, the basal area of the dominant plot's species (m^2/ha), N_msp the number of trees of the dominant plot's species (trees/ha), n, the number of non-inventoried stems (trees/ha), S, slope (%), Alt, the altitude (m), Exp, the exposure, R, the rainfall (mm), R_June, R_July, R_August, the rainfall for the months of June, July and August, respectively (mm), MaxT, the maximum temperature (ºC), MeanT, the mean temperature, (ºC), MinT, the minimum temperature, (ºC) and Rad, the radiation (10 kJ/(m^2×day×µm). and are the parameters.

The joint model's adequacy was analyzed using the determination coefficient, by means of the adjustment of a straight line between the real value and the predicted values and by calculating the bias of the model to determine the accuracy of the joint two-step model.

Data-to-model development

Regeneration model

Sampling was conducted systematically by using a 100 m^2 grid with a random start. A total number of 129 plots in the province of Soria and Segovia were displayed over the selected stands in the years 2004 and 2005. Plots were divided into four quadrants of 4.9 m^2 of surface and a fixed radius of 2.5 meters. In every quadrant, total height, basal diameter, diameter at breast height (1.30 m.) and age of all the seedlings were measured (Table 1). Vigor state, seedling social position (dominant/suppressed) and other signs of damage or grazing were recorded. Furthermore, other variables related to stand and site (Table 1), such as slope,

Table 1. Main characteristic of the data base used in the regeneration model.

Variable	Mean	Standar Deviation	Min	Max
Altitude (m)	951.35	43.41	903.3	1010
Slope (º)	6.29	3.26	5	20
Seed source distance (m)	13.63	9.42	1.4	40.3
Basal Area (m^2/ha)	4.33	3.79	0	21
Woody debris cover (%)	24.24	20.14	0	90
Tree cover (%)	0.43	2.94	0	25
Seedling height (cm)	37.6	156.41	0	1000
Shrub total height (%)	17.37	16.78	0	80
Shrub total height (cm)	35.13	29.79	0	90
Soil pH	6.32	0.26	5.93	6.68
Soil Organic Matter (%)	1.87	0.84	0.92	3.48
Soil sand (%)	84.26	4.67	77	94
Seedling ages (year)	3.62	2.86	0	14
Seedling density (trees/ha)	8222.21	8504.70	0	34285.71

aspect, position, site preparation, residual basal area, distance to the nearest tree, vegetation cover and logging slash were recorded to be able to predict stocking and density of regeneration. Stocking was defined as the percentage of plots on which one or more healthy and dominant seedlings were established. New samplings to extend the model will be developed in the near future in Central Spain. Moreover, this model will be enriched with data from stands in which advanced regeneration exists.

Growth, initiation and complementary models

The data used for the development and adjustment of the parameters of a forest growth model are the key to being able to understand their area of application and their possibilities for use. The majority of the models which we presented have been carried out starting from the network of plots which is maintained by the Sustainable Forest Management Group of the University of Valladolid (Palencia Campus, Spain). The measurements of these plots were carried out between the years 2001 and 2003. The distribution of the plots attempted to cover all the range of qualities, densities and ages (dividing the rotation in four classes of unequal ages: young stands, stands close to the culmination of current annual increment, mature stands and stands of advanced ages). Logically, the last age class, especially in the best qualities, is not completely represented The plots, of a circular shape, are composed of three sub-plots of a radius of 5, 10 and 15 meters with a minimum tally diameter of 75, 125 and 225 mm., respectively. This type of plot was chosen so that the predictions of the model were compatible with the National Forest Inventory, eliminating the last crown in accordance with the results obtained by Bravo et al. (2002). In all the trees, the normal diameter and the total height was measured and a sample of the radial growth was extracted. From the growth data obtained, from auxiliary equations (quality curves, evolution of the dominant height curve, etc.) and from different static equations (thickness of bark equations and relationships between the diameter and height) we proceeded to backdate the dimensions of the trees to their measurements of 5 years ago. This process, which is called “backdating” in international literature, is described in detail by Hann and Hanus (2001). In total, the data base is composed of 3647 trees. Moreover, in 64 plots between 3 and 6 trees were felled. The

Table 2. The principal characteristics of the plots used for developing the growth models of *Pinus pinaster* (92 plots). N: number of trees per ha, G: basal area, Dg: quadratic mean diameter, IDR : index of stand density or of Reineke, Age in years, SI: site index defined as the dominant height at 80 years for the Black pine.

	N (trees/ha)	G (m^2/ha)	Dg (cm)	IDR	Age	SI
Average	824.6	38.84	27.05	778.3	76.8	14.82
Maximum	2450.9	59.64	46.66	1200.8	133	23.58
Minimum	141.5	18.35	10.64	379.5	26	7.11
Standard Deviation	470.2	9.41	7.16	184.5	28.7	4.33

Table 3. Database characteristics used to develop the coarse woody debris (snag and log) models for *Pinus* stands in Northern Spain. N: Trees per ha, BA: basal area, QMD: Quadratic mean diameter and BA_{snags}: snags expressed as basal area, V_{logs}: logs expressed as volume

Variable	Mean	Minimum	Maximum	Standard deviation
Pinus stand (n = 67 plots)				
N (trees/ha)	802.8	25.5	1584.5	341.3
BA (m^2/ha)	23.2	5.6	39.3	8.2
QMD (cm)	222.1	131.7	582.7	62.9
BA_{snags} (m^2/ha)	1.7	0.2	14.9	3.6
V_{logs} (m^3/ha)	3.3	1.4	11.8	3.9

geographical distribution of these plots embraces the natural forests of *Pinus pinaster* in the Southern Iberian System.

Coarse woody debris model

Sixty seven study plots were installed in the year 2005 in *Pinus* spp. Stands. Plots were formed for four subplots joined by two perpendicular transects. The snags inventory was carried out in the four subplots and the log inventory was performed in the transects. Snags inventory was carried out by sampling in spiral 20 trees. Starting with the trees that were closest to the plot's centre, and moving progressively away from it, it was measured if each tree was alive or dead. In tally dead trees (diameter at breast height ≥ 7.5 dbh), the variables recorded were species, snag height, diameter at breast height (DBH), decomposition status, presence of excavated cavities, azimuth and distance (taking as a reference the centre of the plot). Logs inventory was carried out in two perpendiculars transects each 50 m long, which joined the four subplots. Log was considered to down dead trees with a diameter greater than 7.5 cm and a length greater than 1m. The variables species, diameter at the interception point, length, decomposition status and wildlife characteristics were measured. Snag basal area (m^2) and log volume (m^3) were calculated for each tree. Log volume was estimated by means of the equation (Lofroth 1995): $V_i = (\pi^2 d_i^2)/ 8L$, where V: log volume (m^3/ha), d: diameter of each log (cm), L: transect length, in our case 50 m. These individual basal areas and volumes

Table 4. Model parameters for *Pinus pinaster* in Central Spain.

Submodel	Dependent Variable	a_0	a_1	a_2	a_3	a_4	a_5	a_6	a_7	Source
Initialization	HT	32.3287	1.6688	0.1279	11.4522					Lizarralde and Bravo (2007a); Lizarralde et al (2007b)
	HCB	---	---	0.0078	-0.5488	-0.0085				
	HLCW	---	-0.0041	-0.0093	-0.0123					
	LCW	---	0.1826	---	0.1594	0.0014	---			
Auxiliar	Bark	2.8708	0.8533							Lizarralde and Bravo (2007a)
Site quality	H_2	4.016	-0.5031							Bravo-Oviedo et al (2004)
Survival	Probability	2.0968	4.7358	-0.0012						Bravo-Oviedo et al (2004)
Ingrowth	Probability	12.3424	0.1108	-0.2235						Bravo et al (2007)
	Ingrowth	6.7389	-0.2235							
Taper equation	d	0.5314	0.0224	-2.0976	1.5527	5.7966	0.4716			Lizarralde et al (2007a)
Regeneration	Probality	-2.3537	0.0278	0.000364	0.0798					Rodríguez-García et al 2007
	Sapling density	1061.6112	-10.8417	-1.3773	0.8738	0.8409				
Growth	ΔDBH	---	0.2030	---	0.4414	0.8379	-0.1295	-0.0007		Lizarralde and Bravo (2007b)
	Ln(ΔHT)	4.1375	0.3762	-0.5260	0.1727	2.6468				
Coarse Woody Debris	Probability	-43.9984	2.5969	0.4001	4.6954	2.4785	2.5391			Herrero et al, 2007

for each tree were totalled for each plot and plot values scaled up to give a basal area and volume per hectare. (Table 3)

Furthermore, different tree and stand variables were calculated, including species, total basal area for each species, trees per ha., quadratic mean diameter, dominant height, basal area and the number of trees of the dominant plot's species, the number of non-inventoried stems, site conditions (soil texture, organic soil matter, pH, soil type, altitude, stoniness, slope, exposure and radiation), climate characteristics (rainfall, maximum temperature, mean temperature, minimum temperature, dry months rainfall; (obtained using a digital climatic atlas of the Iberian Peninsula, Ninyerola et al. 2005), and forest management conditions (thinning in the past 15 years).

Current status

This model is designed to be used with the data which forest managers usually record in practical forestry. It is advisable, as has already been said, use plots with shape and size compatible with National Forest Inventory or other widespread inventory method. In order to use our model, a list of trees of which the species and normal diameter is known is needed. Table 4 shows model parameters. Furthermore, it is recommendable that the total height and that of the base of the crown of each tree are also known. Once finished, it will be integrated in a computerized system which will allow that the decisions taken by the managers will be faster and more effective. Different components of the model are already at an advanced stage of development and will shortly be made available to users.

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Management Oriented Growth Models for Multifuncional Mediterranean Forests: The Case of the Stone Pine (*Pinus pinea* L.)

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Abstract

PINEA2 is an integrated single tree level model for multifunctional management of stone pine stands in Spain. The present study expounds the structure of the model, the different mathematical relations, and some of its applications. The PINEA2 model has interregional validity and a stochastic formulation, which allows the yield as well as stand and tree development to be simulated for a complete cycle under different management schedules.

Keywords: non-timber product, multifunctional Mediterranean forests, integrated model, Pinus pinea

Introduction

Stone pine (*Pinus pinea* L.) stands can be considered a typical paradigm of Mediterranean multifunctional forests. The Stone pine occupies more than 450 000 ha in Spain, representing more than 50% of the total area covered by the species in the world. Natural stands are mostly located in four regions: Northern Plateau, Central Range, West Andalusia and Catalonia (Figure 1). Since the end of the 19th century, most of the natural stone pine stands in Spain have been managed for the production of both timber and pine nuts whilst at the same time embracing other forests functions like protection against soil erosion, scenic beauty, biodiversity or recreational use. The compatibility and optimization of multifunctional forests requires specific models and tools capable of accurately predicting the production of timber as well as non-timber products under different management regimes.

Since the early 1990s, various tools have been developed in Spain aimed at the multifunctional management of stone pine stands. The most significant of these are the stand

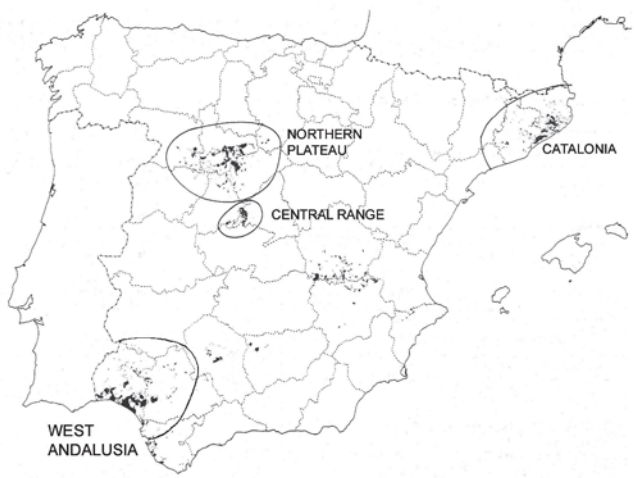

Figure 1. Distribution of the Stone pine and regions in which the PINEA2 model could be used.

level models by García-Güemes (1999), and Piqué (2003) validated for the Northern Plateau and Catalonia, respectively, as well as the single tree model for the Central Range by Cañadas (2000). These models have single regional validity and are heterogeneous with respect to the type of data used, the basic modelling unit and the statistical methodologies applied. Mathematical relations within these models were formulated as deterministic functions, constructed using short term data series for cone production (3 years) and growth (single inventory). These models lacked some basic components such as stem profile equations. Recent use of the models by forest managers has revealed common systematic trends towards overestimation in cone prediction and site quality, together with difficulties for simulating the real distribution of cone production within the stands.

PINEA2: model structure

The PINEA2 model was developed to overcome the aforementioned limitations. It is an interregional, stochastic, single-tree level integrated model, which allows the development and yield of even-aged stone pine stands to be simulated under different management schedules. These simulations are carried out in five-year stages. PINEA2 is a system composed of three basic modules: site quality, state and transition, and an auxiliary module. Each module includes input and output covariates, as well as their associated mathematical functions.

Basic input variables in PINEA2 are: stand age (years), dominant height (meters, defined as the average height for the largest 20% of trees in the plot), stand density (stems/ha) and individual tree diameter (cm) for all the trees within the stand. These variables are defined as *input variables* in each stage of the simulation.

The site quality module first estimates the site index, which is assumed to remain constant throughout the cycle. The state module is employed at each stage of the simulation using input variables recorded or estimated at the beginning of each five-year period as well as the site index to characterize the state of the stand for that period. State variables are either single-tree variables: total height, stem curve, volume of saw and pulpwood timber, crown dimensions and

Table 1. Abbreviations and units.

Stand level			Tree level		
Variable	Abb.	Units	Variable	Abb.	Units
Stand density	N	stems/ha	Breast height diameter	d	cm+
Age	T	years	Total height	h	m++
Dominant height	Ho	M	Breast height section	g	m^2
Dominant diameter	Do	cm	Crown diameter	cw	m
Basal Area	BA	m^2/ha	Height to crown base	hbc	m
Mean quadratic diameter	dg	cm	Section diameter	d_s	mm
Site Index	SI	M	Section height	h_s	dm
Natural Unit	UN	Categorical	Average annual cone production	wc	kg
			Tree saw volume	v_s	m^3
			Tree pulp volume	v_p	m^3
			5-year diameter increment	id_5	cm

+mm in function(4) ++dm in function (4)

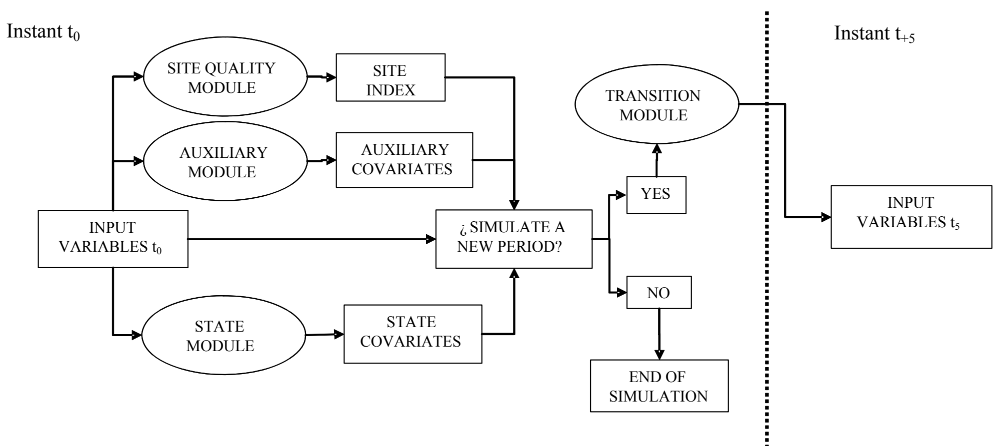

Figure 2. Flow chart for a single simulation stage of the PINEA2 model.

annual cone production; or stand level variables: stem rot probability. Other stand level state variables such as standing and cumulative volume, cumulative cone production, crown cover or mean height can be obtained by aggregating single tree state variables.

The transition module allows stand growth to be simulated by estimating the values for input variables at the beginning of the next period using site index, current input variable values, state covariates and management decisions (e.g. thinning) as predictors. The transition module simulates growth at tree level (diameter increment) as well as stand level (dominant height growth). Finally, the auxiliary module permits the calculation of auxiliary covariates derived directly from input variables such as the quadratic mean diameter, basal area, distance independent competition indices, structural indices, etc.

Table 1 shows the different variables used in the PINEA2 model whilst Figure 2 presents a schematic diagram in the form of a flow chart for a single simulation stage of the PINEA2 model.

Data

The PINEA2 model integrates seven different submodels and mathematical relations constructed independently. The data used to fit the submodels were obtained from the following three sources:

CIFOR – INIA network of permanent plots for sustainable forest management

In 1992, CIFOR-INIA began to establish of a network of permanent plots in natural even-aged stands of stone pine covering a wide range of the distribution area for the species in Spain. The network includes 470 plots located in four different regions (Figure 1): West Andalusia (192 plots, installed in year 1992), Northern Plateau (141 plots, in 1995), Central Range (72 plots, in 1996) and Catalonia (75 plots, in 2000). The plot selection attempted to reflect the range of ages, site qualities and density classes identified in each region. The plots were circular, of variable size, and included 20 trees. Breast height diameter, crown diameter, total height and the height to the crown base were recorded for each tree. Trees were positioned with respect to the centre of the plot. Two five-year diameter increment cores were taken in the five trees nearest to the centre of the plot. Plot ages were estimated by averaging the age measurements from three to five single trees. Age cores were analysed to detect rot symptoms. Diameter increment cores were taken again five years after plot establishment. Every autumn, starting in the year of plot establishment, the cones were collected in the five trees nearest to the centre of the plot. The cones cropped from each tree were classified as *sound* or *damaged*, and the cone crop from each tree was counted and weighed. The longest series (11 years) for cone production are those of the Northern Plateau and the Central Range.

Stem analysis

In 2001, forty-eight dominant trees were felled in the four regions analysed, covering all possible site qualities within each region. The total height of the felled trees was measured, and the stem was divided into sections. Discs were taken from each section, starting at stump height, breast height (1.30 m), and then every 1.25 m. For each disc, annual rings were counted. Data from these analyses were combined with those from a previous sample of 48 dominant trees taken in 1966, giving a total data set of 96 stem analyses. These data were used to develop a dominant height growth model (8) which was used as a site index model (1).

Sample trees

In 1966, the CIFOR-INIA installed a network of permanent plots to study the growth and timber yield of the most important forest species in Spain. This included 36 plots for *Pinus pinea*. From each plot, fifteen sample trees were selected, and the breast height diameter and total height for each tree were recorded. Measurements for diameter over bark were taken at stump height, at 0.5 meters, and then at every meter up to a height of 10 meters. From 10 meters up to the merchantable height (section diameter 7 cm), sections were measured every two meters. These measurements were taken by forestry workers who climbed the trees. A total of 536 trees of different sizes (tree volume ranging between 8 and 1970 dm^3) were used to construct the stem curve equation (4).

Methods

The submodels in PINEA2 were developed independently through the application of different statistical methodologies for data analysis. The functions retained from earlier versions (crown dimension and stem rot probability) were constructed using ordinary least squares regressions and discriminant analysis. New functions, specifically developed for PINEA2, shared several common methodologies, being formulated and fitted as multilevel mixed models and having interregional validity.

Multilevel mixed modelling

The data used to construct the different submodels of PINEA2 present a similar structure since observations for the dependent variable were repeatedly taken for trees growing on plots located within different regions. This hierarchical structure leads to a multilevel lack of independence in the data since an above average similarity tends to exist between nested observations from the same unit (tree, plot, and region) or crossed observations from the same year. Spatial and temporal dependence prevents the use of statistical methods based on ordinary least squares techniques (West et al. 1984; Fox et al. 2001). To deal with this type of data, a multilevel mixed approach has been widely proposed in forest modelling since the seminal works by Gregoire (1987) and Lappi (1986). Multilevel mixed models include in their formulation both fixed parameters, which are common to the population, as well as random components, specific to each level of the systematic variability sampled (tree, plot, year…). A general expression for a two-level mixed model is as follows:

$$y_{ij} = \mathbf{x_{ij}}\boldsymbol{\beta} + \mathbf{z_{ij}}\,\mathbf{b_{ij}} + e_{ij}$$

Where y_{ij} represents a single observation for the dependent variable y taken from unit i (e.g. tree) within unit j (plot); $\boldsymbol{\beta}$ is the vector of fixed parameters; $\mathbf{x_{ij}}$ is the design vector including explanatory covariates for y; $\mathbf{b_{ij}}$ is the vector including random components, distributed according to a multivariate normal with mean zero and variance-covariance matrix **D**; $\mathbf{z_{ij}}$ is a design vector associated with random components; e_{ijk} is a residual error term i.i.d. with mean zero and variance σ^2_e. The main aim when fitting a multilevel mixed model is to estimate components of $\boldsymbol{\beta}$, **D** and $\mathbf{b_{ij}}$.

Interregional comparisons

The interregional validity of PINEA2 was studied by evaluating the necessity of incorporating region-specific parameters into the models. Evaluations were carried out by comparing likelihood functions for models with regional parameters (*full models*) and without them (*reduced models*), using direct or derived likelihood ratio tests (see Khattree and Naik 1995, Huang et al. 2000), as well as by testing the level of significance for the proposed regional parameters.

Results

Site quality module

Site index is an indicator of the potential biomass productivity for a given location and species. Site index is usually considered as a key aspect of forest growth and yield modelling, since it expresses the joint effect of soil and climatic factors on forest development. The PINEA2 site quality module uses the dynamic dominant height growth function developed by Calama et al. (2003). This function was formulated as an algebraic difference equation of the Korf growth function, verifying the properties of base-age invariance for estimates, reciprocity and polymorphism. Site index (SI) is defined as the predicted dominant height (in meters) at a stand age of 100 years, and can be estimated if the dominant height (Ho) of the stand at any other age (T) is known:

$$SI = \exp\left[4.1437 + \left(\ln(Ho) - 4.1437\right) \cdot \left(\frac{100}{T}\right)^{-0.3935}\right] \tag{1}$$

Interregional comparisons were carried out and function (1) was identified as valid for predicting site index in the four analysed regions.

State module

At each stage of simulation, the state of the stand can be characterized through the input variables (breast height diameter for all the trees, stand density, age, and dominant height) together with site index. The state module includes the following mathematical functions and models:

Height-diameter generalized model
The total height for each tree within the stand at each stage of the simulation can be estimated using single tree diameter, dominant height and stand density (input variables) as predictor covariates together with dominant diameter (average diameter for the largest 20% of trees within the plot, directly calculated from input variables). Based on the multilevel nonlinear height-diameter mixed model by Calama and Montero (2004), the following function was developed:

$$h = 1.3 + \left[\left(5.5862 - 0.4563\log(N) + u_1\right)\left(\frac{1}{d} - \frac{1}{Do}\right) + \left(\frac{1}{Ho - 1.3}\right)^{1/2}\right]^{-2} + e \tag{2}$$

Parameter u_1 represents a random plot component, distributed according to a normal with mean zero and variance σ^2_u = 1.0643, e is a tree specific random error term distributed according to a normal with mean zero and heterocedastic variance $\sigma^2_e = 0.0058\ d^{1.4084}$.

Function (2) is forced to pass through the point of dominant-height dominant- diameter, leading to compatibility among single tree height and dominant stand height estimates (see *Transition module*). Predicted single tree heights are useful for calculating variables such as average height, mean arithmetic and quadratic height, percentiles for height distribution, h/d ratio, aggregation indices etc.

