Modeling the influence of alternative forest management scenarios on timber production and carbon storage: a case study in the Mediterranean region

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Abstract
Forest ecosystems are fundamental for the terrestrial biosphere as they deliver multiple essential ecosystem services (ES). In environmental management, understanding ES distribution and interactions and assessing the economic value of forest ES represent future challenges. In this study, we developed a spatially explicit method based on a multi-scale approach (MiMoSe - Multiscale Mapping of ecoSystem services) to assess the current and future potential of a given forest area to provide ES. We integrated a GIS-based model, scenario model, and economic valuation to investigate two ES (timber production and carbon sequestration) and their trade-offs in a test area located in Molise region (Central Italy). Spatial information and trade-off analyses were used to assess the influence of alternative forest management scenarios on investigated services. Scenario A was designed to describe the current Business as Usual approach. Two alternative scenarios were designed to describe management approaches oriented towards nature protection (scenario B) or wood production (scenario C) and compared to scenario A. Management scenarios were simulated at the scale of forest management units over a 20-year time period. Our results show that forest management influenced ES provision and associated benefits. In the test area, the Total Ecosystem Services Value of the investigated ES increases in scenario B for the 85% and decreases in scenario C for the 82% when compared to scenario A. Our study contributes to the ongoing debate about trade-offs and synergies between carbon sequestration and wood production benefits associated with socio-ecological systems. The MiMoSe approach can be replicated in other contexts with similar characteristics, thus providing a useful basis for the projection of benefits from forest ecosystems over the future.

Keywords: Ecosystem services, Forest management, InVEST, MiMoSe model, Trade-off analysis.

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Introduction

Ecosystems provide a range of goods and services that are important for human well-being and environmental health, which are collectively called ecosystem services (ES) (Costanza et al., 1997; TEEB, 2010). Forests deliver multiple, essential ES commonly classified as provisioning (e.g., wood and non-wood products), regulating (e.g., carbon sequestration) and cultural services (e.g., landscape aesthetic value) (MEA, 2005; Haines-Young and Potschin, 2013).

ES have become a key concept in understanding the way humans interact with the natural environment (Costanza et al., 2014; Thorsen et al., 2014). Human activities have shaped ecosystems for millennia across the terrestrial biosphere (MEA, 2005), and forest ecosystems are continuously exploited or degraded by human-induced pressures (Foley et al., 2005; Köchli and Brang, 2005; Haberl et al., 2007; Lindner et al., 2010; Deng et al., 2013; Lafortezza et al., 2008, 2013).

Understanding ES interactions, their trade-offs or synergies, as well as the drivers influencing these interactions represents a challenge for environmental management and can help to identify effective management practices (Rodríguez, 2006; Bennett et al., 2009; Garcia-Gonzalo et al., 2015). To this end, trade-off analysis is used to understand how an ES changes as a function of other ES (Rose and Chapman, 2003; Maass et al., 2005; Rodríguez et al., 2006; Ruijs et al., 2013).

The economic value of ES provided by forests has been assessed since the middle of the last century (Clawson and Knetch, 1966; Hoehn and Randall, 1987; CBD, 2001). More recently, research has been directed at the spatial analysis of ES value. To this end, Van der Horst (2006) and Baerenklau et al. (2010), for example, aggregated the economic value of ES with other relevant forest characteristics at the spatial level.

Mapping and quantifying the supply and demand of ES is a key step toward identifying the appropriate institutional scale for decision making (Swetnam et al., 2011) and for delivering the ES concept in environmental institutions (Daily and Matson, 2008; Kroll et al., 2012; Marchetti et al., 2012). The EU Biodiversity Strategy to 2020 has highlighted the need to map and assess the state of ecosystems and their services in Member States from 2014 to 2020 (EC, 2011; Maes et al., 2013, 2014). Thus, standardized methodological approaches are needed to quantify and map ES (Crossman et al., 2013; Drakou et al., 2015) in order to combine the rigor of small-scale studies with the breadth of broad-scale assessments (Chan et al., 2006).