Crown dimensions generalized models
The PINEA2 model includes two functions developed by Cañadas (2000) and Cañadas *et al*. (2001) which allow crown diameter (cw) as well as height to the crown base (hbc) to be

estimated as a function of stand age and the relation between breast height diameter and total height (an index representing the degree of competition attained by a tree):

$$hbc = h \exp^{\left(-12.54\ \frac{d}{h} - 11.07\ \frac{1}{Tn} - 295.04\ \frac{d}{h}\frac{1}{Tn}\right)} \tag{3a}$$

$$cw = 0.813\ - 0.202\, hbc + 0.169\, d \tag{3b}$$

Tn is stand age at breast height, which is calculated by subtracting a correction factor from age T which indicates the number of years required for a tree to reach a height of 1.30 m (depending on site quality). From crown attributes, it is possible to derive other variables of interest such as crown area and volume, crown ratio, coverture, competition indices, vigour indices etc.

Stem taper equation and volume estimation

Tree volume is estimated in the PINEA2 model by using the taper equation developed by Calama and Montero (2006). This equation allows stem section diameter d_s to be predicted at a given section height h_s for all the trees within the stand at each stage of the simulation using single tree diameter and total height as predictor variables:

$$d_s = d\left(\frac{h - h_s}{h_s - 13}\right) + [1.1924 + u_2 + k_2]\left(\frac{\left(h^{1.5} - h_s^{1.5}\right)\left(h_s - 13\right)}{h^{1.5}}\right) - [2.4463 + v_2]\left(\frac{\left(h - h_s\right)^4\left(h_s - 13\right)}{h^4}\right) \tag{4}$$

Where breast height diameter (d) and section diameter (d_s) are given in mm, and total (h) and section (h_s) heights in dm; (u_2,v_2) are random plot components, distributed according to a normal bivariate with mean (0,0) and variance matrix $D=\begin{pmatrix} 0.3707 & -0.7912 \\ -0.7912 & 2.2413 \end{pmatrix}$; k_2 is a random tree parameter, normally distributed with mean zero and variance $\sigma^2_k = 0.0410$. The proposed taper function was derived from the original stem curve by Amidon (1974), modified to display logical behaviour at both breast height (where it predicts breast height diameter) and total height (where the predicted section diameter is zero). The inclusion of regional parameters was evaluated but did not improve the model.

Bole volume is estimated by the numerical integration of the taper function (4), estimating section diameter at constant height intervals along the stem (e.g. 5 cm). The volume between these intervals is then calculated using Newton's formula and the sum of these is in turn used to calculate total or partial tree volumes. Classification according to commercial end-use of timber is carried out by using interpolation techniques to estimate the height at which the tree reaches the minimum diameter for saw logs (e.g. 30 cm) and/or pulpwood (e.g. 7 cm).

Stem rot affection by Phellinus pini

Phellinus pini is a basidiomycete fungus causing severe rot on standing timber in different conifer species. Rotten timber has less commercial value, since it is useless for sawing. García-Güemes and Montero (1998) developed a discriminant model to predict the most probable level of infection and stem rot by *Phellinus pini* for a given stand. The model used stand age, basal area and dominant height as predictors to define four different levels of infection, associated with losses in the total saw timber available:

F1: Trees are not infected. No losses in saw timber volume
F2: Initial attack. 25% loss in saw timber volume
F3: Intermediate attack. 50% loss
F4: Severe attack. 75% loss

The discriminant function enables each stand to be classified according to these levels of infection:

$$\begin{aligned} &F1: \ -3.0633 + 0.0101\,T - 0.0241\,BA + 0.7773\,Ho \\ &F2: \ -6.1575 + 0.0752\,T + 0.0075BA + 0.7649\,Ho \\ &F3: \ -10.2041 + 0.1261\,T + 0.1126\,BA + 0.6216\,Ho \\ &F4: \ -12.5939 + 0.2217\,T + 0.1218\,BA + 0.1617\,Ho \end{aligned} \tag{5}$$

At each stage of simulation, the value for each function is calculated, and the stand is assigned a level of infection corresponding to the highest calculated value.

Average cone production

PINEA2 includes an empirical submodel for predicting the average annual yield of healthy cones (kg) produced by a single tree for a five year period (wc). Large interregional differences have been detected in cone production (a variable largely controlled by climatic factors and water availability); therefore, despite attempts at interregional modelling (Calama 2004), a single-region modelling approach was adopted. This submodel was developed by Calama *et al*. (2007) and validated only for the Northern Plateau region. A specific model for the Central Range, easily adaptable to PINEA2, was previously constructed by Calama and Montero (2007) and models for the other regions (West Andalusia and Catalonia) are now in development. The model for the Northern Plateau was formulated as a multilevel mixed model, including stand (N) and tree covariates (g, ratio d/dg) together with an ecological type stratification of the territory, based on climate, orography and soil attributes:

$$\log(wc+1) = 1.4796 + 4.2383\,g + 0.5539\,d/dg - 0.2320\log(N) + UN + k_3 + z_3 + s_3 + e_3 \tag{6}$$

UN is a categorical fixed variable indicating the natural strata on which the stand is located (Table 2) and k_3, z_3 and s_3 are tree, period and plot x period random components respectively, following a univariate normal distribution, with mean zero and variances of $\sigma^2_v = 0.0392$, $\sigma^2_z = 0.1148$, and $\sigma^2_s = 0.1792$; e_3 is a residual error term with mean zero and heterocedastic variance $\sigma^2_e = 0.0102\ \exp^{(0.0561d)}$

Transition module

The transition module includes all the mathematical relations which enable the value of the inventory variables (stand density, age, dominant height and single tree diameter) to be defined at the beginning of each five year period of simulation. The components of transition module are as follows:

Five year diameter increment function

The single tree growth function in PINEA2 is a re-parameterization of the empirical single tree five-year diameter increment (id5) function developed by Calama and Montero (2005). The function was formulated as a multilevel linear mixed model including random components specific to each tree, plot and period. Region-specific parameters were considered necessary for Catalonia due to deficiencies in the data series used in fitting the model for this region. The proposed function is:

$$\begin{aligned} \log(id_5+1) = {} & 2.2383 + C_1 - (0.3372 + v_5).\log(d) - 0.02664.Ho - 0.1516.\log(N) \\ & + (0.0412 + C_2)\,SI + 0.3376.d/dg + u_4 + k_4 + z_4 + e_4 \end{aligned} \tag{7}$$

Table 2. Natural units value (Northern Plateau).

Unit	Name	UN value
1	Torozos	0.4457
2	Limestone plain W	0
3	Limestone plain E	-0.4620
4	Valladolid	-0.6850
5	Nava del Rey	-0.6287
6	Viana de Cega	-0.5579
7	Iscar	-0.3241
8	Medina	-0.6903
9	Tudela de Duero	-0.5907
10	River terraces	-0.0283

Where C_1 and C_2 are region-specific parameters with respective values of -0.4805 and 0.0293 if the observation is from Catalonia, and zero if it is from another region; (u_4,v_4) are specific random plot components distributed according to a normal bivariate with mean (0,0) and variance matrix $D_4 = \begin{pmatrix} 0.08614 & -0.01890 \\ -0.01890 & 0.00529 \end{pmatrix}$; k_4 is a random tree parameter normally distributed with mean zero and variance 0.0088; z_4 is a random period effect, normally distributed with mean zero and variance 0.0081; e_4 is a random residual term, normally distributed with mean zero and heterocedastic variance $\sigma^2_{e5} = 0.1039 - 0.0222\log(d)$

Dominant height growth model

Dominant height growth is simulated using equation (1). The dynamic character of the model allows dominant height Ho_{+5} to be estimated for instant t_{+5} using dominant height, stand age at the beginning of the previous period (Ho,T) and the present age of the stand T_{+5} as predictors:

$$Ho_{+5} = \exp\left[4.1437 + \left(\ln(Ho) - 4.1437\right) \cdot \left(\frac{T_{+5}}{T}\right)^{-0.3935}\right] \quad (8)$$

Stand density evolution

During the five year interval governed by the transition module, there can be variations in the number of stems per ha. These changes are generally caused by one or more of the following factors:

- Increment due to natural recruitment of young trees
- Reduction in number of stems per ha due to natural mortality
- Reduction in number of stems per ha due to management practices such as thinning

Recruitment is not currently considered in PINEA2, which only simulates a single biological cycle for a given stand. For practical purposes, it is assumed that after regeneration practices and precommercial thinnings, a 20 year old stand is adequately established when stand density is around 500 stems/ha. A mortality function is not yet available for stone pine, although a model based on National Forest Inventories 2 and 3 is now under development.

However, the low density typical of stone pine stands results in a low level of natural mortality, except in extra-mature classes (age > 140 years). For simulations within a typical range of ages (20–120 years), a mortality rate of 1 % over five years can be considered adequate.

Finally, thinning is simulated by eliminating the proposed number of trees for a given period. Thinning is assumed to be carried out at the beginning of each period. Thinning can be systematic (randomly selecting trees) or selective (defining a given criteria such as social status, tree dimensions etc.), since tree-level information is available at each stage of the simulation.

Practical application of the model

Plot level calibration

The multilevel structure allows local (plot-level) calibration of the model using a sample of additional observations. Plot-level calibration is carried out by predicting the EBLUP (empirical best linear unbiased predictors) for the random plot components (parameters u and v in functions 2, 4 and 7). The height-diameter generalized function, the stem curve and the five-year diameter increment model can be calibrated for a new site as long as a sample of total heights, section diameters or past diameter increment (from cores taken with a Pressler borer) are measured. Calibration is accomplished as proposed by Vonesh and Chinchilli (1997), Fang and Bailey (2001) or Calama and Montero (2004). The plot calibration principle can be extended to every level of systematic variability: i.e. tree, period, region etc.

Deterministic vs stochastic simulations

Deterministic simulations in the PINEA2 model are conducted by assigning a value of zero to all the random components included in the different functions. However, although accurate unbiased estimates for average values at plot level are obtained through deterministic simulations, the real dispersion of the dependent variables (cone production, height, diameter increment) among trees is underestimated. An alternative is to conduct *stochastic simulations*, by adding a stochastic parameter to each tree, derived from a realization of the distribution of random tree components (parameter *k* in functions 4, 6 and 7).

Comparison of management schedules

PINEA2 can be used to compare different long term management schedules and silvicultural alternatives (defined by thinning intensity/periodicity and rotation length) in terms of timber and cone yield, CO_2 fixing, stand structure, etc. As an example of a practical application of the model, we present simulations for a 20 year-old even-aged stone pine stand of 1ha in size, growing in Natural Unit 6 (Table 2) within the Northern Plateau region. The site index for the stand is 15 m, and the initial density is 500 stems/ha. Initial single tree diameters were simulated using the diameter distribution model by García Güemes et al. (2002). Three different management schedules are compared for this stand:

- Timber oriented management: rotation length 80 years; systematic thinning at 35 years and selective low thinning at 50 years, reducing density down to 350 and 250 stems/ha respectively
- Cone oriented management: rotation length 120 years; three selective low thinnings at 30, 40 and 50 years, reducing density down to 250, 150 and 100 stems/ha respectively
- Timber and cone mixed crop management: rotation length 100 years; systematic thinning at 30 years and two selective low thinnings at 45 and 60 years, reducing density down to 350, 250 and 150 stems/ha respectively

Stochastic simulation is achieved by randomly assigning to each tree, at the first stage of simulation, a value randomly realized from the distribution of diameter increment tree random component k_4 (normal with mean zero and variance $\sigma^2_k = 0.0088$). Summary results for the three proposed schedules are shown in Figure 3. This type of simulation, together with economic studies, which consider the income from timber and cone production alongside the costs involved in cone collection, timber harvesting, thinning practices, soil costs etc., as well as optimization techniques, are fundamental to defining optimum management schedules for stone pine stands.

Conclusions: applications, limitations and further development

PINEA2 may prove to be an interesting tool for multifunctional management of stone pine stands, allowing stand growth and yield simulation for a given stand, comparing different management schedules. The model could be useful for forest planning at either short term local scale (forest level) or long-term regional scale, helping to define more appropriate silvicultural options according to management objectives and optimization routines. The single tree character of the model permits us to describe the state of every tree within the stand at each stage, thus facilitating the simulation of, for example, selective thinnings. Other advantages of the model relate to its stochastic formulation, which permits local calibration for new locations, as well as realistic simulation of tree-level development and production.

Despite the advantages of the model, PINEA2 presents some limitations that need to be addressed in future versions. First of all, a complete structure for the model is only available for the Northern Plateau and the Central Range, since some of the submodules (cone yield or crown dimension) are still not validated for Andalusia and Catalonia. Furthermore, from a methodological point of view, PINEA2 includes several different mathematical relations independently fitted using the same data base (CIFOR-INIA net of permanent plots) so simultaneous fitting techniques should be applied. Another important limitation of the model is that predictions do not take into account either past states of the stand or the way in which the stand has reached its current state. The inclusion of thinning factors in future modifications of the model will deal with this limitation.

Subsequent versions will include mortality and recruitment functions which are currently being developed. The inclusion of these functions will allow simulations to be performed for more than one single development cycle. An additional submodule would incorporate the biomass and CO_2 fix equations for whole tree as well as fractions, constructed by Montero et al. (2005). New functions for estimating cone damage probability and pine nut yield will improve the cone production submodel. With respect to forest hazards, a new submodel, now in development, will define fire risk as a function of horizontal and vertical forest structure.

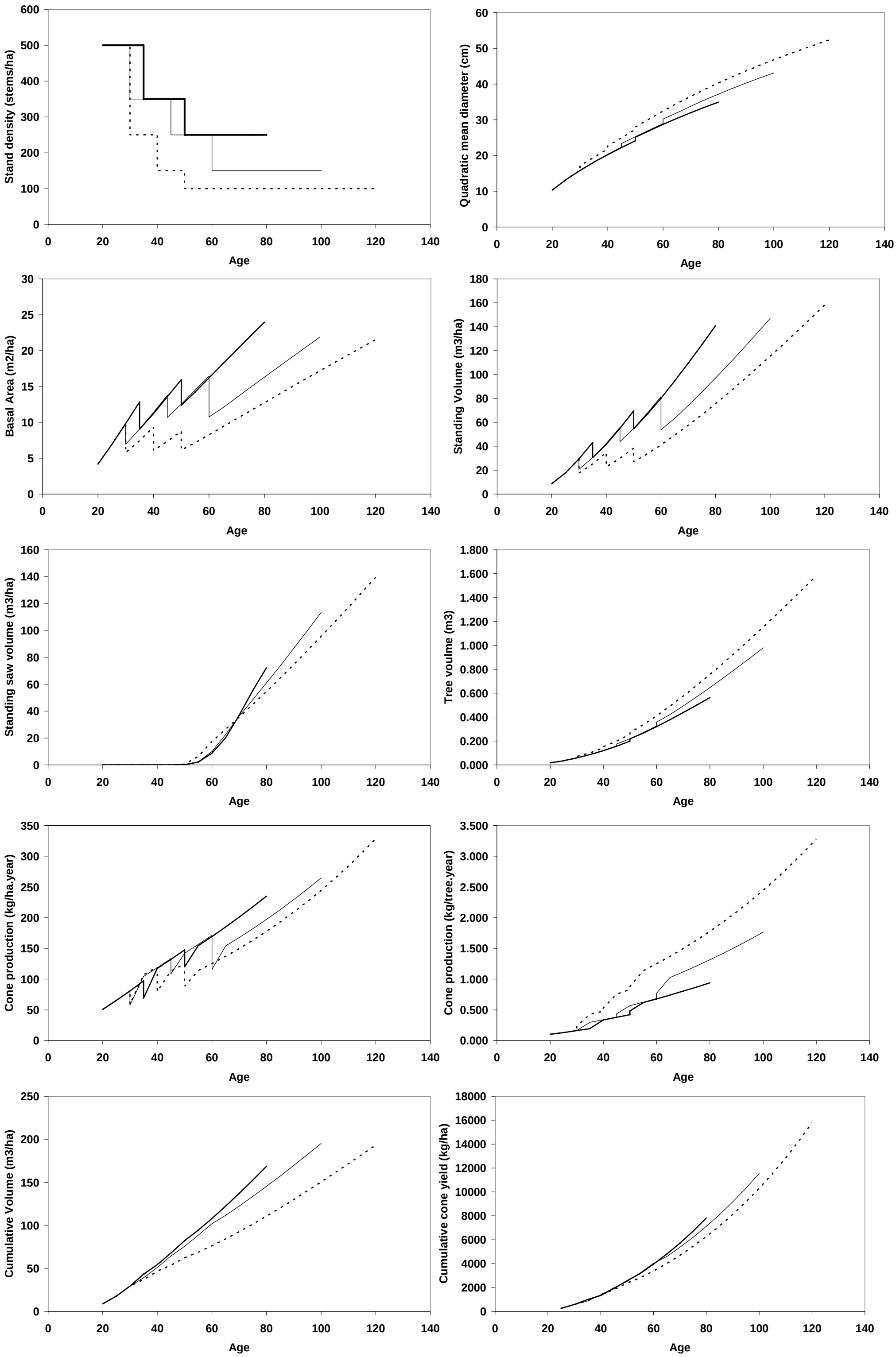

Figure 3. Comparison of the three proposed management schedules.

The extension of the model to plantations and multi-aged stands constitutes another of the basic lines of development. Finally, a major challenge to be tackled in future model formulation is the inclusion of climate and edaphic factors as predictor covariates. The hybrid character of the model will allow robust estimations of stand development and yield under different scenarios of changing environmental, economic or social conditions.

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Management Oriented Growth Models for Multifunctional Mediterranean Forests: The Case of the Cork Oak (*Quercus suber* L.)

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Abstract

In this paper we have tried to identify the specific aspects that must be taken into account when modelling Mediterranean forests. We have also described how these features have so far been incorporated into the modelling process, focusing on the case of ALCORNOQUE 1.0, an integrated growth and yield model at individual tree level for higher density cork oak forests. Finally, we have tried to identify some possible areas of research which will allow us to construct flexible, robust and above all, useful models adapted to Mediterranean forests.

Keywords: Mediterranean forests, Quercus suber, cork, integrated model

1. Modelling in Mediterranean forests

Mediterranean forests are characterized by a set of features that differentiate them from temperate forests and determine the modelling of forest growth and yield. These features can be summarized in five points:

i) Lack of importance of timber production in opposition of non-timber forest products (NTFP)
ii) Competition processes mainly driven by water and soil nutrients
iii) Occurrence of systematic disturbances such as fire and drought
iv) Complexity of the stands
v) High inter and intra-annual climatic variability

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Each of these features influences and affects modelling in Mediterranean forests. NTFP are derived from multiobjective systems providing several commercial products together with other social and environmental services. The harvesting frequency of NTFP (ranging between one and ten years) is shorter compared with timber. In addition, traditional site indices using dominant height have proved to be inadequate for estimating the yield of non-wood products. From a statistical point of view, when modelling NTFP we must consider firstly, that the largest part of non-explained variability is at the tree-within-stand level, suggesting the existence of unobservable tree factors such as microsite or genetics, and secondly, that NTFP data are characterized by a non-normal distribution.

In the Mediterranean area, growth is limited by competition for water and soil nutrients, leading to size symmetric competition processes. This competition can be intra-specific but also inter-specific (especially with the understory layer). Finally, it is necessary to consider the positive interactions among seedlings and herbaceous vegetation due to facilitation processes. In this scenario, traditional distance-dependent indices have performed worse than expected.

Mediterranean forests suffer systematic disturbances such as fires, droughts and pests as well as intense human activity. The effect of these recurrent changes on regeneration, growth and mortality is scarcely understood, but should be considered in the modelling process. Mediterranean Forests typically display high heterogeneity, even within a single stand. This heterogeneity affects stand characteristics (structure, composition, origin and density) and gives rise to a greater complexity in processes from regeneration to mortality.

In relation to this heterogeneity, Mediterranean forests show a great inter and intra annual climatic variability which results in annual values significantly deviated from the average. This climatic variability is particularly important in terms of water availability. The critical values which mark the limits of the relationship between climatic factors and processes need to be identified. Moreover, the acceptance of site quality as a constant in these ecosystems might be reconsidered in the face of this climatic variability.

Previously developed growth models for Mediterranean forests have incorporated some of these aspects into their formulation in various ways. Two of the main approaches for modelling Mediterranean forest are, on the one hand, the ALCORNOQUE 1.0 model, developed for *Quercus suber* and described in detail in this paper, and on the other, the PINEA2 model for *Pinus pinea* which is dealt with in another paper on the subject. Both models describe the characteristics of the Mediterranean forest by taking the tree as the main modelling unit. They then include distance independent competition indices, stochastic formulation and the possibility of tree and stand level calibration for new locations or the construction of specific sub-models for NTFP not governed by site indices. Other alternatives proposed for modelling Mediterranean forests consider the inclusion of stand typologies, ecological-types, diversity or structural indices within the model. Finally, new site indices based on ecological factors have recently been developed for Mediterranean species.

2. The ALCORNOQUE 1.0 model

ALCORNOQUE 1.0 is an integrated growth and yield model at individual tree level for high density cork oak forests (as opposed to woodlands which are associated with lower densities). The model consists of a system of mathematical functions which allow the evolution of growth and yield to be simulated under different silvicultural regimes. The model has been included in a computer program designed to facilitate the management of cork oak forests.

A flow chart displaying the structure of the ALCORNOQUE 1.0 is shown in Figure 1.

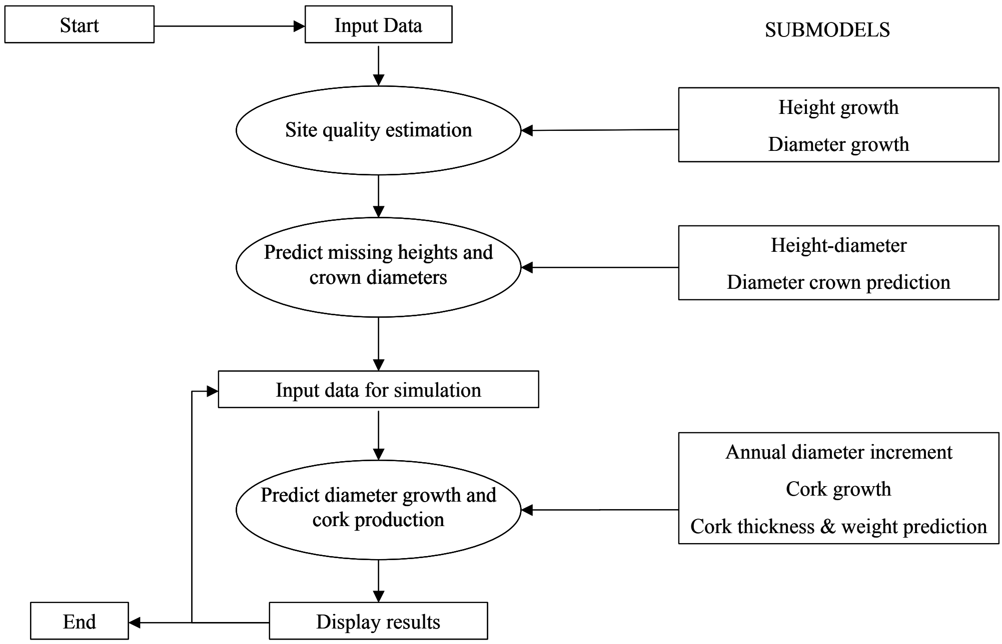

Figure 1. Structure of the ALCORNOQUE 1.0 model.

2.1 Study Area

Data to develop the model were obtained from Catalonia and the Natural Park of *Los Alcornocales* (Figure 2). These are two of the most important production areas in Spain and can be considered representative of high density Spanish cork oak forests.

2.2 Data

Modelling data were gathered from four sources: stem analysis, plots belonging to the CIFOR-INIA network, plots associated with the second national forest inventory and lastly, cork samples.

2.2.1 Stem analysis data

Stem analysis data were obtained from the Natural Park of *Los Alcornocales* and Catalonia. A detailed description of how the data were collected and recorded can be found in Sánchez-González et al. (2005).

Stem analysis data were used for modelling height and diameter growth of dominant trees (Sánchez-González et al. 2005).

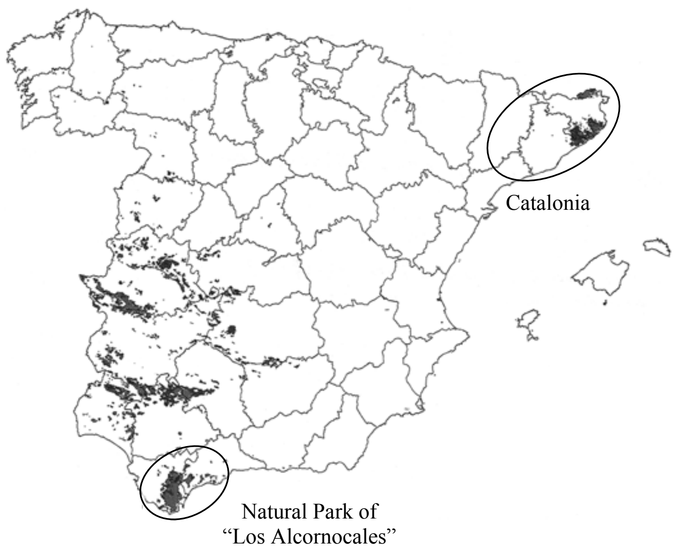

Figure 2. Distribution of *Quercus suber* L. in Spain and situation of the two studied regions.

2.2.2 Plots belonging to the CIFOR-INIA network:

In this case, the data were collected from 41 circular permanent plots of 1257 m^2, 15 square plots of 5000 m^2 from a cork quality and production trial and 18 square plots of 1250 m^2 from a debarking rotation experiment. All of them were established between 1966 and 1993 in regularly stocked stands covering a wide range of site conditions in the Natural Park of *Los Alcornocales.* They were measured twice. The first measurement was made at plot installation coinciding with a cork harvest and the second one was made at the next cork harvest (from 9 to 11 years after). 72 of these plots were used for diameter increment modelling (Sánchez-González et al. 2006), while 59 of them were used for the development of a cork thickness model (Sánchez-González et al. 2007a).

2.2.3 Second National Forest inventory

431 plots mainly located in Catalonia and the south of Andalusia were chosen from the 2NFI database using BASIFOR software (Río et al. 2001) for developing the height-diameter equation and crown diameter prediction model (Sánchez-González et al. 2007b). The plots selected met the following criteria: (1) at least 75% of the basal area was cork oak, (2) at least 50% of the number of trees per hectare were cork oak, (3) basal area above 10 square meters per hectare, and (4) number of trees per hectare above 100.

2.2.4 Cork samples

For cork growth modelling (Sánchez-González et al. 2007c), data from cork samples obtained from 432 trees in 22 permanent plots were used. A detailed description of how the data were collected and recorded can be found in Sánchez-González et al. (2007c).

2.3 Methods

To develop the aforementioned sub-models, the following methodologies were applied:

2.3.1 Difference equation method

The difference equation method was used for the height and diameter growth of dominant trees as well as for the cork growth models. Several dynamic models were tested for modelling the growth of dominant trees using the Algebraic Difference Approach. In the case of the cork growth model, candidate functions were formulated using the Generalized Algebraic Difference Approach. In both cases, the Lundqvist-Korf, Richards and Hossfeld functions were used as candidate equations.

The parameters for the height and diameter growth models for dominant trees (Sánchez-González et al. 2005) were estimated using the Goelz & Burk method (Goelz and Burk 1992; 1996). The functions were chosen according to the following considerations: goodness-of-fit, predictive ability, biological sense and compliance with the assumptions of homoscedasticity, lack of autocorrelation and normality of residuals. Once the best growth equation had been selected, the parameters of the chosen function were redefined in order to determine whether differences existed between regional height and diameter growth. In the diameter growth model for dominant trees, site index and height to diameter ratio were incorporated into the equations by expanding the parameters of the growth function. Finally, a double-cross validation method was used to characterise the model error. Thus, the models were fitted *n* times, leaving out each tree once, so that the number of fittings was equal to the number of trees.

The base-age invariant parameter estimation technique used in the cork growth model (Sánchez-González et al. 2007c) was the Iterative Evaluation Approach (Tait et al. 1988; Cieszewski et al. 2000). This method used nested iterative procedures to fit the global and site-specific parameters. In order to address the occurrence of serial correlation resulting from repeated measurements taken from the same tree, which is a situation common to cork samples, the error term was modelled using a first-order autoregressive error structure (AR(1)). The Iterative Evaluation Approach including the AR(1) error structure was programmed using a SAS macro which uses the SAS/ETS procedure PROC MODEL (SAS Institute Inc. 2004), allowing for dynamic updating of the residuals. The evaluation of the models was based on numerical and graphical analyses of the residuals. Another important step in evaluating the models was to perform graphical analyses of the residuals, noting in particular the appearance of the fitted curves overlaid on the trajectories of the cork growth for each individual. Finally, a double-cross validation method was again used to characterise the model error.

2.3.2 Empirical and semi-empirical approaches

To model the annual diameter increment (Sánchez-González et al. 2006) two approaches were compared: the empirical and the semi-empirical models. In the empirical model, diameter increment was directly expressed as a function of tree and stand characteristics. This model was fitted by linear regression analysis using the PROC REG procedure of the SAS/STAT software (SAS Institute Inc. 2004b). Stepwise regression was used to select the tree and stand variables related to the variation in the diameter increment. The variables considered were the following: initial tree dimension, relative tree dimension, competition and site productivity.

In the semi-empirical model, diameter increment (*idu*) was expressed as a function that multiplies the potential diameter growth (idu_{po}) by a competitive adjustment factor (*mod*) (the modifier). The potential annual growth of each tree within the plots (idu_{po}) was calculated by dividing the potential growth by the time interval between two consecutive cork harvests and subtracting a correction factor to take into account the variability associated with different climatic conditions in the years when the plots were measured.

This correction factor was estimated by decomposing the deviation between the real and the expected potential annual diameter increment into four components according to the following model (Miina 1993, 2000):

$$E(idu_{po_j}) - idu_{po_{tj}} = \mu + u_t + u_j + \varepsilon_{tj} \tag{1}$$

where idu_{potj} is the real annual diameter increment (cm) of dominant tree j in year t; $E(idu_{potj})$ is the expectation of the annual diameter increment (cm) of dominant tree j in year t by using equation (1); μ is the intercept of the model; u_t is a random year effect u_t ~ $\mathrm{Nidu}_{po}(0, \sigma^2_{tr})$; u_j is a random tree effect u_j ~ $\mathrm{Nidu}_{po}(0, \sigma^2_{jr})$; and ε_{jt} is the random within tree error term ε_{jt}~ $\mathrm{Nidu}_{po}(0, \sigma^2_{\varepsilon})$. The variance components (σ^2_{tr}, σ^2_{jr}, σ^2_{ε}) and the random tree and year effects (u_t, u_j) were estimated using the maximum likelihood procedure of the computer software PROC MIXED in SAS/STAT (SAS Institute Inc. 2004b).

The competitive adjustment factor or modifier must be bounded between 0 and 1, so the following equations were tested as modifier functions:

$$mod = \left[1 - e^{-X}\right] \tag{2}$$

$$mod = \frac{1}{1 + e^{-X}} \tag{3}$$

where X is a linear function expressed as a combination of variables indicating initial dimension of the tree, competition, stand density and site productivity

Finally, the semi-empirical model was fitted with the observed diameter increment as a dependent variable, using ordinary nonlinear least squares regression associated with the PROC MODEL procedure of the SAS/ETS software (SAS Institute Inc. 2004a). The evaluation of the models derived from both the empirical and semi-empirical approaches, was based on the accuracy of the model predictions by calculating the bias and precision of the models. The models were also evaluated qualitatively by plotting the mean value and the standard error for the residuals as a function of the predicted and the independent variables. The double-cross validation method was used to characterise the model error. Since an independent validation data set was not available, the PRESS (Prediction Sum of Squares) residuals were used. These residuals were calculated by omitting each observation in turn from the data, fitting the model to the remaining observations.

2.3.3 Mixed model approach:

Finally, to develop the three sub-models: the height-diameter equation, the crown diameter prediction model (Sánchez-González et al. 2007a) and the cork thickness model (Sánchez-González et al. 2007b), the mixed-effect model approach was used. In the case of cork thickness, a calibration of the model from a small additional sample of observations was proposed as a practical approach for predicting cork thickness and weight.

When developing mixed models, the following assumptions are assumed: (1) that random effects are independently distributed among sampling units and are distributed following a multivariate normal distribution with mean zero and variance matrix ***Di*,** where ***Di*** is a variance-covariance matrix for random effects and (2), that the error terms are independent and distributed following a multivariate normal distribution with mean zero and variance $\boldsymbol{R_i}$, where $\boldsymbol{R_i(\beta, b_i, \rho)}$ is the variance-covariance matrix of the error terms and depends on both fixed and random effects as well as on a set of unknown parameters $\boldsymbol{\rho}$. The first assumption cannot be verified if spatial correlation exists among the sampling units, implying that the nearest observations are more similar than the rest. In the height-diameter equation and crown prediction model, it was assumed that the random parameters were independent, whilst several spatial structures were compared for the cork thickness model.

The approach used in modelling variance and correlation structures is basically the same for linear mixed-effects models as for non-linear ones. Details can be found in Lindstrom and Bates (1990), Pinheiro and Bates (1998) and Vonesh and Chinchilli (1997). The linear mixed-effects models were fitted using the restricted maximum likelihood method implemented in the PROC MIXED procedure of the SAS/ETS software (SAS Institute Inc. 2004a), while the SAS macro NLINMIX was used to fit the non-linear models. In those equations with more than one parameter, a determination was made as to which of the parameters in the model would be considered a mixed effect parameter and which would be considered as parameters of a purely fixed effect (Fang and Bailey 2001).

Nine generalised height-diameter equations were selected as candidate functions to model the height-diameter under cork relationship. All functions tested are non-linear and constrain the height-diameter relationship to pass through the point (1.30, 0) as well as through the point of dominant height-dominant diameter (H_0, D_0). The equations analysed for the crown diameter predictor model were: linear, parable, power, monomolecular and Hossfeld I. In both models, the evaluation was based on Akaike's information criterion, the Schwarz's bayesian information criterion, the –2 x logarithm of likelihood function and on numerical and graphical analyses of the residuals. Once the best crown diameter equation had been selected, several variables characterizing the stand were included in the mixed model as fixed effects (Hökkä 1997; Pinheiro and Bates 1998; Singer 1998). The double-cross validation method was used to characterise the error of both models using the PRESS residuals.

The description of cork thickness as a stochastic process involved two stages. In the first stage, a multilevel linear mixed model was fitted, in order to characterize the variability structure and remove the effects of the spatial autocorrelation. In the second stage, the explanatory covariates were identified by studying the correlation between random effects and possible explanatory covariates. In a mixed model approach, it is possible to calibrate the model by predicting the random component specific to a new tree, plot or cork harvest, using a complementary sample of cork thickness measured in that unit. Prediction of the random components was carried out using empirical best linear unbiased predictors (EBLUP). The accuracy of the calibration was evaluated using cork thickness measurements from the ten plots in the calibration data set, comparing various subsample size alternatives (1, 2, 4, 6, 8 and 10 trees randomly selected). For each plot and subsample size, 100 random realizations were performed, including different trees in the calibration subsample each time.

2.4 Submodels

2.4.1 Height growth of dominant trees

The dynamic equation derived by McDill and Amateis (1992) from the Hossfeld model was selected:

$$h_2 = \frac{20.7216}{1-\left(1-\frac{20.7216}{h_1}\right)\left(\frac{t_1}{t_2}\right)^{1.4486}} \tag{4}$$

where: h_i is the height (m) at age t_i (years).

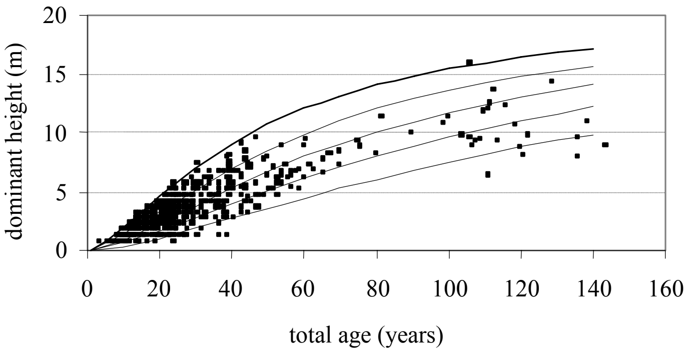

Figure 3. Height growth model for dominant trees in Spanish cork oak forests represented for the site quality classes defined. The dots represent the height-age pairs from the sample.

The analysis of regional differences indicates that a single height growth model could be used for both regions. Furthermore, after analyzing the variability of the mean value of absolute prediction errors by age class and site quality for different prediction intervals, it was concluded that the interval applied in the model should be less than 40 years.

This model was used to define a site index for cork oak stands as the dominant height reached at the age of 80 years. Five quality classes ranging from 14 m for Quality I to 6 m for Quality V were defined. The height model defined by equation (4) is represented graphically in Figure 3 for each site quality class. The age-height pairs from the sample are also shown on the graph.

In addition, this model allows us to estimate the minimum time that a regeneration block must be closed off to livestock. Assuming that this period of protection must last until young trees reach a height of 2 m, the number of years will range from 10 years for the best quality to 30 years for the worst.

2.4.2 Diameter growth of dominant trees

The dynamic equation derived from the Richards function considering only parameter a_3 as related to site productivity was selected to describe diameter growth of dominant trees:

$$d_2 = (83.20 + 5.28\ SI - 1.53\ h/d)^{1-\frac{ln(1-e^{-0.0063\ t_2})}{ln(1-e^{-0.0063\ t_1})}}\ d_1^{\frac{ln(1-e^{-0.0063\ t_2})}{ln(1-e^{-0.0063\ t_1})}} \tag{5}$$

where: d_i is the diameter at breast height under cork (cm) at age t_i (years); *SI* is the site index (m); *h/d* is height to diameter ratio (cm/cm).

As in the height growth model, the analysis of regional differences indicates that a single diameter growth model could be used for both regions. Site quality and density were included in the model, both of them in the asymptote parameter. Again, after analyzing the variability of the mean value of absolute prediction errors by age class, site quality and by height to diameter ratio class for four time prediction intervals, it was concluded that the interval applied in the model should be less than 40 years.

The diameter growth model defined by equation (5) is represented graphically in Figure 4 in terms of the different site index classes and mean values of height to diameter ratio for each site index class. The figure also displays the diameter-age pairs from the sample disks.

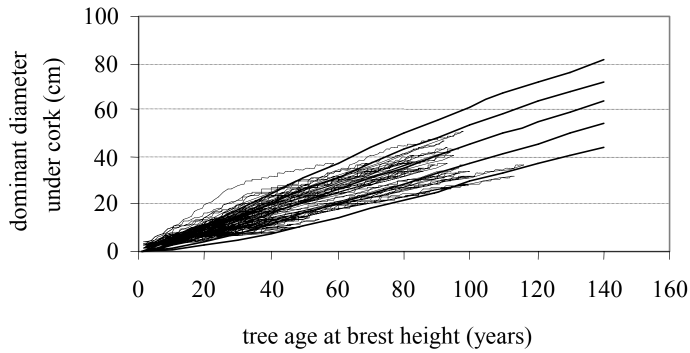

Figure 4. Diameter growth model for dominant trees in Spanish cork oak forests represented in terms of different site quality class and mean values of height to diameter ratio for each site quality class. The dotted lines represent the diameter growth curves of the sampled trees.

Taking 140 years to be the upper limit for the production of quality cork, the diameter values obtained range from 35.5 to 85.7 cm. In addition, this model allows us to estimate the minimum time required for a tree to reach a stage at which it can be debarked for the first time (attaining a virgin cork radial width of 2.7 cm). This would be 20 years under the best site conditions.

2.4.3 Annual diameter increment

In the development of the annual diameter increment model, empirical (6) and semi-empirical (7) approaches were compared. Both models included initial size, site productivity and stand density as predictive variables, the latter two being incorporated into potential annual growth in the semi-empirical model:

$$idu = 0.18 + 7.89\frac{1}{N} - 1.02\frac{1}{SI} + 2.45\frac{1}{du} \tag{6}$$

$$idu = idu_{po}\frac{1}{1+e^{-(0.73+94.97\frac{1}{N})}} \tag{7}$$

where *idu* is the annual diameter increment under cork (cm); *N* is the number of trees per hectare (trees ha^{-1}); *SI* is the site index (m); *du* is the diameter at breast height under cork (cm); idu_{po} is the annual potential diameter increment (cm).

The fact that density has been included in both models, whereas neither the relative tree dimension nor the competition variables (BAL) were identified as significantly related to diameter increment, indicates that competition in cork oak forests is size symmetric and that the competition for underground resources is more relevant than competition for light.

In order to illustrate the behaviour of the diameter models, the diameter increment vs. diameter at breast height has been plotted (Figure 5). Assuming a constant density and a site index of 10 m, diameter increment has been calculated from the empirical (eq. 6) and the semi-empirical models (eq. 7). Both models behaved similarly and are equally valid to describe diameter growth. The choice between them may be based on the availability of variables for the calculation of potential diameter growth.

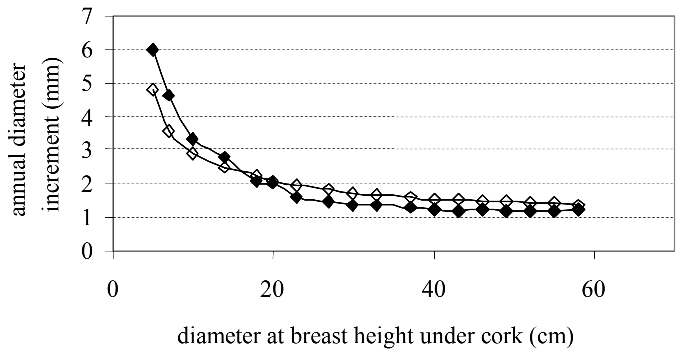

Figure 5. Annual diameter increment vs. diameter at breast height (du) in empirical (open diamond) and semi-empirical methods (solid diamond). Used predictor values: SI=10 m, N=400 trees ha^{-1}.

According to these models, adult cork oak growth ranges from 1.3 to 2 mm per year during the cork prediction period (for trees of sizes within the range of the data set used to fit the model). This result for diameter increment is similar to that reported in previous studies (Lamey 1893; Caro 1914; Montoya 1985; Montero 1987; Gourlay and Pereira 1998; Costa et al. 2002; Sánchez-González et al. 2005).

2.4.4 Height-diameter equation

The function that best describes the height-diameter relationship is the Stoffels and Van Soest function with the parameter varying randomly between plots and constrained to pass through the point of dominant height-dominant diameter.

$$h_{ij} = 1.3 + (H_0 - 1.3)\left(\frac{du}{D_0}\right)^{0.4898+u_i} + e_{ij} \qquad (8)$$

where h_{ij} is the total height of the *j*th tree in the *i*th plot (m); H_0 is the dominant height in the *i*th plot (m), *du* is the diameter at breast height under cork (cm), D_0 is the dominant diameter under cork in the *i*th plot (cm), u_i is the random effect associated with the *i*th plot, with mean zero and variance 0.064, and e_{ij} is the residual error term of the *j*th observation in the *i*th plot, with mean zero and variance 1.4447.

Figure 6 shows the behaviour of the selected height-diameter model (8) for five dominant heights corresponding to the site index classes defined for cork oak forests in Spain (Sánchez-González et al. 2005) assuming a dominant diameter of 40 cm for all of them.

This model gives reasonably precise tree height estimates and is recommended for cork oak forests within the following range of conditions: dominant height greater than 14 m and diameter values between 7.5 and 75 cm.

2.4.5 Crown diameter prediction model

The parable function without intercept and with the b parameter divided into a fixed part and random between-plot component was selected for predicting crown diameter.

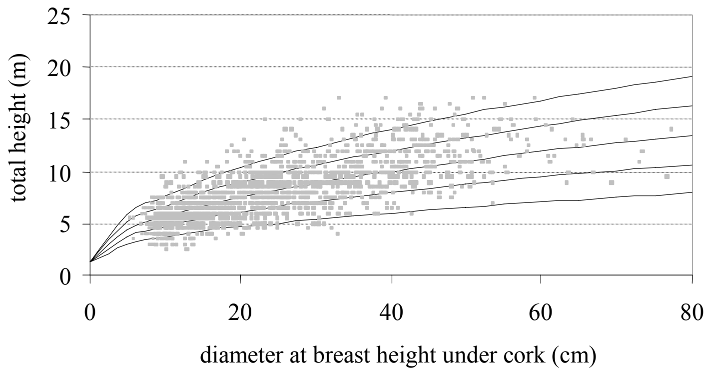

Figure 6. Height-diameter relationship (assuming a dominant diameter of 40 cm for each site quality) for five dominant heights corresponding to the site index classes defined for cork oak forests in Spain.

$$cw_{ij} = \left(0.2416 + 0.0013Dg + u_i\right)du - 0.0015du^2 + e_{ij} \tag{9}$$

where cw_{ij} is the crown diameter of the *j*th tree in the *i*th plot (m); *du* is the diameter at breast height under cork (cm), *Dg* is the quadratic mean diameter in the *i*th plot (cm), u_i is the random effect associated with the *i*th plot with mean zero and variance 0.0007 and e_{ij} is the residual error term of the *j*th observation in the *i*th plot with mean zero and variance 1.0577.

Figure 7 shows the evolution of crown diameter with diameter at breast height under cork for three quadratic mean diameter classes.

The crown diameter model provides reliable crown width estimations and is recommended within the following range of conditions: quadratic mean diameter values between 6 and 57 cm and diameter values between 7.5 and 80 cm. The model is sensitive to quadratic mean diameter variations, so it could be used to characterize cork oak forest structure, which in turn is used to simulate stand development.

2.4.6 Cork thickness and weight prediction models

None of the models which considered explanatory covariates were used because at best, the percentage of explained variability was less than 2%.

$$cb_{\mathrm{iijk}} = 25.8 + u_i + v_{ij} + w_{ik} + e_{ijk} \tag{10}$$

where u_i, v_{ij}, and w_{ik} are random components specific for each plot, tree and plot*cork harvest, realizations from univariate normal distributions with mean zero and variance 19.5, 5.7,and 3.9 respectively, e_{ijk} is a residual error term, with mean zero and variance 7.5.

The description of cork thickness as a stochastic process with random parameters at plot, tree, and plot*cork harvest levels, allows the model to be calibrated using cork thickness calibration measurements from four trees per plot.

Calibration allows us to estimate cork weight and facilities the classification process according to its final use in the cork industry by using Monte Carlo simulations to assign random components for each tree within the plot.

$$w \;=\; cb\;.\;sh\;.\;cu.\;cork\;density \tag{11}$$

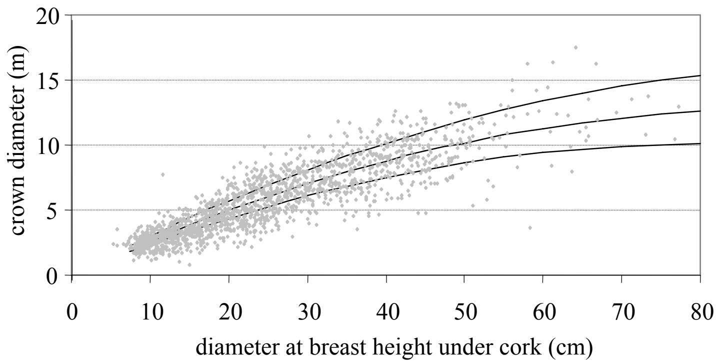

Figure 7. Crown diameter vs. diameter at breast height under cork for three quadratic mean diameter classes.

where *w* is cork weight (kg), *sh* is stripped height (m); *cu* is circumference at breast height under cork (m) calculated from diameter at breast height under cork; *cork density* and *cb* is predicted cork thickness.

2.4.7 Cork growth model

The GADA formulation derived by Krumland and Eng (2005) from the Richards model by considering a_1 and a_3 to be related to site productivity was selected for describing cork growth in Spanish cork oak forests.

$$cb_2 = cb_1 \left(\frac{1 - e^{-0.04t}}{1 - e^{-0.04t_0}} \right)^{\frac{0.57 + 1.86}{X_0}} \tag{12}$$

where:

$$X_0 = \frac{1}{2}\left(\left(ln(cb_1) - 0.57\, ln\left(1 - e^{-0.04t_0}\right) \right) \pm \sqrt{\left(ln(cb_1) - 0.57\, ln\left(1 - e^{-0.04t_0}\right) \right)^2 - 4 \cdot 1.86\, ln\left(1 - e^{-0.04t_0}\right)} \right)$$

cb_i is the cork thickness (mm) at age t_i (years).

Figure 8 shows the fitted curves for accumulated cork thickness in complete years of 15.75, 22.5, 29.25, 36 and 42.75 mm at 9 years for the selected model overlaid on the trajectories of observed values over time.

The cork growth model developed allows us to estimate the final accumulated growth in complete years from the second year after cork extraction with more than 80% reliability, or nearly 90% if the prediction is made in the second half of the rotation.

3. Final remarks

In this paper we have tried to identify the specific requirements that must be considered when modelling Mediterranean forests. We have also described how these might be taken into account by focusing on the case of the ALCORNOQUE 1.0 model.

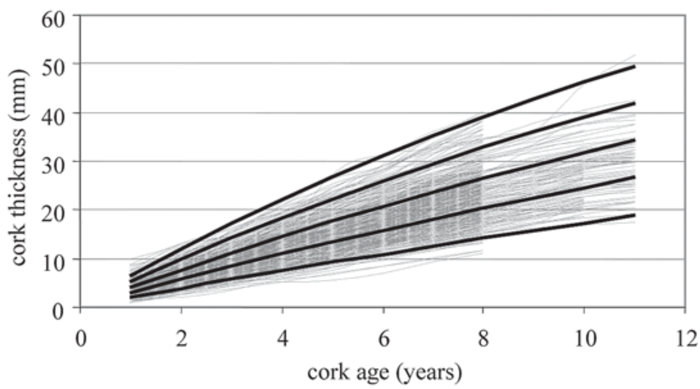

Figure 8. Cork growth curves for accumulated cork thickness in complete years of 15.75, 22.5, 29.25, 36 and 42.75 mm at 9 years for the selected model overlaid on the trajectories of observed values over time.

As part of this framework, some possible future lines of research might be:

- The inclusion of ecological factors as covariables in all the stand processes from regeneration to mortality.
- The development of new competition indices that better describe the symmetric competition of Mediterranean forests.
- The integration of growth and yield models with risk assessment models that take into account not only fires but also drought and pests.
- The development of multivariate models that include non-timber forest products and other forest services.
- The adaptation of statistical tools to describe the reality as precisely as possible.

In short, the models should be flexible, allowing links with different spatio-temporal scale models; from ecophisiology to landscape. These models should be robust enough to incorporate climate-change scenarios and serve as a useful tool for evaluating the sustainability of forest management schedules.

ALCORNOQUE 1.0 contributes significantly to improving our knowledge of cork oak growth in dense stands and is an important tool in terms of the valuable information provided for sustainable management of cork oak forests. However, ALCORNOQUE 1.0 is a preliminary version which requires further improvement in order to achieve its full potential. Subsequent versions will include a module for low density cork oak forests (dehesas), regeneration and mortality submodels and a new function that defines site index based on ecological factors.

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Modeling the Risk of Forest Fires in Catalonia (North-East Spain)

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Abstract

The inclusion of the risk of fires into forest management enables the manager to analyse the risk and uncertainty due to forest fires, and assess the expected losses that fires may cause on the forest outputs. On the other hand, forest management and planning offers an appropriate framework for identifying efficient measures for long term fire prevention. If fire risk is to be included in forest management planning, models for assessing the probability of fire occurrence and expected fire damage are required. As stands are regarded as the basic and indivisible forest management units, it is logical to develop stand-level models as the first step of including fire risk considerations into forest planning. These models must be based on such stand variables the future value of which is known with reasonable accuracy and that are under the control of the manager. In that way the manager will have the possibility to minimise the expected losses due to fire as a management objective in numerical planning calculations. Such models have been developed for Catalonia (North-East Spain) using data from the Spanish national forest inventory and perimeters of fires that occurred in Catalonia. The models showed that both the occurrence of fires and the potential damage caused by fires are related to stand characteristics such as species composition, tree size, stand structure, and to topogeographical variables such as elevation and slope.

Keywords: fire risk; risk assessment; fire occurrence model; damage model; tree survival model

1. Introduction

Fire can be considered as a natural element of the Mediterranean forest affecting the species composition and structure of the landscape (Trabaud 1994). Fire is the most important cause of tree mortality in the Mediterranean basin (Alexandrian et al. 2000), and a threat to private and public goods provided by Mediterranean forest. In Catalonia, forest fires are perceived by the public as the main environmental problem (Tábara 1996). During the last decades the problem of forest fires in Catalonia has taken a new dimension due to changes in the fire regime, larger and more destructive fires are more common (González and Pukkala 2007). One reason for the change in the fire regime is the lack of proper forest management due to abandonment of rural areas and the low profitability of traditional forest practices. This leads to the accumulation of forest fuels.

In this context, the inclusion of fire risk analysis in forest planning is clearly justified. Such analyses help to reduce the uncertainty by anticipating the outcomes of management alternatives in a systematic way, and identifying management options that reduce the expected losses due to fire (Gadow 2000). The inclusion of fire risk in forest management planning requires a set of models for assessing, in a quantitative way, the risk of fire attached to different forest types and management options. This information can be generated using statistical methods and historical data on forest fires and forest characteristics. Once such models are developed, they can be included in forest simulators. In this way, the risk of forest fires can be considered explicitly in forest planning problems.

The present paper is an overview of some of the most recent efforts taken within the Medforex project to model the risk of forest fires for forest planning purposes.

2. Modelling the risk of forest fires in Catalonia

The concept of *Risk* has been defined as the expected loss due a certain hazard for a given area and period of time (UN 1992). In forest planning, the principal components of the risk of fires are:

(i) the probability of fire occurrence in an area and period of time, and
(ii) the potential damage caused by fire once it happens.

If the risk of fire is to be included in forest planning, models for assessing the probability of fire occurrence and the potential damage caused by fire are required. These models should be able to predict the long-term consequences of management alternatives. Therefore, they must be based on predictors, which can be easily calculated with a reasonable accuracy, and which are under the control of the forest manager. As stands are regarded as the basic forest management unit it is reasonable to developed models for the stand level. Based on this rationale, the following models were developed:

1. A model that predicts the probability of fire occurring in a forest stand.
2. A model that predicts the damage caused by fire in a forest stand, once fire occurs in that stand.
3. A model to predict the survival probability of each individual tree in a burned stand.

The data used for modelling the occurrence and effect of forest fires consisted of the National Forest Inventory data and fire occurrence data (Figure 1). The 2nd Spanish National Forest Inventory for Catalonia with fire perimeters was used for the occurrence model while data from the 2nd and 3rd NFI were used for the damage and survival models. The Spanish National Forest Inventory consists on a systematic sample of permanent plots, distributed on a square grid of 1 km, with a re-measurement interval of 10-years (ICONA 1993; DGCN 2005). The fire data consisted of the perimeters of fires larger than 20 ha, determined on a 1:50 000 scale by the *Department de Medi Ambient I Habitatge* and the *Intitut Cartográfic de Catalunya*. The fire perimeters were used to determine which plots were affected by fire during the time period considered in the model.

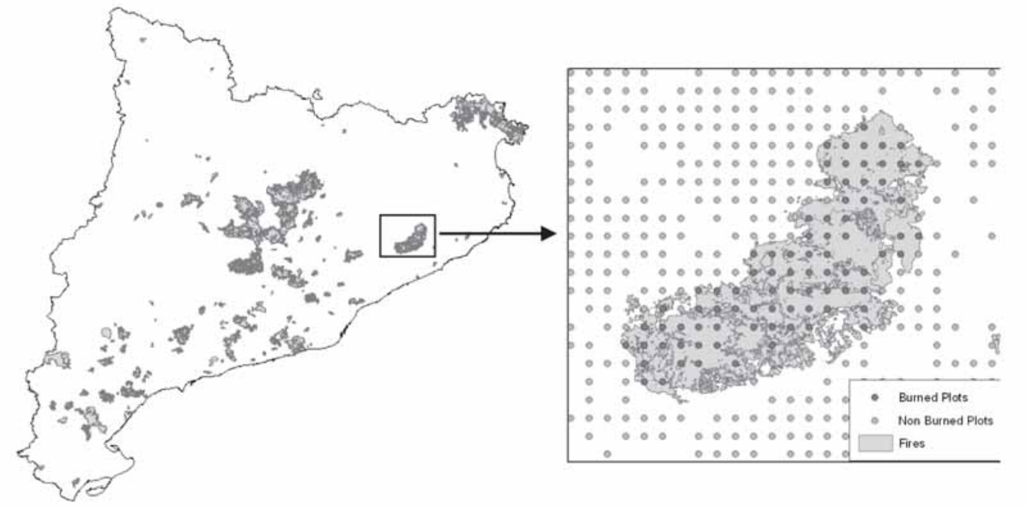

Figure 1. Distribution of forest fires that occurred in Catalonia during 1989-2002 and a part of the national forest inventory plots used in the study.

2.1 Modelling the probability of fire occurrence

The perimeters of the fires occurring in Catalonia 12 years after the 2nd measurement of the NFI plots were used to determine which NFI plots were affected by fire during that period and which plots were not. From a total of 10 855 plots used in the study, 770 were affected by one of the 231 fires (>20 ha) occurring in Catalonia during the period 1991–2002. With this information, binary logistic regression was applied to determine which stand characteristics had a significant effect on the probability of fire occurrence (González et al. 2006), and the following model was developed:

$$P_{fire} = \left(1 + e^{\left(-1.925 - 2.256\times ELE - 0.015\times Dg + 0.012\times G - 1.763\times P_{hard} + 2.081\times\left(\frac{SD}{Dg+0.01}\right)\right)}\right)^{-1} \quad (1)$$

where P_{fire} is the 12-year probability of fire occurring in a given stand, *ELE* is a transformation of the elevation, *Dg* is basal area weighted mean diameter of trees (cm), *G* is the stand basal area (m^2/ha), P_{hard} is the proportion of hardwood species present in the stand, and *SD* is the standard deviation of the diameters trees (cm). The last predictor *(SD/(Dg +0.01))* expresses the relative variability of tree diameters. The variable is close to 1 in rather uneven ages stands, and approaches to 0 in homogeneous stands. Variable *ELE* is equal to ln(max{1, *Elevation*-6)), where *Elevation* is in hundreds of meters above sea level. This transformation of elevation was used to express a trend in the modelling data which indicates that most fires occur at elevations lower than 700 metres.

The variables used as predictors in the model can be divided into variables dependent on forest management (*G*, *Dg*, P_{hard}, and the relation between *SD* and *Dg*) and variables related climatic conditions (*ELE*). The probability of fire occurrence is high for forests located between 0 and 700 metres a.s.l. Dense stands with an abundant conifer mixture and with high variation in

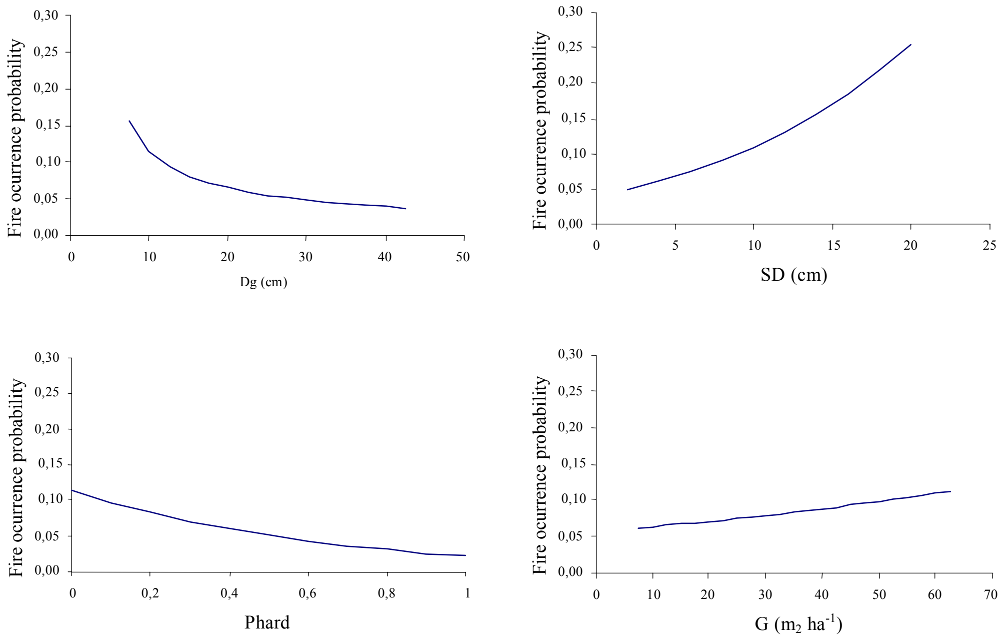

Figure 2. Effect of basal-area-weighted mean diameter (*Dg*), standard deviation of dbh (*SD*), proportion of hardwood of the number of trees (P_{hard}), and total basal area (*G*) on the probability of fire occurrence.

diameter tend to increase the probability of fire occurrence (Figure 2). On the contrary, the presence of hardwoods and the absence of small trees reduce the probability of fire.

2.2 Modelling stand damage and tree survival in fires

The effect of a forest fire on a forest stand, was estimated from the changes observed in the plots of the NFI plots affected by fire during the period between the 2nd and 3rd measurement (aprox 10-years). A total of 722 plots were affected by fire during that period. The use of permanent plots by the Spanish National Forest Inventory allowed us to identify which trees present in the 2nd NFI died before the 3rd NFI. This permitted us to model the damage caused by fire, and the probability of tree survival probability in forest fires as a functions of different stand and tree-level characteristics (González et al. 2007a).

The stand-level damage model had the following form:

$$y = b_0 + b_1 G + b_2 Slope + b_3 Pine + b_4 \left(\frac{G}{D_q + 0.01} \right) + b_5 \left(\frac{s_d}{D_q + 0.01} \right) + e \qquad (2)$$

where $y = \ln(P_{dead} / (1 - P_{dead}))$, P_{dead} is the proportion of dead trees in the stand (in terms of number of trees), *G* is the stand basal area (m^2ha^{-1}), *Slope* is the percentage of altitude change per distance change (%), *Pine* is a dummy variable which equals 1 if the stand is dominated by pines (> 50 % of basal area is pine) and 0 otherwise, s_d is the standard deviation of the breast height diameters of trees (cm), D_q is the quadratic mean diameter (cm) of trees, and *e*

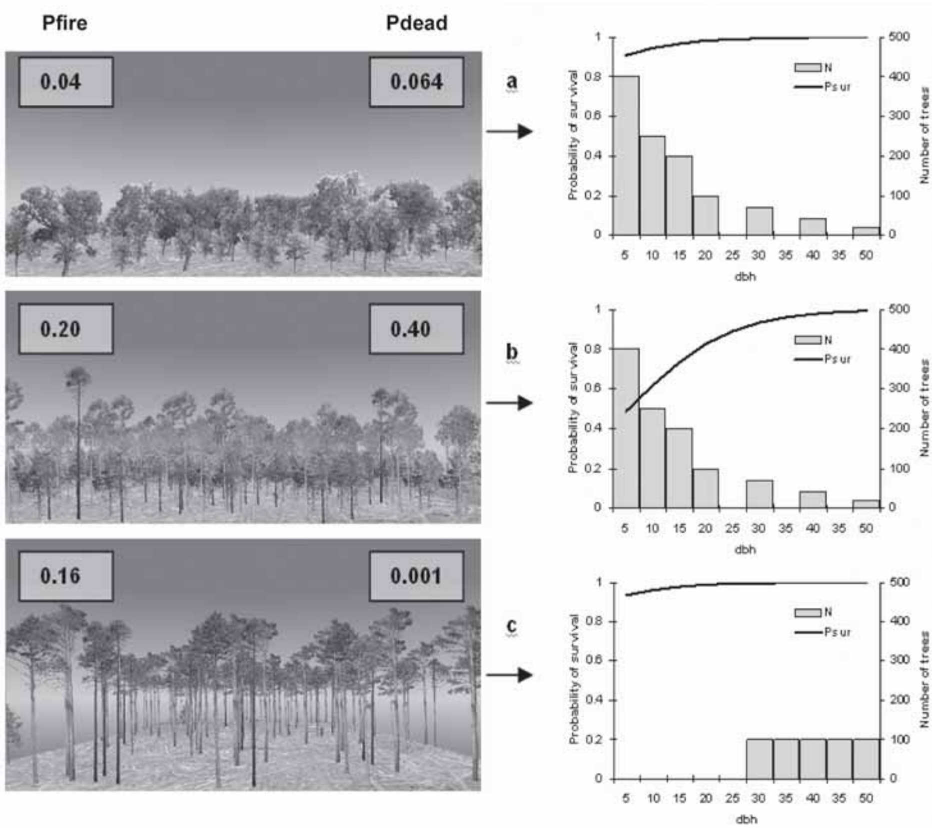

Figure 3. Fire risk depending on the stand structure and composition (altitude 700 meters and slope 12%). The images on the left represent different forest stands and their predicted probability of fire occurrence (Pfire, from Eq. 1) and damage in proportion of dead trees (Pdead, from Eq. 2). On the right, the diameter distribution (N, number of trees per diameter class) and the survival probability for the trees in each diameter class (Psur, from Eq. 3) are shown.

is the standard deviation of the residual (standard error). The predictor $G/(D_q+0.01)$ is non-linearly related to the number of trees per hectare. The last predictor $s_d/(D_q+0.01)$ expresses the relative variability of tree diameters. The variable is close to 1 in rather uneven stands and approaches to 0 in homogeneous stands.

Additionally, a model was developed using binary logistic regression to predict the probability of a single tree to survive a fire event, depending on the size of the tree and the level of damage caused by fire at stand-level.

$$P_{sur} = \left(1 + e^{-(b_0 + b_1 d + b_2 P_{dead})}\right)^{-1} \tag{3}$$

where P_{sur} is the probability of survival, d is the diameter of the tree at the breast height (cm), and P_{dead} is the proportion of dead trees.

From the models it can be concluded that once a fire takes place, stands on steep slopes, with an abundance of small trees and high variation in tree size are the most vulnerable to fire in terms of damage (proportion of dead trees). The smallest trees in these stands are the most susceptible to die (Figure 3b). These results are in accordance with previous studies in terms of indicating the most vulnerable forest structures (Pollet and Omi 2002; Agee and Skinner 2005) and trees (Ryan and Reinhardt 1988; Peterson and Arbaugh 1989; Beverly and Martell 2003; Hély et al. 2003; Rigolot 2004). The models suggest that conversion of uneven stands into even-aged forest structures and adequate thinnings are feasible silvicultural means to reduce the risk of fires (both fire occurrence and fire damage).

3. Conclusions and application of the models

In the Mediterranean region, the heaviest investments regarding fire management have been on fire-fighting equipment rather than prevention. However, during the last decades fire prevention has gained reputation as an efficient and cost-effective way to deal with forest fires. In this context, forest management planning can make a major contribution to reduce the long-term fire vulnerability of our woodlands, providing new tools for active and sustained fire prevention policies, which can be integrated into the forest management process, providing new opportunities for rural areas.

The presented fire occurrence, damage and survival models were developed based on variables measured on regular inventories. Some of the variables used in these models depend on forest management, meaning that the risk of fire can be modified through forest management. These characteristics make the models especially appropriate for forest management planning purposes, and allow their use in multiple studies and applications related with the inclusion of fire risk into management planning. These models, based on empirical data, are in line with the concepts and knowledge widely accepted by the experts on forest fires about forest characteristics that have an impact on this type of risk.

The models presented can be used to simulate the development of forest stands under the risk of forest fires, by generating stochastically fire occurrences and damages caused by them. Stochastic simulations have been found to be useful to solve stand level optimization problems were minimizing the risk of forest fires was an objective (González et al. 2005b; González et al. 2007c). These stand level optimizations can be used to define management guidelines for forest stands under the risk of fire. Another application for such stochastic simulations is their use in regional scenario analyses (González 2006). Scenario analyses can use a network of inventory plots, like the Spanish national forest inventory one, and a set of predefined management instructions, to predict the development of the forest resources in a certain region. Finally, the models can be used to calculate fire resistance indexes for alternative management schedules of stands. These indexes can be used to calculate various landscape metrics for fire resistance, which can be used as objective variables on numerical optimization. Landscape metrics help in reaching such landscape compositions and configurations that are resistant to forest fires (González et al. 2005a).

Variables not included in the models, like the abundance of ground vegetation and dead fuels, play a mayor role in defining the behaviour and the severity of forest fires (Rothermel 1983; Finney 1999). These variables affect the vulnerability of a forest stand to fire, and can be controlled through forest management. However, the inclusion of such variables requires information which is difficult to obtain or predict over long periods of time and across heterogeneous landscapes (He and Mladenoff 1999). Attempts were made to include ground vegetation as a predictor in the models but the results were not encouraging. Other approaches, such as modelling the knowledge of experts in fire ecology and fire fighting, proved to be an interesting an viable way to analyze the effect that ground vegetation have on the vulnerability of a forest stand to fire (González et al. 2007b).

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Integrating Landscape Connectivity in Broad-Scale Forest Planning: A Methodology Based on Graph Structures and Habitat Availability Indices

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Abstract

Maintaining landscape connectivity is a key strategy for the conservation of the biodiversity and the ecological functions of forests, which has led to an increasing interest in integrating connectivity in forest planning and management. We here propose a practical methodology based on graph structures and habitat availability indices that overcomes several limitations of other existing approaches. Graph structures may be used for quantitatively describing a forest landscape as a set of functionally interconnected patches, and are a powerful and effective way of overcoming computational limitations that appear when dealing with large data sets and performing complex analysis regarding forests connectivity. On the other hand, it has been shown that the connectivity problem should be considered within the wider concept of habitat availability in order to be useful for forest planning and conservation. We present the new Conefor Sensinode 2.2 software as an analysis support tool for identifying the forest habitat patches that most contribute to the maintenance of landscape connectivity. We apply this methodology to the analysis of treecreeper (*Certhia familiaris)* forest habitat in Catalonia (NE Spain) through the new probability of connectivity index, allowing us to identify the key public forests in which a specific management oriented towards the conservation of the habitat of this species is required.

Keywords: Forest connectivity; graphs; habitat availability; forest planning; ecological networks

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1. Introduction

Maintaining landscape connectivity is currently considered a key strategy for the conservation of the biodiversity and the ecological functions of forests (Forman 1995; Rochelle et al. 1999; Crist et al. 2005; Crooks and Sanjayan 2006). The dispersal and viability of many forest-dwelling species may be seriously limited in the fragmented landscapes of the Mediterranean and other European regions, which has led to an increasing interest in considering connectivity in forest planning and management. For instance, in the resolution 4 ("Conserving and enhancing forest biological diversity in Europe") of the Fourth Ministerial Conference on the Protection of Forests in Europe held at Vienna in 2003, the Signatory States and the European Community commit themselves to "prevent and mitigate losses of forest biological diversity due to fragmentation and conversion to other land uses and maintain and establish ecological connectivity".

However, although many different indices have been proposed and used in this context, there is a lack of comprehensive understanding of their sensitivity to pattern structure and their behaviour to different spatial changes, which seriously limits their proper interpretation and usefulness. Recent comparative analyses (Pascual-Hortal and Saura 2006a; Saura and Pascual-Hortal 2007) have shown the weaknesses of different commonly used connectivity indices for prioritizing the most important areas for the maintenance of landscape connectivity. Most of the examined indices did not fulfilled all the desirable properties for decision-making, with the only exceptions of two new connectivity indices, the integral index of connectivity (Pascual-Hortal and Saura 2006a) and the probability of connectivity (Saura and Pascual-Hortal 2007). The first index was developed according to the binary dispersal model (in which two forest habitat patches are just either connected or not connected, with no intermediate modulation of the strength or feasibility of the connection between them), while the latter has been developed based in the richer probabilistic dispersal model, in which there is a certain probability of direct dispersal between two forest patches. The probability of connectivity is in general preferable to the integral index of connectivity, since the more detailed dispersal model allows it to attain some relevant improved properties for forest landscape analysis and planning (Saura and Pascual-Hortal 2007). Additionally, a recent study (Pascual-Hortal and Saura 2007) has evaluated how the planning decisions based on connectivity indices (e.g. through the identification of most critical forest areas for landscape connectivity) are affected by the scale of the analyzed maps. The integral index of connectivity and the probability of connectivity were found to be two of the most robust and suitable indices for comparison across scales. Such robust indices enable planners to use them for undertaking connectivity analysis with the confidence that the obtained results and subsequent planning decisions are not largely biased by the particular spatial scale of the analysed dataset (Pascual-Hortal and Saura 2007).

Here we intend to describe (1) the key concepts (graphs and habitat availability) underlying the new indices and the methodology for the integration of connectivity in forest planning (Figure 1) (2) the new probability of connectivity index as an improvement to other existing indices in this context (3) the new Conefor Sensinode 2.2 software as a decision-support tool in which the new developments have been implemented (Figure 2), and (4) an example of application for the treecreeper (*Certhia familiaris)* forest habitat in Catalonia (NE Spain) to illustrate the use and potential of the methodology for broad-scale forest planning purposes.

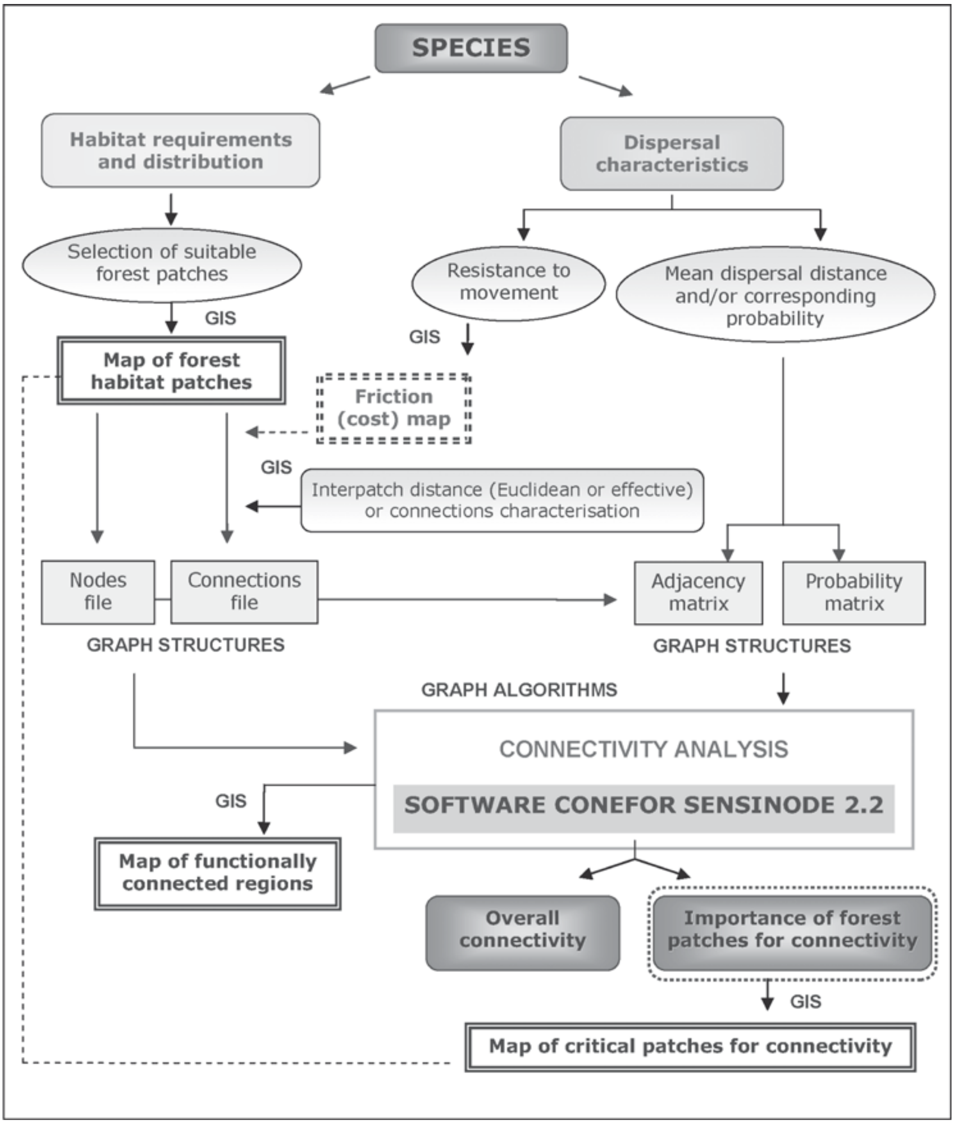

Figure 1. Schematic outline of the methodology for the analysis of forest landscape connectivity through the Conefor Sensinode 2.2 software.

2. The forest landscape as a graph

A graph represents a forest landscape as a set of nodes and links such that each link connects two nodes (e.g. Urban and Keitt 2001; Pascual-Hortal and Saura 2006a). Nodes represent the spatial units that are considered for the connectivity analysis. They will typically be forest habitat patches or forest cells, but they may as well correspond to any other forest unit discriminated in the landscape based on ecological, administrative or management criteria (e.g. ownerships, public forests, forest blocks, natural parks, etc.), depending on the scale and objectives of the analysis. Links represent the functional connection between a pair of nodes; the existence of a

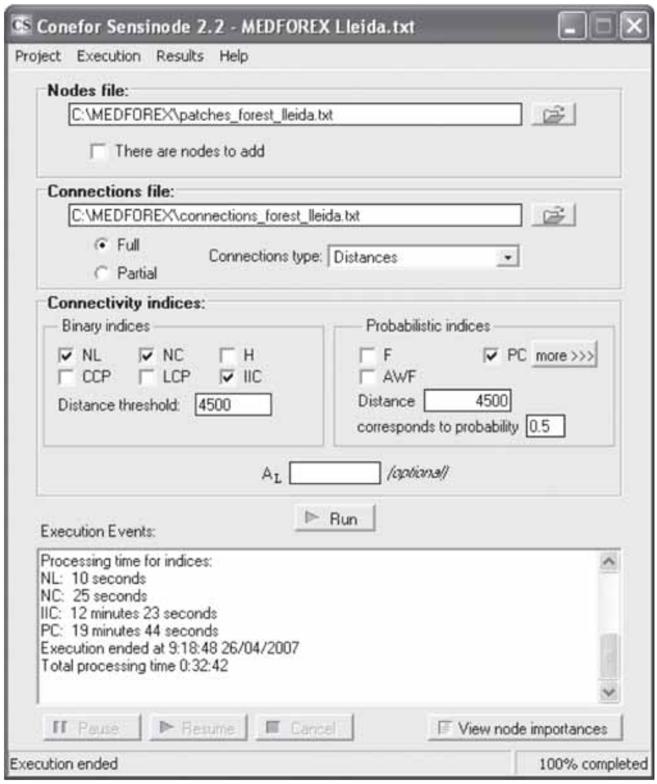

Figure 2. Main screen of the Conefor Sensinode 2.2 software developed by J. Torné and S. Saura at the University of Lleida.

link implies the potential ability of an organism to directly disperse between these two nodes. Links may be characterised by a probability of dispersal (in the probabilistic dispersal model), which is typically obtained as a function of distance. Distances between nodes (patches) can be obtained as Euclidean distances or, preferably, as minimum cost distances that take into account the variable movement preferences and abilities of the animal species through different land cover types (Adriaensen et al. 2003; Chardon et al. 2003; Nikolakiki 2004; Stevens et al. 2004; Theobald, 2006). We will here assume that the connections (links) are symmetric (undirected graphs). A path is a route along adjacent nodes (nodes connected by links) in which no node is visited more than once. A component (connected region) is a set of nodes for which a path exists between every pair of nodes. An isolated patch makes up a component itself. There is no functional relation (no path) between patches belonging to different components. A graph component disconnects when a part of it, after a change in the landscape, becomes not reachable from some other part, causing an increase in the number of components in the landscape.

The applications of graph theory to the analysis of landscape connectivity have increased rapidly in recent years (Ricotta et al. 2000; Bunn et al. 2000; Urban and Keitt 2001; Jordan et al. 2003; Brooks 2006; Pascual-Hortal and Saura 2006a, 2006b, 2007; Theobald 2006; Bodin and Norberg 2007; Saura and Pascual-Hortal 2007). Graph structures may be indeed used for quantitatively describing a forest landscape as a set of spatially or functionally interconnected patches, and are a powerful and effective way of overcoming computational limitations that appear when dealing with large data sets and performing complex analysis regarding forests connectivity. This is particularly important in a context where the spatial scales traditionally considered in forest resources management should be broaden to adequately characterize connectivity, which occurs at the landscape level. The wider geographical perspective (landscape level analysis of forests) significantly increases the computational complexity of the analysis by enlarging the amount of spatial data to be analysed, with the spatial relationships between the different landscape elements becoming more intricate.

3. Forest habitat availability and topoecological indices

It has been noted that a connectivity-related index that is not sensitive to habitat abundance or habitat loss will for sure perform badly in the prioritization of habitat patches for conservation (Pascual-Hortal and Saura 2006a). Forest habitat loss will worsen the conservation status of that forest and would never be considered a positive change for forest-dwelling species, with the relatively rare exception of those sink patches that may be actually having a negative effect for population conservation (Delibes et al. 2001). However, many connectivity indices do consider certain types of forest habitat loss as ‘positive’ or at least non-negative (even when they do not affect to any sink habitat), and identify for example as ‘more connected’ a landscape with three interconnected 10-ha forest patches (with links existing between all these patches) than a landscape with two unconnected 1000-ha forest patches, even when the former three patches may be the result of a large fragmentation and forest area loss process in just one of the big isolated patches, and when one of the isolated big patches alone provides much more connected forest area than all the small patches together (even if they are strongly interconnected). It has been shown that a forest patch itself should be considered as a space where connectivity occurs; otherwise, it will surely perform poorly as a guideline for conservation, and its use in this context should be avoided (Pascual-Hortal and Saura 2006a). In some cases the connected area existing within the forest patches themselves may be much larger than the one existing due to the connections between forest patches. This is the key of the ‘habitat availability’ concept, characteristic of the landscape that integrates both habitat abundance and habitat connectivity (Pascual-Hortal and Saura 2006a). For a forest being easily available for an animal or population, it should be both abundant and well connected. Therefore, habitat availability for a species may be low if forest habitat patches are poorly connected, but also if the forest is very connected but highly scarce. The new developed integral index of connectivity and probability of connectivity are habitat availability indices in this sense (Pascual-Hortal and Saura 2006a, Saura and Pascual-Hortal 2007). A successful integration of connectivity considerations in the forest planning decision-making should be performed through this habitat availability concept and type of indices.

On the other hand, many connectivity indices are only based on topological properties and do not allow in principle any difference in patch area or quality to be included in the calculation (Ricotta et al. 2000). Those indices that only consider topological properties do not take into account essential forest characteristics that should be considered in the decision-making process (Pascual-Hortal and Saura 2006a). On the contrary, the new developed integral index of connectivity and probability of connectivity include in their expressions a descriptive variable for patches that allows incorporating in the analysis differences in the area, habitat quality, or any other relevant attribute of the forest patches in the landscape. This improves the ecological realism of the analysis, and meets the need for the development of new ‘topoecological’ indices that introduce qualitative differences among distinctive patches in the calculation of topological indices (Ricotta et al. 2000).

4. The probability of connectivity index

The probability of connectivity index (PC) (Saura and Pascual-Hortal 2007), ranges from 0 to 1 and increases with higher connectivity. PC is based on dispersal probabilities (p_{ij} , probability of a direct dispersal (step) between nodes *i* and *j*), therefore improving the characteristics of the recently proposed integral index of connectivity, which is based on the binary connections model (Pascual-Hortal and Saura 2006a). PC is defined as the probability that two animals randomly placed within the landscape fall into habitat areas that are reachable from each other (interconnected) given a set of *n* habitat nodes (e.g. forest patches) and the connections (p_{ij}) among them. It is given by the following expression (Saura and Pascual-Hortal 2007):

$$PC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i \cdot a_j \cdot p_{ij}^{*}}{A_L^{\,2}} \tag{1}$$

where a_i and a_j are the areas of the nodes (e.g. forest patches) *i* and *j*, and A_L is the total landscape area (area of the study region, comprising both forest and non forest patches). These variables can also refer to other attributes different from patch area such as habitat quality or some other node attributes that may be considered relevant for the analysis (area-weighted quality, habitat suitability, core area, etc.). The product probability of a path (where a path is made up of a set of steps in which no node is visited more than once) is the product of all the p_{ij} belonging to each step in that path. p^*_{ij} is defined as the maximum product probability of all possible paths between patches *i* and *j* (including single-step paths). If nodes *i* and *j* are close enough, the maximum probability path will simply be the step (direct movement) between nodes *i* and *j* ($p^*_{ij} = p_{ij}$). If nodes *i* and *j* are more distant, the 'best' (maximum probability) path would probably comprise several steps through intermediate stepping stone nodes yielding $p^*_{ij} > p_{ij}$. When two nodes are completely isolated from each other, either by being too distant or by the existence of a land cover impeding the movement between both nodes (e.g. a road), then $p^*_{ij}=0$. When $i=j$ then $p^*_{ij}=1$ (it is sure that a patch can be reached from itself); this relates to the habitat availability concept that applies for PC, in which a patch itself is considered as a space where connectivity exists.

The PC index has been shown to present the following relevant desirable properties for integrating connectivity in forest landscape planning (Saura and Pascual-Hortal 2007), improving the characteristics and performance of other existing indices in this respect:

- It has a predefined and bounded range of variation (independent of the particular analyzed landscape) from 0 to 1.
- It can be computed both on vector and raster data.
- It attains its maximum value when a single forest patch covers the whole landscape and indicates lower connectivity as the forest gets more fragmented and the distance between forest patches increases.
- It always considers negative the loss of an entire forest patch or a part of a forest patch, no matter if the patch is isolated or connected to some other patches.
- It detects as more important the loss of bigger patches and the loss of key stepping-stone patches, identifying as less critical those key stepping-stones patches that when lost leave most of the remaining habitat area still connected.
- It is unaffected by the presence of adjacent forest patches in the landscape, as may be discriminated from each other by having different habitat quality, or as imposed by ownership, administrative or management limits (e.g. protected areas boundaries, forest management blocks or compartments, etc.).

Two different types of outcomes are possible when analyzing present landscape connectivity through an index such as PC. On one hand, a single index value may characterize the degree of connectivity of the whole landscape; this provides an idea of the current status of the landscape, but is simply descriptive and not particularly relevant for specific landscape planning purposes. On the other hand, an operational connectivity analysis would pursue identifying the most critical landscape elements for the maintenance of overall connectivity (Keitt et al. 1997; Jordán et al. 2003; Pascual-Hortal and Saura 2006a). Most critical landscape elements (typically forest patches) would be those whose absence would cause a larger decrease in overall landscape connectivity. The ranking of landscape elements by their contribution to overall landscape connectivity according to the PC index can be obtained by calculating the percentage of importance (*dPC*) of each individual element (Keitt et al. 1997; Urban and Keitt 2001; Pascual-Hortal and Saura 2006a; Rae et al. 2006):

$$dPC(\%) = 100 \cdot \frac{PC - PC^{'}}{PC} \tag{2}$$

where PC is the index value when the landscape element is present in the landscape and PC’ is the index value after removal of that landscape element (e.g. after a certain forest patch loss). The dPC values are very useful for decision-making in forest landscape planning, since they allow identifying the most critical nodes (e.g. forest patches) for the maintenance or improvement of landscape connectivity, in which a forest management oriented to the conservation of the forest habitat should be implemented.

5. The new Conefor Sensinode 2.2 software

Conefor Sensinode 2.2 (CS22) is a new program for evaluating the importance of forest patches for maintaining landscape connectivity based on graph structures and habitat availability indices (Figures 1 and 2), including the probability of connectivity index described above. CS22 is conceived as a tool for decision-making support in forest landscape planning and habitat conservation. It approaches connectivity from a functional perspective. That is, not only the spatial arrangement of the habitat is taken into account (structural connectivity) but also the dispersal distances and behavioural response of individuals or species to the physical structure of the landscape (functional connectivity) (Theobald 2006) (Figure 1). Some simple data preprocessing within GIS is needed (Figure 1) to provide the input information as required by CS22. The outputs provided by CS22 (Figure 1) can be saved in standard text and DBF files that can be easily imported in any GIS, spreadsheet or other software for further analysis. CS22 also includes the possibility of evaluating the connectivity improvement provided by new potential forest patches that may be added in the landscape, in addition to the quantification of the importance for connectivity of those forest patches that already exist in the landscape.

CS22 has been developed by Josep Torné and Santiago Saura in the University of Lleida by modifying, reprogramming in C++ and including new indices and features in the Fortran source codes developed by Dean Urban (Duke University) in the LandGraphs package (Sensinode 1.0). CS22 only requires a standard computer running a Windows operative system (Figure 2). However, when dealing with very large applications, the amount of RAM memory (GB) will limit the maximum number of nodes that can be processed in a single run of CS22 (usually about 10000 to 15000 nodes for standard computers with 1 or 2 GB RAM). Additionally, for the most computationally intensive indices (such as the probability of

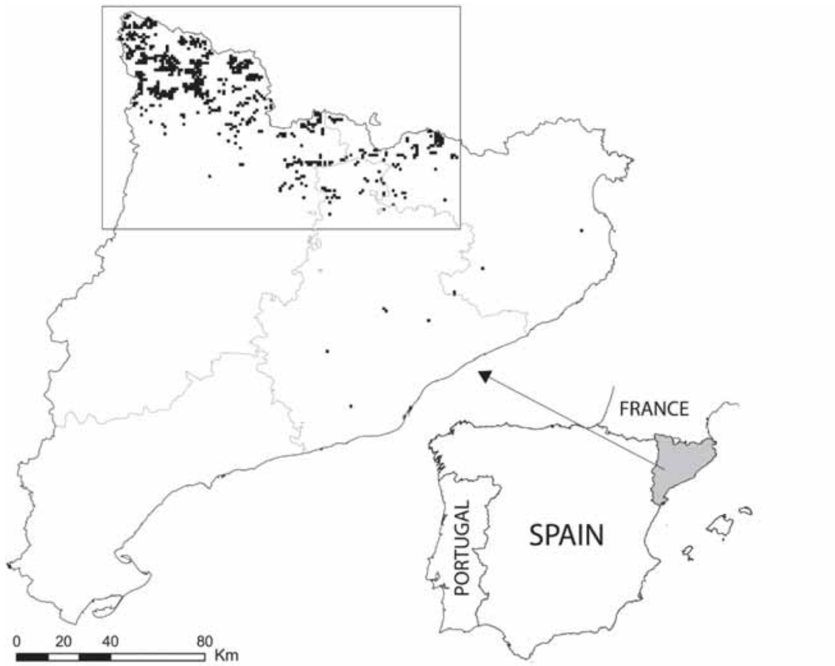

Figure 3. Location of the study area, Catalonia, in the map of Spain. Black cells correspond to treecreeper distribution (UTM 1×1 km cells in the Catalan Breeding Bird Atlas 1999–2002).

connectivity), the computer processing speed (GHz) of the CPU will determine how fast the node importance values can be calculated and, depending on the total number of nodes, if the full processing can be accomplished in a reasonable time. In standard computers, the analyses through the probability of connectivity index are generally limited to landscapes with a maximum of 2000 or 3000 nodes. A free copy of Conefor Sensinode 2.2 and further details on this software can be obtained from http://www.udl.es/usuaris/saura/cs22.htm or from http://www.conefor.udl.es/cs22.htm.

6. Application to a case study

To illustrate the use and effectiveness of the methodology and the probability of connectivity index for forest landscape planning, we analysed the connectivity of the forest habitat of the Eurasian treecreeper (*Certhia familiaris)* in the region of Catalonia (Pascual-Hortal and Saura 2006c), located in the Northeast of Spain (Figure 3). Catalonia is a heterogeneous region comprising the provinces of Barcelona, Girona, Lleida and Tarragona and with a total extension of 32,107 km^2, including mountainous areas like the Pyrenees (with an altitude up to 3,143 m) and a long coastline along the Mediterranean Sea. The climate is, according to Papadakis classification, mostly Mediterranean temperate, with presence also of maritime temperate climate in the coast and temperate cold climate in the Pyrenees. According to the Third Spanish National Forest Inventory, about half of the total area of Catalonia are forests

Table 1. List of the most important public-managed Catalan forests in relation to their contribution to overall forest habitat connectivity for the treecreeper. The accumulated connectivity importance value (dPC) in each forest is obtained as the sum of the individual importances (dPC, Equation 2) of the forest patches (cells orr portions of them) falling within each public forest. Only the ten public forests with the highest accumulated dPC are listed. Forest total area includes both habitat and non-habitat areas within that forest.

Forest ID	Name of forest and municipality	Forest total area (ha)	Accumulated connectivity importance dPC (%)
L-321	Muntanya (Espot)	7480.4	21.9
L-323	Riberes de Sant Nicolau (La Vall de Boí)	269.9	20.3
L-257	Valarties (Naut Aran)	6316.6	18.1
L-297	Bandolèrs, Dossau, Beret, Ruda e Aiguamòg (Naut Aran)	8139.5	15.3
L-244	Fenerui, Beciberri i Xelada (Vilaller)	3303.5	11.5
L-282	Artó Còsta (Vielha e Mijaran)	1009.2	11.2
L-283	Pales de Sòto (Vielha e Mijaran)	625.5	10.8
L-183	Port de la Bonaigua (Alt Àneu)	2391.0	9.3
L-281	Ribèra deth Toran (Canejan)	4108.4	7.3
L-301	Sendrosa (Naut Aran)	203.7	7.0

with a canopy cover above 20%, with *Pinus halepensis*, *Pinus sylvestris*, *Quercus ilex* and *Pinus nigra* as the most abundant forest tree species. The Eurasian treecreeper (*Certhia familiaris*) is a sedentary bird typically dwelling in oldgrowth forests that can be considered an umbrella species. It is suggested to suffer from certain types of forest management, because it is absent from clear-cuts and young regrowth stands and its breeding densities are three times higher in oldgrowth forests than in intensively managed forests (Haila et al. 1989). Its preferred habitat in the Catalan Pyrenees correspond to beech (*Fagus sylvatica*), silver fir (*Abies alba*) and mountain pine (*Pinus uncinata*) forests. The main threats facing this species, which is classified as *Near threatened* (NT) following the IUCN criteria (2001), are the increase in forest fragmentation and alteration of forest structure by the exploitation of mature forests, the development of new ski runs and the proliferation of recreational forest trails, which decreases the habitat suitability and availability for the treecreeper.

Forest habitat distribution data were obtained from the Catalan Breeding Bird Atlas (Estrada et al. 2004), which provides the estimated probability of occurrence of the treecreeper in 1×1 km UTM cells covering all Catalonia, as a result of field sampling and niche-based modeling (Estrada et al. 2004). All cells with a probability of occurrence greater than 0.1 were selected in this study as forest habitat patches to be analyzed (Figure 3). The probability of occurrence in each cell was considered as a measure of habitat quality and as the relevant patch attribute for the analysis (a_i variable in PC index, see Equation 1), indicating that patches with higher probability of occurrence are more suitable for treecreepers. To include the species dispersal ability in the analysis we considered an average dispersal distance of 2.87 km, as derived from the application of the empirical dispersal models by Sutherland et al. (2000) for the specific treecreeper body mass (Aho et al. 1999) and diet type. We set to 0.5 the dispersal probability (p_{ij}) associated to that distance of 2.87 km, and calculated all the remaining interpatch p_{ij} by applying a negative exponential function (Urban and Keitt 2001) of the edge-to-edge distance between forest patches (1 x 1 km forest cells).

The application of the probability of connectivity index (PC) and the Conefor Sensinode 2.2 software (Figure 1) to the analysis of forest patches importances (1 x 1 km cells) allows identifying and prioritizing the forest sites and public forests that most contribute to overall landscape connectivity for the treecreeper (Table 1), as evaluated by dPC (Equation 2). This outcome is particularly useful for forest landscape planning, as it allows concentrating conservation efforts and adapting forest management to the species requirements in those areas that are most important for connectivity maintenance, in which an eventual habitat loss would have more critical impact on the remnant habitat network and on the habitat availability for this species. A few public forests concentrated remarkably high values of accumulated connectivity importance, quantified as the sum of the individual importances (*dPC*) of each 1×1 km cell (or portions of it) falling within each public forest (Table 1). These priority forests were located mostly in the municipalities of Espot, la Vall de Boí, and Naut Aran (Table 1). In these most important forests, a management specifically oriented to the conservation of the treecreeper habitat should be applied. Specific forest management recommendations for preserving an adequate habitat for this passerine species concerns conserving mature forests, preferably over 100 years old (Haila et al. 1989), avoiding clear-cuts during the breeding season (spring), minimizing edge length with the agricultural matrix and preserving large tree trunks where the treecreeper can feed on invertebrates (spiders are its main food supply). A mean stem circumference of about 60 cm is recommended at core areas and, if possible, also at their home ranges, which may be as large as 12 ha during the rearing of broods (Suorsa et al. 2005). Finally, the maintenance of relatively high timber volumes is desirable, since it has been shown that stands below 150 m^3/ha present considerably lower treecreeper occupancy rates (Suorsa et al. 2005).

Other applications of the methodology have been reported elsewhere, including the analysis of forest connectivity for the capercailllie through the integral index of connectivity (Pascual-Hortal and Saura 2006b) and for the goshawk through the probability of connectivity index (Saura and Pascual-Hortal 2007).

7. Conclusions

The proposed methodology (graph structures, habitat availability concept, probability of connectivity index, and Conefor Sensinode 2.2 software) is a helpful decision support tool for incorporating connectivity into broad-scale forest planning, as it allows identifying which forest patches are more relevant for the maintenance of overall landscape connectivity. We have shown the potential of the methodology in an example of application to a case study for the treecreeper in Catalonia (NE Spain), where the most critical public forests for landscape-scale connectivity have been identified. In general, this may provide valuable guidelines for orienting forest management and focusing conservation efforts directly on those forest areas that are critical due to their attributes and specific network location within the landscape mosaic.

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Mediterranean Forest Values

Towards a Supportive Policy Framework for Mediterranean Forest Lands

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Abstract

The objective of this paper is to analyse the social frame changes which have taken place over the past few decades in the Mediterranean European countries as they move from primary to tertiary societies and the consequences for forested areas become evident. The potential for abandonment and the widespread incidence of forest fires are currently the main issues to be addressed. A new policy framework for Mediterranean forests is proposed and specific innovations recommended. These innovations are based on the legal structure, economic and institutional governance, non wood production, environmental and recreational services and estate value optimisation, assuming that the higher the value of Mediterranean forests, the better the means will be to protect them from threats like large-scale forest fires. Recent developments in environmental policies such as the climate change debate, green taxation or the water directive reform are identified as opportunities to strengthen the framework supporting Mediterranean forest lands.

Keywords: Mediterranean forestry; internalisation of positive externalities; economic environmental instruments; payment for environmental services

1. An introduction to the social frame changes of Mediterranean forests and consequences for the future of Mediterranean forests[1]

Three distinct socioeconomic phases characterize modern societies and, thus, the social framework of forests: primary, secondary (industrial) and tertiary (services). Primary or agrarian societies were maintained in the Mediterranean countries of the European Union (EU) longer than in other industrialized nations given their later industrialization. This later

1 See Giddens (2001).

industrialization was characterized by an accelerated and imperfect process that was soon overtaken by the globalisation of markets and a rapid shift towards tartarisation which has been driven since the late 1960s mostly by tourism.

The primary phase lasted for thousands of years and up to just a few decades ago it marked the social imaginary as well as the legal framework kept by a strong inertia that could be described as sweet and irreversible decline. Civil Codes with a pronounced Roman law heritage are a good example of this, especially considering the attention given to rural issues from a primary perspective. The secondary or industrial period was, by contrast, so imperfect and limited in time that it had little influence on the long-term elements of the socio economic framework. The recent trend towards an economic dominance of the tertiary sector has been the result of social changes without any proactively designed framework which, consequently, means it is far from being either optimised or regulated (anarchic).

Despite the etymological root of the term (*foris*), forests are closely linked to society, especially in areas that have been intensively settled for such a long period of time. The predominating social framework as well of present times as of recent times, exerted a greater influence on the forests than generally acknowledged.

In primary societies, the role of forests was limited to being a product supplier for local uses and a land reservoir for other rural land uses (marginal agriculture and pasture land generally). Forests risked being substituted or overused given the substantial pressure and the low specialization rates applied to them. On the other hand, markets reflected the true costs and the multi-product orientation of agriculture and forestry minimising the risk of fires as well as of losing significant biodiversity if human pressure did not surpass a specific threshold.[2] Finally, at the end of the primary phase, forestry was (mis)used in many Mediterranean countries by authoritarian regimes as a labour sink to compensate rural unemployment (Ortuño and Ceballos 1977; Mendes 1999 and Colpi et al. 1999).

During the brief secondary phase, markets underwent a rapid process of globalisation as a result of improved land communication facilities and economic integration processes (the European Economic Community and later the EU). Pressure on forests dropped significantly as marginal agriculture declined sharply, allowing forests to quickly recover both in biomass (growing stock) and area. The demands made on forests were restricted to wood as a bulk commodity and soon the disadvantages of Mediterranean forests eliminated them from the world market place. The positive process of forest recovery was, to some extent, conditioned by the new challenge of extended forest fires favoured both by land and forest abandonment given the lack of profitability as well as the monotony and weakness of the secondary recovery stages of forest vegetation in the Mediterranean climate. Increasing attention to landscape issues was and still is a clear sign of tertiarization.

Over the last twenty years, numerous social processes have moved society from the secondary to the tertiary logic: greater geographical mobility, rising wealth as well as population (the so-called Florida or California effect), increasing free time, aging or the dematerialization of economic demands, among others. Mediterranean forests, which had a subsidiary role in the previous phases, suddenly become one of the key frame elements of strong tertiary possibilities for tourism-based economies by providing high appreciate landscape values. However, their economic framework was, despite this, confusing given its grounding in the previous primary and secondary phases reflected in the low quality wood production, slow growth (droughts), frequent poor sites, capitalization processes of forest recovery still ongoing, high fire risk and even paradoxically, as direct and immediate consequences of tartarisation, high social and environmental demands and pressure which

2 See Laguna (1997), Otero et al. (2006), Castellnou (2007) and Plana (2007).

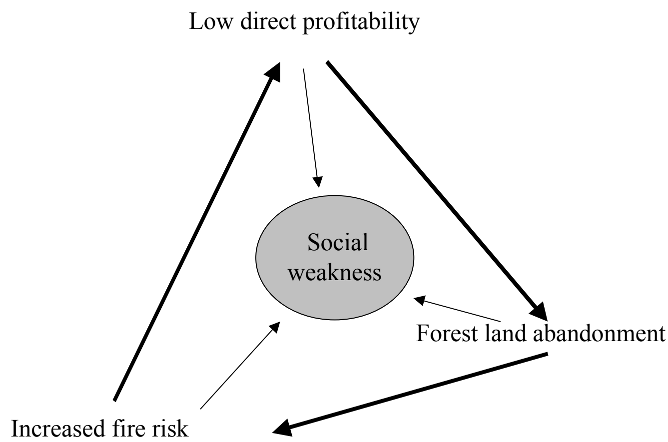

Figure 1. Perverse or vicious circle of Mediterranean forests.

were linked not to market supply but rather on command and control mechanisms (top-down policies) and extended social free riding of the main services and volatile products of the forests (i.e., parking, mushrooms, pine nuts, medicinal plants or wild fruits).

Two reasons support the belief that this dysfunctional framework is unsustainable: the high risk for forest fires resulting from extended land abandonment because of its lack of profitability as well as the imposed environmental demands arising from, quite ingenuously, urban populations without voluntary compensation or market procedures.

The main explanation for this short-term policy approach is most probably the rapid shift towards tertiary societies. Mediterranean countries have experienced in a relatively short period of time the accelerated change in their population structure from prevailing rural settlements to highly urbanized patterns. External factors such as unfair competition like that from non renewable raw materials (concrete, iron, plastics, fossil energies, etc.) as well as that between agriculture and forestry resulting from the different EU competences and funding, further worsen this disadvantaged situation.

The consequences of all this are, in economic terms, inefficient and insufficient allocation of resources for forestry maintenance and high economic stress in Mediterranean forestry, defined as macrodecoupling (Wolf, 1993 and Mendes, 1999). The higher the quotient output of environmental services outside the market supply/output of marketable products, the worsened degree of macrodecoupling that functions exactly as its opposite (microdecoupling) regarding negative external effects (e.g. pollution). In other words, if a polluter is not charged for his/her pollution, the allocation of resources will be more inefficient and more unjust the higher the pollution rate is. This functions in a manner that is identically perverse, but with the exact opposite effects regarding the production of environmental services.

This is the primary reason for the social weakness of Mediterranean forestry when compared with that of other parts of Europe, a situation that feeds back into the inadequacy of its political framework driven mostly by countries with a lower degree of macrodecoupling and thus, a lower degree of relative output of environmental services (boreal, Atlantic).

2. Towards a new paradigm for Mediterranean forests

The situation described above requires more than a new framework, it needs a new paradigm for Mediterranean forests. A first step is to gain a more knowledge regarding the evaluation of forest externalities in the region. Developments in this area of environmental economics in recent years together with the research conducted through the MEDFOREX project have allowed for reasonable progress (Merlo and Croitoru 2005). Nevertheless, this knowledge by itself is a necessary, but not sufficient, requisite for the needed changes. For decades, environmental economists have insisted on the convenience of internalising external factors as key instrument to optimize the allocation of scarce resources (Pigou 1920).

In its still incomplete implementation, most of the attention is restricted to negative external effects, whereas positive external effects have been quite neglected. The symmetric application for positive external effects seems to be the most promising and consistent strategy for overcoming the observed deficiencies. Contemporary authors have used different arguments and proposed measures and actions to this end regarding Mediterranean forests[3] as well as the international forest dialogue between the south and north. International organisations are paying more and more attention to the principle of payment for environmental services (FAO 2007), especially regarding carbon balance, watershed services and biodiversity preservation.

The key elements of the proposed framework include the following six areas:

1) Legal framework

The existing legal framework must be adapted to the existing tertiary socioeconomic reality. This is especially true for access and user rights or hunting. If ownership rights have been very efficient in relation to material economies, there is no evidence that this will not work for services. Sustainability, monitoring and accounting can be better implemented when linked to ownership based on forest management plans rather than to a free riding appropriation.

2) Improved economic governance

Most of the existing management units are completely outdated due to the scattered ownership which impedes an integrated ecosystem based on management, especially in the transformation of marketable services of recreational and environmental services (access, parking, mushrooms, hunting, etc.) (Mantau et al. 2001). In competitive economic sectors, share companies have facilitated the differentiation between ownership and management, harmonizing competitive sizes with professional management and an extended distribution of shares.

3) Improved political governance

The present challenges require complementing the inflexible old structures, characterised by abrupt borders between public and private bodies. New mixed bodies should be established so as to be capable of optimising the advantages of both the public and the private, based on mutual confidence, social consensus and complementarity. On the other hand, public

3 See Merlo and Rojas (2000), Rojas (1995, 1998, 1999, 2004 and 2006), Segura (2004), Tamames (2001) and Spanish Forest Federal Law (2003) and AIFM (2002).

administrations should concentrate on separating the regulatory function from other operational tasks (forest management, service provision) involving protected areas. Finally, monitoring, reporting and external review must be the basic procedures in public policies.

4) Eliminating free riding in volatile products

The appropriation of volatile products should be regulated by market logics, respecting ownership rights as for other forest products like wood. A sustainable supply can only be assured if integrated into management planning and its monitoring.

5) Optimizing internalisation of environmental and recreational services

The more environmental, recreational and other forest services contribute to market supply procedures, the more efficient their supply will be. Intervention may then concentrate not on compensation of market failures but rather on controlling the sustainability threshold and strategic planning.[4]

Regarding carbon sequestration, there is no objective reason why the new carbon-trade related economy must exclude forests, despite their key role in the carbon atmospheric balance, especially in the EU carbon trade. Forest carbon fixation may today compensate up to 2% of a country's emissions but cannot be traded by the forest owners. If forest carbon balances enter the carbon markets, the possibilities for enhancing the fixation efficiency of forests will considerably improve when comparing the present anarchic output of forests.[5]

The EU Water Directive has broadened the concept of internalisation of the costs in the water cycle but it is quite limited regarding the environmental services provided by forests. The planned review of this Directive should be taken as an opportunity to enlarge its environmental approach to include the watershed function of forests and the coherent economic instruments for optimising the land uses regarding their hydrological optimisation as more than 85% of the non sealed land is privately owned. This is especially relevant in Mediterranean countries considering climate change, frequent drought and heavy rainfall periods as well as increasing population in the coastal and island areas.

Positive landscape-related externalities emerge economically through tourism. In order to assure a continuous economic supply, a part of tourism revenue should be transferred to the activities that ensure the landscape frame is complemented by a sound economic circuit, particularly if its perdurability is threatened by fires. There are several options (e.g., Balearic Ecotax) to assure this transfer from tourism to forests, e.g. 1) charging conventional sun and beach tourism and allocating these resources for green tourism investment, namely the recovery of heritage, 2) establishing infrastructure and landscape incentives. With this model, a progressive and healthy shift from conventional tourism to a more sustainable tourism, generating higher shares of positive rather than negative external effects, may be achieved.

Nature, forests as well as geographical denominations are increasingly used in advertising, almost always based on a free riding supply. The fact that copyright is not used and frequently these images or denominations are shared by many owners shall not hollow the rights of the suppliers of the image rights. Politically, this opens the door to promising incentives for a sound maintenance of landscapes.

4 See National Forest Programmes (COST Action E 19).
5 In Spain the present fixation rate of forests is approximately 6% of the carbon emissions, counting only aerial biomass.

In addition to a limited social appropriation of certain volatile products as mushrooms or berries, public access is a delicate issue. On the one hand, the lack of regulation contributes to the degradation of ecosystems and of their tertiary potential. On the other hand, complete restriction and their transformation into strict market services may be socially unacceptable in most countries and even counterproductive in terms of public's acceptance of the other measures proposed. This contradictory situation is, to some extent, analogical to copyright and public access to culture and knowledge.

Reasonable intermediate solutions as in the latter case the creative commons should be envisaged as a practical step forward. Here the highest priority lies in differentiating massive, economic and destructive access and use of nature from a soft social use. An economic use of natural assets based on free riding practices should any more be acceptable. Here for clear legal definitions – and not as usual diffuse tolerated uses – as well as strict restriction of motorised access to nature are of a key relevance as most economic uses can be delimited from the social uses by the necessity of vehicles. As many of these economic uses are founded on black economies, then its regulation will also contribute to the shift towards a regular transparent economy as a positive political by-product.

Regarding biodiversity there are two fundamental options for its internalisation. On the one hand, land stewardship can mobilize private resources for funding the marginal costs of biodiversity conservation. An appropriate legal and fiscal framework should facilitate the mobilization of private resources in this issue. On the other hand, public planning is generating major differential restrictions between specific areas included in protected zones, especially Natura 2000, from those outside. The costs of this declaratory policy shall not fall only on the present managers of the affected land (farmers, forest owners) but rather they should, under equity considerations, be assumed automatically by the prescriptor (Bianco, 1998).

6) Mobilizing private resources though rising estate values

As discussed in previous publications (Merlo and Rojas, 2000), positive externalities may emerge under certain circumstances. If this has not received any attention from policy makers (norms and planning), they will remain limited to an anarchic output far from optimal. Curiously, most of the resources of the primary period have considerable potential for modern tertiary use if they are preserved and complemented by modern technology. By contrast, secondary processes driven by the specialisation and production of bulk commodities have practically no potential and they even impede any tertiary options, as well as the degradation phases of primary resources.

In fact, over the last 20 years, but especially in the past decade, the profitability of Mediterranean forestry has collapsed, yet estate values in the region have climbed to unexpected and apparently illogical levels. This process can only be explained by market discount of future tertiary values of nature much more than today visible. However, this process is not evenly distributed but rather strongly related to existing constructions and their dimension and flair and the surrounding landscape quality. In fact, the high proportion of forests and low rates of forest fires in the Balearic Islands can best be explained by the type of land settlement there and the high values forests have acquired by mobilizing extremely relevant private resources for forest and landscape conservation.

It would be strictly illogical, from this perspective, to refuse this possible funding option, despite the need for research in order to understand its main patterns so that forest and land

6 In many cases due to legal restrictions or tertiary demands, the selling of a part of the estate has to be excluded as an option.

use planning can introduce it as a main variable. In any event, liquidity should be considered in this strategy since investments are often profitable under private calculations but are limited due to lack of liquidity as the only profitable option if the whole selling of an estate.[6]

3. Conclusions and options for future research

The notable features of this new framework as well as research priorities focused on Mediterranean forests are enumerated as follows:

- New instruments of economic governance should be developed to assure that organisation structures can combine broad ownership distribution with efficient governance, especially related to image and denomination rights as well as marketing of recreational and environmental services (Mantau et al. 2001).
- New mixed bodies should be designed to address the present challenges based on cooperation between the private and public sectors. In these two cases, intra- and inter-sectorial benchmarking may allow for benefit from other experiences.
- Research related to production, utilisation, processing and marketing of non wood forest products needs further support, especially in high potential products (cork, mushrooms, medicinal plants, etc.).
- Carbon balances of forest estates should be obtained in a simple and transparent manner through forest management and planning practices (Rojas 2001) as a first step for integration in the EU carbon trade.
- The revision of the EU Water Directive should move towards an integrative approach to the watershed. Biased approaches restricted to negative environmental effects (internalisation cost) should be adjusted to environmental services provided by forests using coherent economic instruments. Research must provide practical indexes based on forest management and planning techniques to measure these services.
- Constant economic circuits for landscape services strengthened by tourist assets need to be coherently designed with all the elements taken into consideration (qualitative shifting from mass to specialized tourism, heritage, etc.).
- Access and appropriation of volatile forest products must be regulated and conducted in a socially acceptable way, favouring the distinction between economic and social uses and unacceptable activities like unrestricted vehicle access to forests.
- Land stewardship needs an adequate legal and fiscal framework in order to develop its potential as a private funding source.
- Patterns of optimising potential land values should be identified in order to include results of research in land planning, norms and land use practices.
- Financial products to overcome liquidity restrictions in a context of increasing estate values should be identified and developed.

In conclusion, the measures proposed in this paper are based on the assumption that the higher the value of Mediterranean forests, the better protected they will be from threats such as forest fires. These measures are seen from the perspective of the recent developments in environmental policies like climate change debate, green taxation and reform of the water directive, which are considered as crucial opportunities to strengthen the supportive policy framework of Mediterranean forests.

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Comparing Income from Cork Oak Forests and Shrublands: Case of Ain Snoussi, NW Tunisia

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Abstract

The Tunisian cork oak forests suffer from a lack of natural regeneration and a depletion of cork oak trees. This would ultimately cause their substitution with expanding shrubs. The objective of this paper was to compare the present discounted value of income at infinite time horizon under two scenarios. The first one supposed a perpetual assisted presence of cork oak trees using artificial regeneration. The second scenario assumed a perpetual cover of shrubs. For this purpose, a cost-benefit analysis was conducted to estimate total income and capital income. The comparison showed that, compared to shrubs, cork oak regeneration led to total income gains estimated at 5,117 TND/ha in 2002, within a discount rate of 2%. Nevertheless, the income distribution showed that although cork oak plantations were profitable for the State, they were not so for the local population. Therefore, some funds with governmental aids for assisted cork oak regeneration should be identified. In addition, a financial mechanism is needed to compensate for income losses suffered by local households.

Keywords: cost-benefit analysis; cork oak forest; forest income; Tunisia

1. Introduction

The cork oak forest plays an important role in Tunisia, first given its dominant surface area of 90,000 ha in 2003 (CNT/DGF, 2005), making it the second species after Aleppo pine; secondly, through the main contribution of cork to forest revenues and exports; and thirdly, by the different products and services offered to the local population such as forage, fuelwood,

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acorns, aromatic and medicinal plants, mushrooms, honey and forest employment. Also, this forest is well-known for its biodiversity and its role in soil protection and reduction of dam siltation.

Almost all forests are state-owned. Therefore, the State has nearly exclusive rights on forest uses. Only local inhabitants detain usage rights for livestock grazing, firewood and non-wood forest products (NWFP) harvesting, and cropping of bordering plots of land. Families work and make their own investment in livestock rearing and crop growing. In addition, poor local inhabitants are involved in forest labor, within weak market regulation, in order to give them additional income (Chebil et al. 2005).

Cork oak woodlands have been reduced by one-third in the last five decades due to overgrazing, overcutting and forest fires. Soil erosion occurs in many overgrazed lands, and especially in shrublands. In addition, the high tree damage rate, estimated at an average of 14% in the region, is caused by tree damage inflicted in cork stripping, natural death of aged trees and lack of natural regeneration (Hasnaoui et al. 2005). In this context, the conservation of cork oak woodlands could be threatened if artificial regeneration programs are not undertaken.

The objective of this research is to compare the economic gains/losses of two management scenarios at *Ain Snoussi* in northern Tunisia (Table 1): (i) cork oak plantation followed by an ideal management scenario that involves cork oak natural regeneration on an infinite time horizon, and (ii) permanent maintenance of shrublands.

A social economic analysis was conducted. The government (as State forest owner) and family incomes are discounted. Government outputs are compounded mainly in cork and partially in firewood as resource rent. Households benefit from grazing resources, fuelwood and some other gathered forest products, but the government takes on the main silvicultural investment and management costs.

Because there is no market for forest land, forest market price cannot be objectively measured. The objective here is to compare the present discounted income indicators of the *Ain Snoussi* forest, and their distribution between the Tunisian government and local families. Considering the low interest rate per year (0.75%) given by the Japanese Bank for forest management and the increased interest of the society in environmental services, a base discount rate of 2%, net of inflation, was chosen.

All market and self-consumption uses of the forest in *Ain Snoussi* were included in the economic analysis, taking grazing resources estimated price into account. The value of acorns was also estimated on the basis of the nutritive content of forage units (0.9FU/kg). A sensitive analysis on consumption of forest grazed resources and cork price was done. For comparison purposes, indicators were obtained for both scenarios.

However, public social and environmental benefits such local livelihoods as soil erosion reduction and biodiversity are omitted because of the lack of physical data and cork oak forest environmental services valuation (Campos et al. 2007a and 2007b).

The Ain Snoussi area covers 3,735 ha in the region of Ain Draham in the northwestern mountains of Tunisia. It has a humid climate with an annual rainfall that varies from 1,000 to 1,500 mm. Mixed and pure cork oak stands occupy the main share of the forest area: 74% of the total surface. The rest is in turn made up of shrublands (6%), grasslands (1%), and the remaining 19% is devoted to various crops and miscellaneous uses including constructed areas and unproductive lands (Daly et al. 2005).

Despite its high density, 586 trees/ha on average, the forest is aged and non-renewed. In fact, there is an absence of young plants: only 7% of the trees are less than 60 years old and 47% of the trees are between 100 and 160 (Stiti et al. 2005).

Animal breeding is the main activity of the local inhabitants who take advantage of various natural forest resources and commodities including open grazing, fuel-wood and other non-timber products. Thus, renewing cork oak stands involves some opportunity costs related to grazing resources use, since temporary livestock is prescribed. As an alternative, plant protectors could be used without a loss of grazing production, but, they would have a higher investment cost. Both investment requirements and opportunity costs discourage cork oak renewal.

2. Methods

2.1 Revenues

Aside from cork (C) and cork oak firewood (F_C) sales, the Tunisian State also profits from aromatic plant (AP) and mushroom resource rent (M) (their standing price) as fees for the right to harvest them, and from hunting rents (licenses) and taxes (H) paid to the forest administration by wild boar hunters in the area as well. On the other hand, local inhabitants profit from fuelwood for self consumption (F_S), honey (HO), whose revenue corresponds to the market value of its estimated productivity, from cork oak acorn collection (A), and from grazing resources (G) used by local households as livestock fodder in an ‘open-access’ context (Daly et al., 2005). Thus, Ain Snoussi State cork oak forest owner ($R_{AS,P}$) and local household ($R_{AS,H}$) revenues derived from cork oak forest management are as follows:

$$R_{AS,P} = C + F_C + AP + M + H, \tag{1}$$
$$R_{AS,H} = F_S + HO + A + G. \tag{2}$$

Goods and services match 2002 market or imputed prices (without considering subsidies or taxes on products). Cork, firewood and other forestry outputs are valued at farm gate prices. Natural resource property rights and subsistence familiar economies limit the options to separately value free-access grazing resources rent and livestock keepers’ self-employed labour cost in Ain Snoussi. This work applies a simulated trade-off for joint grazing resource rent and livestock self-employed income valuation as an alternative to substitute goods pricing (Campos et al. 2006). The cost of grazing forage units was estimated from the livestock activity net value added. Only final outputs were taken into account and the simulation function used the two unknown variables, namely forage resource cost and self-employed labor cost (Chebil et al. 2005). Thus, grazing resources rent is estimated using a subjective forage unit (FU)[1] price (0.07 TND FU^{-1}) times the number of FU extracted by animals grazing under the two scenarios. The trade-off assumes that self-employed income is equal to half of an employee wage rate.

2.2 Management expenditures

The total private expenditures (E) split into the same cost concepts as national systems of accounts are labour costs (LC), intermediate consumption (IC) of raw materials and services, and fixed capital consumption (FCC). The aggregation of IC and FCC is named as consumption expenditures (CE)[2]. Labour costs include both self-employed (SLC) and employee (ELC) labour costs. In this application, SLC is estimated for each of the forestry goods or services that are appropriated by local households as a residual value between household revenues and consumption expenditures ($CE_{AS,H}$):

$$SLC = R_{AS,H} - CE_{AS,H}. \tag{3}$$

1 A forage unit (FU) represents the energy contained in a kilogram of barley.
2 Assuming that fixed gross investment equals fixed capital consumption.

2.3 Capital and total incomes

This analysis takes into account the expected revenues and expenditures of the whole cork oak silvicultural cycle and of shrubs. Net cash flows at market prices (NCF_{MP}) are estimated as the differences between annual revenues (R) and expenditures (E) within the entire cork oak silvicultural cycle equal to capital income at market prices (CI_{MP}).

$$CI_{MP} = R - E, \tag{4}$$

Total private income (TI_{MP}) is the sum of capital and labour incomes:

$$TI_{MP} = CI_{MP} + LC_{.} \tag{5}$$

2.4 Cost benefit analysis of cork oak natural regeneration

The cost-benefit analysis techniques are applied for estimating present discounted values (PDV) from future incomes of both scenarios. The present discounted values of capital and total incomes of a cork oak forest continuous natural regeneration scenario are compared with the corresponding figures of shrublands.

The first scenario (Cl) consists of a first cycle of plantation (138 years) followed by similar cycles of natural regeneration (every 144 years) (Chaar et al. 2005), which entails that a forest owner would have to invest in silvicultural treatments, whereas the second scenario, which corresponds to a series of one-year cycles of shrub production, does not involve management expenditures (Table 1).

The present discounted value along the entire first cycle (T_1) of the cork oak forest (in TND ha^{-1}), is obtained[3] by:

$$V_1 = \sum_{t=1}^{T_1} \frac{1}{(1+r)^{t-1}} x_1(t) \text{ ; where r is the annual discount rate.}$$

$$x_1(t), t \in [1,...,T_1], \text{ with } T_1\text{=138.} \tag{6}$$

The production of the second cycle of natural regeneration (V_2) and the following cycles are assumed constant. Hence, the discounted value at infinite time horizon of the permanent regeneration corresponds to the following:

$$V_{2,\infty} = \sum_{i=0}^{\infty} \frac{1}{(1+r)^{144.i}} V_2 \text{ ; Tn = 144.} \tag{7}$$

This equation is simplified by the sum of the geometric series:

$$V_{2,\infty} \cong \left[\frac{1}{1 - \frac{1}{(1+r)^{144}}} \right] V_2 \,. \tag{8}$$

Finally, the discounted value of income for sustainable cork oak is defined as follows:

3 The PDV at infinite horizon does not require that residual land values be taken into account (Samuelson 1976).

Table 1. Physical indicators used under the two management scenarios.

Initial State	shrublands	
Type of management scenario	Cork oak plantation followed by natural regeneration	Shrubs
Type of cycle		
First cycle	Plantation	Shrubs
Second and subsequent	Natural regeneration	Shrubs
Duration of cycle (years)		
First cycle	138	1
Second and subsequent	144	1
Age at first cork stripping (year)	31	
First cycle	49	
Second and subsequent		
Number of cycles for stripping cork (year)		
First cycle		
Second and subsequent	9	
	12	
Average of annual cork production (kg/ha/year)		
First cycle	217	
Second and subsequent	234	
Grazing resources		
Forage units (FU/ha)		
First cycle	380	493
Second and subsequent	398	

Source: Calculations based on data provided by Chaar et al. (2005)

$$V_{n,\infty} = V_1 + \frac{1}{(1+r)^{138}}\left[\frac{1}{1-\frac{1}{(1+r)^{144}}}\right]V_2 \tag{9}$$

For shrublands, annual average revenues and expenditures are considered for calculating the annual income represented by τ_{ma}. The infinite time horizon present discounted value is obtained by the following formula:

$$V_{ma,\infty} = \left(\frac{(1+r)}{r}\right).\tau_{ma} \tag{10}$$

2.5 Sensitivity analysis

Both scenarios are evaluated using different discount rates in a range between 2% and 10%, presuming that different profitability rates from cork oak renewal investment may be required. Also, sensitivity analyses to cork price and forage price are done.

Discounted values of income are compared to estimate labour and total income gains and losses due to the conversion of shrublands to cork oak forest.

Table 2. Main revenues and expenditures of the first cycle of plantation (current TND /ha, year 2002).

Year	Activities	Expenditures (E)			Revenues (R)	
T_1* (138 years)		Employees cost (EC)	Consumption expenditures (CE)	Total (E)	Sales (S)	Total (R)
1	Plantation cost	1111	165	1277		
1	Fencing and other management activities	190	320	510		
15	First thinning	360	62	439		
15	First formation pruning	180		180		
15	Open to grazing			0		
31	First cork stripping	169	32	218	973	1031
31	Second thinning	370	41	411	81	81
31	Second pruning	180	20	200		
43	Second stripping	442	87	546	2037	2108
55	Third cork stripping	622	118	740	3543	3622
55	Second thinning	258	25	283	467	467
67	Fourth cork stripping	598	111	726	3406	3485
79	Fifth cork stripping	630	120	767	3730	3803
79	Third thinning	264	29	293	538	538
91	Sixth cork stripping	549	101	667	3139	3213
103	Seventh stripping	596	113	726	3530	3598
103	Fourth thinning	319	35	354	649	649
115	Eighth stripping	434	88	539	2602	2669
127	Ninth stripping	480	95	592	2734	2801

* The regeneration felling occurs at year 139 and the final felling at year 175 of the first cycle (T_1), therefore, they are considered in the following cycle (Table 3).
Source: Physical data based on Chaar et al. (2005).

3. Results

3.1 Revenues and expenditures

Tables 2, 3 and 4 give the main revenues and expenditures during the different cycles of production for the cycle of plantation, that of natural regeneration and that of shrublands, respectively. For the first one, expenditures are great at the beginning of the cycle, and revenues from cork are only obtained at year 31. The situation is quite different for the cycle of natural regeneration where the cash-flow is positive from the first year (regeneration cutting). These results are used for CBA to evaluate investment in cork oak forests and shrubs.

3.2 Comparison between both scenarios

In pure market conditions, the Ain Snoussi cork oak forest natural regeneration generates higher total and capital income PDV figures than in the shublands scenario due to the high amount of cork and wood (Table 5).

Gains are estimated at 5,117 TND/ha and 718 TND/ha, respectively, using a discount rate of 2%.

The investment alternative could generate larger amounts of employee labour income (5,631 TND/ha), more than six times that of the shrublands scenario (Table 5). However, the

Table 3. Main revenues and expenditures of the second cycle of natural regeneration and subsequent cycles (current TND /ha, year 2002).

Year T_{n-1}	T_n	Activities	Expenditures (E): Employees cost (EL)	Consumption expenditures (CE)	Total (E)	Revenues (R): Sales (S)	Total (R)
	1	Fencing and shrub clearing	308		308		
139	1	Tenth stripping	485	92	594	2,613	2,628
139	1	Regeneration felling	435	48	483	886	886
151	13	Eleventh stripping	272	52	338	1,426	1,441
153	15	Open to grazing and acorns collection			17		61
163	25	First thinning	360	37	414	675	740
163	25	Twelfth cork stripping	243	46	289	1,280	128
163	25	First formation pruning	180	243	423		0
175	37	Thirteenth stripping	223	42	282	1,183	1,245
175	37	Final felling	601	67	668	1,224	1,224
	49	First cork stripping	176	34	227	201	271
	49	Second thinning	456	51	507	927	927
	49	Second pruning	180	20	37		0
	61	Second stripping	419	80	516	1,752	1,836
	73	Third cork stripping	588	112	717	3,226	3,302
	73	Third thinning	281	31	312	573	573
	85	Fourth cork stripping	523	100	640	2,991	3,067
	97	Fifth cork stripping	603	115	735	3,466	3,536
	97	Fourth thinning	319	35	345	649	649
	109	Sixth cork stripping	457	87	561	2,623	2,693
	121	Seventh stripping	518	99	634	2,931	2,999
	121	Fifth thinning	123	14	137	250	250
	133	Eighth stripping	501	95	613	2,702	2,770

Source: Own elaboration based on Chaar et al. (2005) and Selmi (personnel communication, Forest subdivision of Nefza).

Table 4. Costs and benefits of shrubs in Ain Snoussi (Current TND /ha, year 2002).

Year	Expenditures (E)				Revenues (R)		
	Labour costs (LC)						
T_n	Employees (EL)	Self-employed (SEL)	Consumption expenditures (CE)	Total (E)	Sales (S)	Self-consumption (FO_{SC})	Total (R)
1	17.2	19.6	1.6	38.4	22.8	64.0	86.8

Table 5. Comparison of present discounted values between the two management scenarios (2002 TND / ha) – Discount rate: 2%.

Indicators	Cork oak	Shrubs	Gains (+)/ Losses (-)
Total income (TI)	9461	4344	5117
Employees Labour Cost (EL)	5631	878	4753
Self-employed Labour Cost (SEL)	645	999	-354
Grazing resources value	1438	2263	-825
Capital income (CI)	3185	2467	718

Table 6. Comparison of Total income (TI) between cork oak and shrubs (2002 TND / ha) – Discount rate: 2%.

Type	Cork oak	Shrubs	Gains (+)/ Losses (-)
	Total income (TI)		
Government	6262	282	5980
Family	2547	3183	-636
Enterprises	652	879	-227
	Capital income (CI)		
Government	1205	179	1026
Family	1902	2184	-282
Enterprises	77	104	-27

value of grazing resources from the cork oak forest, 1,438 TND/ha, is less than that of shrubs, 2,263 TND/ha.

3.3 Income distribution

Families and enterprises suffer losses of -636 TND/ha and -227 TND/ha, respectively, in their total income due to the conversion of shrublands into a cork oak plantation. The prohibition from grazing and the reduction of forage resources seriously affect family income. Nevertheless, the population greatly benefits from employee income. Despite the high investment in plantation and silvicultural treatments, the government income is enhanced by cork oak plantation (Table 6). This can be explained by the high cork and wood production.

3.4 Sensitivity analysis

The sensitivity analysis shows that income of both scenarios greatly depends on discount rates. For a cork oak plantation, the total income discounted value remains positive even at a discount rate of 8.2%, although the capital income is positive only at a discount rate lower than 2.9%.

Table 7. Sensitivity of total income (TI) and capital income (CI) to discount rates (2002 TND / ha).

Indicator	Rate	Cork oak	Shrubs	Gains (+)/ Losses (-)
Total Income (TI)	2%	9461	4344	5117
	3%	4653	2925	1728
	4%	2480	2215	265
	5%	1342	1789	-447
	10%	-247	937	-1184
Capital Income (CI)	2%	3185	2467	718
	3%	-24	1661	-1685
	4%	-1415	1285	-2700
	5%	-2092	1016	-3108
	10%	-2770	532	-3302

Table 8. Sensitivity of total income (TI) to cork prices (2002 TND / ha).

Scenario	Index of cork price in 2002=100				
	50	75	100	125	150
Cork oak	6466	7964	9461	10958	12456
Shrubs	4344	4344	4344	4344	4344
Gains	2122	3620	5117	6614	8112

Table 9. Sensitivity of total income (TI) to the value of forage resources (2002 TND / ha).

Scenario	Index of forage value in 2002=100				
	0	25	50	100	150
Cork oak	8023	8382	8742	9461	10180
Shrubs	2082	2647	3213	4344	5476
Gains	5941	5735	5529	5117	4704

Nevertheless, this scenario becomes advantageous, compared to shrubs, at a discount rate of 4% for total income and 2% if the capital income indicator is considered (Table 7).

It is also clear that these two indicators are very sensitive to cork price. A variation in price by 50% induces an increase or decrease by one third (Table 8). This has a high impact on the difference between the two scenarios. On the other hand, sensitivity to grazing resource imputed values does not have a significant impact on the difference of income between the two scenarios (Table 9).

4. Discussion

Cork oak forest renewal decision simulation can illustrate the estimated magnitude and the sort of income losses and gains accrued from cork oak regeneration, and how it affects the different economic agents that depend on or profit from controlled cork oak forest goods and services.

Planting and continuous regeneration high financial costs could explain the Tunisian cork oak forest State owners difficulty for encouraging natural regeneration of the trees. Furthermore, implementing grazing restrictions for the sake of sustainable forest management may create conflicts between Tunisian State owners and local inhabitants of Ain Snoussi. Even so, on the other hand, regeneration failure could also be aggravated by the forest owners' misperception of cork oak forest future scarcity (note that oak depletion is a gradual process) (Campos et al. 2007b).

Within a discount rate of 2%, cork oak artificial regeneration produces capital income gains compared to shrublands. However, temporary restrictions on grazing resources usage would significantly affect local family income. In this context, cork oak regeneration requires not only a financial plan for funding the initial investment expenditures, but also some compensation for local livestock keepers' income losses.

On the other hand, it is possible that the omission of environmental values, such as the role of cork oak forests and shrublands in purifying[4] water resources, reducing soil erosion, decreasing dam silting, sequestering carbon, and conserving biodiversity, underestimates actual forest income. To include them, quantifications would have to be made for a complete application of extended cost-benefit analysis. In addition, the emergence of the carbon market could help improve the economic indicators that depend on carbon quantity and price. Recent studies conducted at Ain Snoussi showed that the quantity of carbon fixed by the cork oak forest (3.6 t/ha/year) is much higher than that fixed by shrubs (0.1 t/ha/year) (Sebai et al., 2005).

5. Conclusion

This work focused on incomes derived from private goods and services controlled by forest owners and the holders of certain natural resources usage-rights. Therefore, private owner income (capital income) and total income PDVs offer an incomplete view of the social benefits of cork oak renewal (Campos et al. 2007b).

The high cost of plantation and large private income losses for local livestock keepers explain the administration's lack of effort in renewing cork oak forests. Cork oak artificial regeneration therefore greatly depends on an accurate financial plan which takes into account the development of the local economy. This work strives to give insight to the income losses that local families may incur if cork oak natural regeneration treatments and restrictions are applied.

Mediterranean cork oak forests give refuge to exceptional levels of biodiversity, provide watershed and habitat, sequester carbon, offer historically meaningful landscapes, and are pleasing to the eye. Neither those benefits nor the cost of losing cork oak forests has been accounted for (assessment of which is a costly and sometimes controversial task). Although those benefits have not been considered, their existence and the evidence of gradual cork oak forest depletion could warrant the creation of a specific programme on cork oak conservation, all the while paying special attention to mitigating livestock keepers' income losses from cork oak sustainable silviculture in North Africa (Campos et al. 2007b).

4 Woody vegetation could decrease the water amount that reach the basin rivers. Nevertheless, it is unknown the sign of the change in water flow to the rivers changing land woody land uses as they are the cork oak forest and scrubland.

Even if plantation of cork oaks cannot be economically profitable at high discount rates (more than 2%), cork oak forests should nevertheless be preserved and developed to satisfy the needs of future generations both commercially and environmentally.

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A Proposal for Defining a Dynamic Cumulative Function to Determine Multifunctional Values for a Forest Resource

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Abstract

The paper aims at defining a suitable framework to specify a dynamic cumulative algorithm which could provide, from the various forest functions, aggregate values as a function of the growth of the bio-mass. The scientific framework relates to the "Wake theory, or Kielwasser theorie", binding the development of various functions to the dimension of the bio-mass.

After recalling the multiple utilities and services performed by the forest and setting the specification for each of those here considered, a dynamic approach for the introduction and performance of each of them is described and the derived values are cumulated according to the level of the biomass considered.

It is suggested that this approach could be further explored and developed to provide opportunities for determining the Total Economic Value of a forest.

Keywords: multi-functionality; dynamic approach; forest growth; cumulative economic functions.

1. Introduction

The issue of multi-functionality amply considered in literature, from the points of view both of forest economics and of policy. At present it is fully acknowledged that forests perform several functions, some of which individually, others correlated and/or combined. In the past, although recognised, these facts presented some difficulties in their evaluation and it was customary to consider only the main relevant ones, i.e. those with a commercial base, as timber production and fruit collection. Other services, or functions, presented methodological problems and their theoretical background in the past was not clearly defined, or specified in mathematical functions.

Only recent work on externalities, namely forest externalities, (Medforex activities, to quote just one of the most prominent), has brought back interest on these issues and is relevant to provide some methodological suggestions to cope with the solution of problems in this domain.

At present is ever more frequent the request (Merlo et al. 2000) for providing a comprehensive evaluation of the total functions performed by a forest for many different purposes, namely managerial activities, design and planning exercises, policy decisions and so on. Considering forest life, all the relevant functions are taken so far separately and to provide a comprehensive approach needs to relate them to an all – embracing function (Gregory 1987).

The Wake theory, or Kielwasser theorie (Rupf 1960), provides a suitable paradigm to obtain this result, binding all functions performed by the forest, except timber growth, to this latter. This means that, for instance, the landscape aesthetic function develops according to the volume of the biomass, the greater the latter, the greater the former. However, there are also certain lags to be considered since, to go back to the previous example, it takes at least 15–20 years of forest growth to get some appreciable results in terms of a pleasant view, which will be further enhanced, as the young trees turn into high stands.

The aim of this paper is to suggest a suitable algorithm which could provide a progressive aggregation of values for each of the forest functions considered, as the trees grow . This is, therefore, a rather different approach from those aiming at defining , for instance, the Total Economic Value as a whole by stated ,or revealed preferences. This proposed approach may come rather useful in those cases where the valuers' profession is constrained by legal procedures accepting, or still rejecting, the hypothetical determination of values.

Of course, these functions are in the real world performed in a dynamic way, following laws which are specific to their nature and at the same time depending on the size of the biomass. Moreover, relating the functions to the basic one, it is necessary to provide a "binding system" which could, at any stage of the biomass level, sum up the values related to those functions which are at different stages of their development.

The algorithm presented will be utilized later in a real case, just to give evidence of its suitability to perform on a group of selected functions. Of course, the results achieved do not pretend to be exhaustive and able to provide already the Total Economic Value. Nevertheless, if the most relevant functions are considered and it is possible to define their dynamics and their timing, it could be presumed that the former could be approximated and most of all be determined at various stages of the life of the forest.

2. The forest functions

One could present a very large set of functions performed by a forest. For the sake of expediency we shall here refer only to a small set of functions listed as follow:

1) productive (timber, fruits, cork, etc.)

$$x(t) = \frac{A}{1 + ke^{-\lambda At}}$$

where
A = equilibrium value, yielding capacity
k = A/x_0-1
Λ = potential rate of growth

2) carbon fixing

$$C(t) = \frac{A}{1 + ke^{-\lambda At}} + zt$$

Where:
z= constant

3) soil erosion protection

$$I(t) = \frac{k_2}{\lambda} \int_{t_0}^{t} \frac{\lambda A e^{\lambda At}}{e^{\lambda At} + k} dt$$

$$I(t_0) = k_2 \int_{t_0}^{t} \tau_2(t) dt$$

Where:
t_0 = initial time
τ_2 = time lag
k_2 = specific proportionality constant

therefore

$$I(t) - I(t_0) = \frac{k_2}{\lambda} \ln \frac{e^{\lambda At} + k}{e^{\lambda At_0} + k} - k_2 \int_{t_0}^{t} \tau_2(t) dt \quad \text{if } \tau_2 \text{ is not constant}$$

$$I(t) - I(t_0) = \frac{k_2}{\lambda} \ln \frac{e^{\lambda At} + k}{e^{\lambda At_0} + k} - k_2 \tau_2 (t - t_0) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{if } \tau_2 \text{ is constant}$$

4) landscape aesthetics

$$P(t) = \frac{k_3}{\lambda} \int_{t_0}^{t} \frac{\lambda A e^{\lambda At}}{e^{\lambda At} + k} dt$$

$$P(t_0) = k_3 \int_{t_0}^{t} \tau_3(t) dt$$

where
τ_3 = time lag of scenic function according to the previous functions
k_3 = specific proportionality constant

Therefore

$$P(t)-P(t_0)=\frac{k_3}{\lambda}\ln\frac{e^{\lambda At}+k}{e^{\lambda At_0}+k}-k_3\int_{t_0}^{t}\tau_3(t)dt \quad \text{if } \tau_3 \text{ is not constant}$$

$$P(t)-P(t_0)=\frac{k_3}{\lambda}\ln\frac{e^{\lambda At}+k}{e^{\lambda At_0}+k}-k_3\tau_3(t-t_0)\ \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{ if } \tau_3 \text{ is constant}$$

5) recreational

$$R(t)-R(t_0)=\frac{k_3k_4}{\lambda}\int_{t_0}^{t}\ln(e^{\lambda At}+k)dt-(t-t_0)\ln(e^{\lambda At}+k))-k_4k_3\tau_3(t^2-tt_0)-k_4\tau_4(t-t_0)$$

if τ_4 is constant

$$R(t)-R(t_0)=\frac{k_3k_4}{\lambda}\int_{t_0}^{t}\ln(e^{\lambda At}+k)dt-(t-t_0)\ln(e^{\lambda At}+k))-k_3k_4\int_{t_0}^{s}\int_{t_0}^{t}\tau_3(u)dudt-k_4\int_{t_o}^{t}\tau_4(t)dt$$

if τ_4 is not constant

where:

$$\int_{t_0}^{t}\ln(e^{\lambda At}+k)dt=t\ln(e^{\lambda At}+k)\Big|_{t_o}^{t}-\lambda A\int_{t_0}^{t}\frac{te^{\lambda At}}{e^{\lambda At}+k}dt$$

The specification of these functions varies greatly as some of them relate directly to the growth of the biomass itself and others instead are function of it and moreover have different starting points according to the coverage and dimension of stands.

All these functions in the dynamic process are "time correlated" and specifically depend on:

- the related proportionality constants k_i (k_1, k_2, k_3 e k_4) which show how the analysed function grows in relation to the biomass growth;
- the time lag, or response time, of the analysed function with respect to the growth function of the biomass τ_i ($\tau_{2,}$ τ_3 e τ_4)

The envelope function of all these different functions should appear as in Figure 1, where:

x(t) = productive function
z(t) = increase due to the carbon fixing function
$\Delta I = I(t)-I(t_0)$
$\Delta P = P(t)-P(t_0)$
$\Delta R = R(t)-R(t_0)$
Vi = different values from functions.

The total value of a forest – at least considering the above mentioned 5 functions – at any time (t), could be determined by the following formula:

V(t) = x(t) + zt + ΔI + ΔP + ΔR

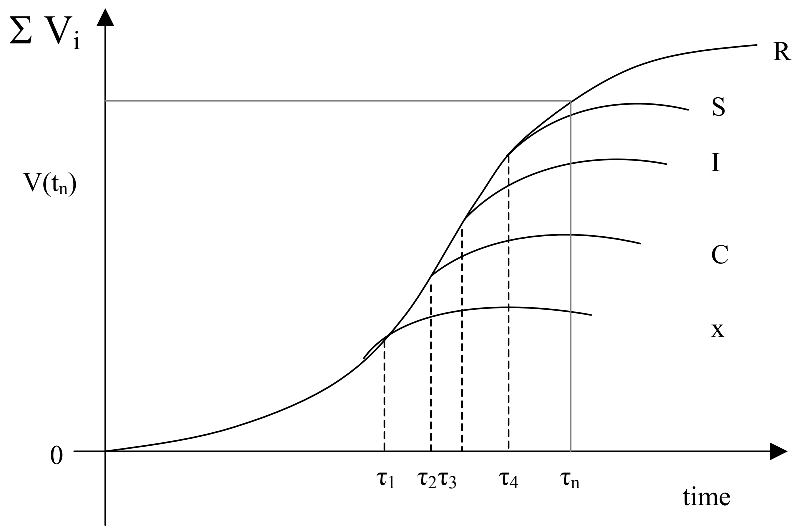

Figure 1. The cumulative progress of multiple functions.

3. Concluding remarks

This paper provided a methodological suggestion to aggregate values relating them to a physical dimension, or to the time progression. Functions specified here were considering the interactions amongst themselves, each of whom influenced the others, both spatially and timely.

This effort represented an attempt to define a methodology in the complex domain of the economic analysis of market and environmental goods. The dynamic approach followed in this case provided a strong appeal for a more thorough involvement in taking care of the development of forest.

The developed algorithm still requires further improvements, but all together, proved satisfactory and manageable by an empirical test, which is shown in the case study in appendix. However, several problems rose in handling calculus on the recreational function and this will require further improvements along with sophisticated computing procedures. As for further developments, a key issue relates to the determination of the values to attribute to time lags τ_i which could present variable sizes according to the degree of development of the eco-system referred to.

By dealing with more case studies, problems related to parameters, such as k_j and z, may find suitable solutions, since their definition is of basic relevance for each function and they respond to specific characteristics of the environmental, social, economic and geographic context considered.

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Appendix

The case study refers to a spruce forest in Tuscany (Italy) of which the growth parameters are known and shown in Table 1.

Table 1. spruce forest basic data.

Age (years)	Number of trees	Total stem volume of the biomass (m^3)	Average yearly increase (m^3)	% yearly increase (m^3/100 m^3)
10	3500	44	4.4	—
20	2930	133	6.65	8.81
30	2200	246	8.2	5.56
40	1700	380	9.5	4.06
50	1380	523	10.46	2.87
60	1150	649	10.82*	1.87
70	1000	742	10.60	1.19
80	900	808	10.10	0.78
90	900	851	9.46	0.45
100	900	875	8.75	0.23
110	800	885	8.05	0.11
120	800	892	7.47	0.07

* maximization of the incremental function

Other relevant elements to determine the total value considering the 5 functions above analysed:

- market price for spruce timber at 57 €m^3
- the productive function of forest here relates only to timber, neglecting other contributions
- t_0 = 10 years
- x_0 = 2131,8 €*
- t_m = 120 years
- A = 43217,4 €**
- k = 19,27
- $\lambda = 1{,}5 \cdot 10^{-6}$

* the whole stem measurement of the biomass has been reduced by 15% (44 m^3), equivalent to the initial period at 10 years; this amount has been multiplied by the price of 57€ to obtain the value of the forest biomass at time t_0.
** represents the maximum value given to the forest for the biomass only. It comes from the reduction of 15% of the whole tree measurement refer at t = 120 years, that is 892 m^3 , then multiplied by 57€/ m^3.

By using formula (13) in the text, that is:

$$V(t) = x(t) + zt + \Delta I + \Delta P + \Delta R$$

where:

1) $$x(t) = \frac{43217{,}4}{1 + 19{,}27 e^{-(1{,}5 * 10^{-6}) 43217{,}4 t}}$$

2) $C = \dfrac{43217{,}4}{1 + 19{,}27 e^{-(1{,}5 * 10^{-6}) 43217{,}4 t}} + zt$

3) $\Delta I = I(t) - I(t_0) = \dfrac{k_2}{1{,}5 * 10^{-6}} \ln \dfrac{e^{0.064t} + 19{,}27}{21{,}17} - k_2 5(t - 10)$

τ_2 = costant = 5 years

4) $\Delta P = P(t) - P(t_0) = \dfrac{k_2}{1{,}5 * 10^{-6}} \ln \dfrac{e^{0.064t} + 19{,}27}{21{,}17} - k_3 50(t - 10)$

τ_3 = costant = 50 years

5) $\Delta R = R(t) - R(t_0) = \dfrac{k_3 k_4}{1{,}5 * 10^{-6}} \int_{t_0}^{t} \ln(e^{0{,}064t} + 19{,}27) dt - (t - 10) \ln(e^{0{,}064t} + 19{,}27)) - k_3 k_2 50(t^2 - 10t) - k_3 20(t - 10)$

τ_3 = costant = 50 years
τ_4 = costant = 20 years (20 years after 50)

The results which will be obtained present several problems which have been now largely overcome, such as the definition of proportionality coefficients kept constant for k_2, k_3, k_4, z, which express the relationship with respect to the biomass growth x(t) of soil erosion protection, landscape aesthetics, carbon fixing, and recreational functions. As an oversimplifying example, let set:

$k_2 = k_3 = k_4 = z = 19{,}27$

However, for more realistic evaluations it is necessary to determine the real values of proportionality coefficients k and z which change according to different environmental, social, and economic systems. In particular, the values of k_i must consider the specific features relating to the environmental system, such as data about topography, geology, hydrology, forest botany, climatology.

Finally at time t = 60 years, in monetary terms applying the above quoted data we shall get:

x (t) = 31091,65
ΔC = z 60
ΔI = 14511915,84
ΔP = 14511915,84

$$\Delta R = 247555333 \int_{10}^{60} \ln(e^{0{,}064t} + 19{,}27) dt - 209{,}3 - 55680665 = 42.597.114.834$$

Summing up all these values, comes a very large total amount, sounding rather unrealistic. There are many good reasons for this strange results and, just to mention a few, we suspect that the k coefficient at 19,27 is certainly too large; another reason lies on the gross assumption to keep k_i equal, just for sake of easy handling.

As mentioned before the framework of this methodology sounds reasonable, but in term of implementing its use a more refined set of data, namely coefficients and time lags should be made available as realistic information.

Real Options in Managing Environmental Systems

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Abstract

This work is part of ongoing research that applies a real options approach to management decisions that have uncertain and irreversible environmental impacts. It considers the environmental consequences of two different types of development in an area of natural Mediterranean grove forest in the early stages of recovery following a major forest fire. The main analytic results of the models are that: in contrast to the standard real options result, the timing of development may not be a factor in the optimal decision and that the socially optimal magnitude of development will always be smaller than the private optimum.

Keywords: natural resource economics; externalities; criteria for decision making under risk and uncertainty; forestry.

1. Introduction

This work is part of research in progress applying the Real Options Approach to environmental and resource management problems. Specifically, it considers two land-use alternatives in a forest in the early stages of recovery following a major forest fire in 2005. The area in question represents 5.5% of the 8,500 ha Carmel National Park in Northern Israel and is one of relatively few sizeable naturally occurring Mediterranean grove forests in the country. The park itself contains a mix of planted forest and natural grove forest and has multiple physical, ecological and other environmental functions. It has watershed and carbon sink properties and is a source of biodiversity, providing habitat to a large number of endemic species. The uses and economic value of a number of the species native to the park has been detailed by Shamir et al. (2005) There are additional sources of economic value from recreation, amenity and existence functions. (Shechter et al. 1998) Although the park is primarily intended for preservation and conservation uses, there is also limited development within its boundaries. This includes two towns and several predominantly agricultural

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communities whose existence predated the park's creation. The park is subject to heavy pressure from development in surrounding areas. On one side, it is bordered by Haifa, Israel's third largest city. There are a number of other smaller but expanding municipalities near or adjacent to the park as well as industry and transportation networks.

The study area is, from an ecological standpoint delicate but recovering. Its ecosystem functions, compared to the pre-fire state are low. There is little vegetation, habitat or recreational value. Nevertheless, based on evidence from recovery following an earlier major fire in 1989 and from growth in other similar forest locations, left undisturbed, the area will, within 30–40 years recover its major functions.

In order to improve our understanding of the implications of the intense development pressure to which the area is subject and the range of disturbance that different development projects can cause, this research compares the ecological and economic dimensions of preservation and two types of development. The two are distinguished by the extent to which they alter the parameters of the area's recovery. The first (Type One) is already observed in the park, and involves the installation of infrastructure such as underground cables, or other over-ground structures such as access paths and parking facilities at the park's edge. These installations reduce the total preserved area but cause negligible ongoing disturbance beyond the construction phase. There is no pollution, impediment to movement of wildlife or other changes that would cause shifts in mating and other animal behaviours, seeding and germination of plant species or threshold levels of viability. The second type of development (Type Two) is more intrusive and causes ongoing disturbance that can affect the rate of recovery. The example most relevant for the study area is the proposed expansion of road infrastructure within the park's boundaries. There are plans to build a by-pass road connecting Haifa with the new northern portion of the Trans-Israel Highway. Its main route is through the park. Of the various proposed routes, several are close enough to the study area to cause disturbance; for example as the result of congestion, pollution and increasing unrestricted access to the site. (Taipero 2006)

We analyse and compare the value of environmental change under three scenarios: preservation/conservation and Type One and Type Two development. Changes at the ecological level are embedded within an economic choice framework and the decision-maker's task in to optimally choose the timing and magnitude for each project taking into account the benefits accruing to development and its expected environmental impacts over time. The optimal decision is the one that exactly equates the marginal benefit derived from the project to the marginal environmental costs. It recognizes that there is a degree of mutual exclusivity among the different uses of the resources and different degrees of irreversibility for different uses. The results are quantitatively similar to a distributionally-neutral social planner identifying the optimal level of development, the optimal externality and reservation value for environmental quality that defines the optimal timing. Qualitatively, there is a difference because the environmental cost is imposed on the resource user/developer. The spirit is very close to the User Pay Principle (UPP) in that it explicitly incorporates the environmental externality in a profit/utility maximising framework. The advantage of this is that we can make direct comparisons with comparable decisions that ignore environmental consequences. (e.g.: returns to a private developer versus the loss of public preservation benefits)

2. The models

The two development problems are analysed in stochastic optimal stopping framework. There is a single stochastic state variable representing the state of the physical/ecological environment and two control variables, the magnitude and timing of development. They have a common baseline ecological constraint prior to development. Since each project differs in its environmental impact, after development, the evolution of the environment is distinct for each case. The solution is in the form of an optimal level of development and reservation level of environmental quality that defines the boundary between the continuation and stopping regions.

2.1 Benefit flows and the objective function

The objective function for the problem contains two benefit flows, those from development and those environmental benefits foregone due to development.

2.1.1 Development benefits

Consider a project whose value is an increasing concave function ***W(.)*** of the amount invested, ***I***; $dW(I;k_t,\xi)/dI \geq 0$ and $d^2W(I;k_t,\xi)/dI^2 \leq 0$. The return to this project is deterministic and independent of the condition of the environmental resource. An additional assumption is that prices and interest rates are fixed, allowing us to ignore dynamic market conditions that may create value to delay, in the deterministic case.[1]

2.1.2 Marginal environmental impacts: loss of preservation benefits

In each period, there is a benefit flow from the preservation functions of the environmental resource. Let $q(k_t)$ be the time-invariant measure of monetary value, such that $q'(k_t) > 0$. The stochastic state variable k_t is a physical index/measure of the resource. Examples of this metric include (but are not restricted to), biodiversity indicators, site size, and total biomass. The economic value associated with the physical metric, $q(k_t)$ would include the commercial values of biodiversity (eg: pharmaceutical values of plant species), existence and bequest values, recreation and harvest values.

The preservation benefits of interest are cumulative and are considered over the continuation and stopping regions of the problem. If development occurs at $t=\tau$, $0 \leq \tau$, the cumulative benefit of preservation is:

1. $\int_0^\infty q(k_t)dt = \int_0^{\tau^-} q(k_t)_{|I=0} dt + \int_{\tau^-}^\infty q(k_t)_{|I>0} dt$

Comparing equation (1) to the alternative of never undertaking the project, $\int_0^\infty q(k_t)dt = \int_0^\infty q(k_t)_{|I=0} dt$, a negative externality implies that:

2. $\int_0^{\tau^-} q(k_t)_{|I=0} dt + \int_{\tau^+}^\infty q(k_t)_{|I>0} dt < \int_0^\infty q(k_t)_{|I=0} dt$

And the difference between the two sides of the inequality is the marginal externality associated with the project. Since both cases (i.e. never develop and develop at $t=t$) are identical over the period, $t=[0, t]$, the marginal external costs are:

1 The value of delay with dynamic but deterministic prices is analysed in Dixit and Pindyck (1994) pp. 109, 130-32 and in Oksendal (1998) ch.10.

3. $\int_{\tau^+}^{\infty} q(k_t)_{|I=0} dt - \int_{\tau^+}^{\infty} q(k_t)_{|I>0} dt$

The Objective Function: Let ***J*** be the discounted value of the project, including the option to choose its magnitude and timing. The decision problem is:

4. $J(k_\tau, I^*) = \max_{k_t, I} E\left[W(I;k_t,\xi)e^{-r\tau} - \left(\int_{\tau^+}^{\infty} q(k_t)_{|I=0} dt - \int_{\tau^+}^{\infty} q(k_t)_{|I>0} dt\right), 0\right]$

2.2 Baseline environmental constraint – geometric Brownian motion with drift

The accumulation of preservation benefits over all times for which ***I=0*** is identical for each of the models. $E\left[\int_{\tau^+}^{\infty} q(k_t)_{|I=0} dt\right]$ is therefore common to each of the management scenarios considered.

Consider the geometric Brownian motion specification for the evolution of $\boldsymbol{k_t}$, where $\boldsymbol{\alpha_0}$ is the non-random drift and $\boldsymbol{\sigma_0}$ is the instantaneous standard deviation of the process:

5. $dk_t = \alpha_0 kdt + \sigma_0 kdz$

When $\boldsymbol{\alpha_0}$ is positive, this process is a reasonable approximation of periods during which ecosystems are improving. In addition to the post-fire recovery case documented in this work, other examples include population growth of wolves in the absence of periodic culls, disease or other significant disturbance. (Bakshi and Saphores, 2004)

For a known initial value, $\boldsymbol{k_0}$**,** then for any point, $t<\tau$,

6. $E[k_t] = k_0 e^{\alpha t}$

2.3 Cumulative environmental benefits under Type One and Type Two development

Let $\boldsymbol{p_k}$ be the unit price of $\boldsymbol{k_t}$. This could be the market price of non-timber forest products; fees for different recreational uses; amenity values imputed from hedonic market studies; or the value of ecosystem services. Using a linear specification for the preservation benefits, an explicit expression of the cumulative expected preservation benefits is obtained by substituting equation (6) into $\boldsymbol{q(k_t.)}$. If development never takes place:

7. $E\left[\int_0^{\infty} q(k_t)_{|I=0} e^{-rt} dt\right] = pk_0 \dfrac{1}{r-\alpha}$

and

8. $E\left[\int_{\tau^+}^{\infty} q(k_t)_{|I=0} e^{-rt} dt\right] = pk_0 \dfrac{1}{r-\alpha} e^{-(r-\alpha)\tau}$

We assume that the larger the magnitude of development for each of Type One and Two, the larger the environmental impacts and represent these effects with the function, $\boldsymbol{f_i(I)\ (i=1,2)\ (d f_i(I)/dI > 0)}$.

For Type One development, the process of environmental change switches from geometric Brownian motion to one that combines geometric and arithmetic Brownian motion. From $\boldsymbol{t=\tau^+}$ onwards:

9. $dk_t = \alpha(k - f_1(I))dt + \sigma_0 kdz$

and the decision-maker's maximisation problems is:

10. $J_1(I^*) = \max_I E\left[W_1(I;k_t,\xi)e^{-r\tau} - p_k f_1(I)\frac{1}{r-\alpha}e^{-rt},0\right]$

In the second type of development, the process remains geometric Brownian motion, but the drift is smaller after development takes place since the rate of recovery is slower. From $\boldsymbol{t=\tau^+}$ onwards:

11. $dk_t = (\alpha - f_2(I))kdt + \sigma_0 kdz$

And the decision-maker's maximisation problem is:

12. $J_2(I^*) = \max_{I,k_t} E\left[W_2(I;k_t,\xi)e^{-r\tau} - p_k k_t e^{-rt}\left(\frac{1}{r-\alpha} - \frac{1}{r-(\alpha - f_2(I))}\right),0\right]$

2.4 Solutions to the models and discussion of results

In order to solve the decison-maker's problem, the optimal level of development, $\boldsymbol{I^*}$ and the reservation value of environmental benefits $\boldsymbol{k_t}$ must be identified. The first is obtained by equating to zero the derivative of $\boldsymbol{J}$ with respect to $\boldsymbol{I}$ and the second by substituting $\boldsymbol{I^*}$ into the more general solution to the boundary plane. Interested readers are referred to Oksendal, (1998) and Dixit and Pindyck, (1994) for general solution techniques. Freeman and Zeitouni (2004) elaborate the specific solution to the above type of problem.

In the case of type one development, the $\boldsymbol{I}$ must satisfy:

13. $W_1'(I) = \frac{\alpha}{r(r-\alpha)}$

and in the case of Type Two development:

14. $W_2'(I) = k_\tau \frac{f_2(I)}{(r-(\alpha - f_2(I)))^2}$

For both types of development, the optimal magnitude when environmental impacts are internalised, is smaller than the private optimum (i.e. $\boldsymbol{I^*}$ that satisfies the condition $\boldsymbol{W'(I)=0}$). This is the standard result predicted in externality theory. The two types of models differ in their prescription for the optimal timing of development. Since the reservation value, $\boldsymbol{k_t}$ appears nowhere in the objective function for Type One development (equation 10) the model returns the cost benefit result that if some positive level of development is to occur, it should occur in the current period (i.e. $\boldsymbol{t=0}$). In contrast, the reservation value for Type Two development is greater than the initial value when the drift parameter is positive (i.e. $\boldsymbol{k_\tau} > \boldsymbol{k_0}$; $\alpha > 0$) implying that there is a positive value to waiting until $\boldsymbol{t=\tau}$. The exact timing is determined by a direct calculation of the point at which the value functions for the continuation and stopping regions are equal. Further, the scope of development that is optimal will be greater that the size that would be dictated for the same type of development at $\boldsymbol{t=0}$.

A final significant result for badly damaged, but recovering forest landscapes is that if the environmental function is very low and expected to remain so over a long period, there may be a very strong bias toward development since both the baseline value of preservation and the marginal damage caused by development are low relative to the returns to development. This holds over all specifications of the preservation benefit function. While this result may be economically *efficient*, it may not be *desirable*. It provides a variant of Cropper et al.'s (1979) economically optimal extinction and illustrates the types of non-convexities that need to be considered in this and similar modelling efforts (Dasgupta and Maler, 2003).

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The Mediterranean basin constitutes a unique mosaic of terrestrial, freshwater and marine ecosystems, as a result of a distinct regional climate imprinted on a dynamic topography. Mediterranean forest ecosystems provide multiple wood and non-wood forest products and services which are crucial for the socio-economic development of rural areas as well as for the welfare of the urban areas of the Mediterranean region. Mediterranean forests require special attention due to several questions:

(i) They constitute a unique world natural heritage and play a key role in the welfare of urban and rural Mediterranean societies. The goods and services that they produce are very diverse (multi-functionality) and have a great market (many non-wood products) and non-market value (externalities).
(ii) They represent an exceptional richness in terms of biodiversity.
(iii) They are very vulnerable to numerous factors: forest fires, over-exploitation, degradation and desertification.
(iv) Their conservation and management affects the availability of soil and water resources, this last one being a key strategic resource for Mediterranean societies.
(v) Their future (as being in a transitional zone) is seriously endangered by climate change.

The seminar "Scientific Tools And Research Needs For Multifunctional Mediterranean Forest Ecosystem Management" was organized in Solsona, Spain on November 27–28, 2006. The seminar brought together Mediterranean scientists from several disciplines to discuss and present new techniques, tools and results that can provide solutions to the complex Mediterranean forest ecosystem management problems.

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