Many studies have investigated the impact of land use change scenarios on ES (e.g., Burkhard et al., 2009) by adopting, for instance, the InVEST (Integrated Valuation of Environmental Services and Tradeoffs) model. InVEST is a large-scale scenario model that simulates variations in biodiversity and ES under different future-oriented land use changes (e.g., Nelson et al., 2009). From global to landscape scale, the InVEST model has been recently used to explore the potential impacts of land use change under alternative policy scenarios (Lawler et al., 2014), evaluate environmental and financial implications for ES provision among different planning scenarios (Goldstein et al., 2012), and assess the impact of conservation policies on biodiversity and habitat quality (Wu et al., 2014).

In Europe, the InVEST model has been mainly applied to assess watershed regulating services in the Czech Republic (Harmáčková and Vačkář, 2015) and map pollination services at the landscape scale (Zulian et al., 2013). Many studies have been carried out in the Mediterranean region. For example, in Spain InVEST has been applied to the evaluation of hydrological services, i.e. water quality and quantity (Terrado et al., 2014; Bangash et al., 2013; and Marquès et al., 2013). Although the InVEST model is tailored to the need of simulating land use change scenarios and services provision, never before has there been an attempt to apply InVEST for assessing the impact of alternative management approaches on ES provision over time.

In the context of global climate change, understanding how different forest management practices affect the provision of forest ES at different scales still remains a key challenge for decision-makers (Scarascia-Mugnozza et al., 2000; Kolström et al., 2011; Wang et al., 2012, 2013).

Several studies have analyzed the effects of alternative forest management options and harvesting intensities on landscape pattern and habitat suitability (Radeloff et al., 2006; Shifley et al., 2006, 2008). Other studies have used forest management models and tree growth simulation models

integrated with a Geographical Information System (GIS) to estimate the impacts of different forest management strategies (e.g., different rotation lengths and different harvesting intensities) on ES to understand the trade-offs and potential synergies among multiple ES (e.g., timber production and carbon sequestration) (Yousefpour and Hanewinkel, 2009; Buma and Wessman, 2013; Cademus et al., 2014) often associated with biodiversity (Seidl et al., 2007; Yousefpour and Hanewinkel, 2009; Temperli et al., 2012; Kašpar et al., 2015) or water yield (Cademus et al., 2014).
In the forestry sector, timber (or biomass) production and carbon storage and sequestration are the most studied ES. Timber and carbon, which are considered indicators for the provisioning and regulating services delivered by forests (Maes et al., 2014), are competing services as an increase in timber production generally determines a reduction in carbon sequestration.
National Forest Inventory (NFI) data have been used to analyze the trade-offs between carbon sequestration and timber production at the national and regional scale (e.g., Backéus et al., 2006; Cademus et al., 2014), but only a few studies have investigated the effects of alternative silvicultural strategies on ES provision at the operational level of the forest management unit (FMU) (Seidl et al., 2007). The possibility to simulate at the FMU scale the effect of alternative management strategies on the supply of ES is crucial, especially in the Mediterranean region which is characterized by a small-scaled and fragmented ownership structure.
In Italy, only a few studies have explored forest ES. Ferrari and Geneletti (2014), Schirpke et al. (2014), and Häyhä et al. (2015) mapped and assessed multiple ES in Alpine forests (Northern Italy). Morri et al. (2014) evaluated the supply and demand of forest ES between coastal areas and upstream lands in an area of the Apennine Mountains (Central Italy). Zurlini et al. (2014) evaluated land cover transformations, processes and provisioning ES from local to global scale. It is worth noting that these studies lack a standardized approach as well as attempts to upscale results at a broader scale (e.g., regional or national).
In this study, we present a spatially explicit method based on a multi-scale approach (MiMoSe - Multiscale Mapping of ecoSystem services) to assess the current and future potential of a given forest area to provide ES. We integrated a GIS-based model, scenario model, and economic valuation to investigate two ES (timber production and carbon sequestration) and their trade-offs in the Molise region (Central Italy). Spatial information and trade-off analyses were used to assess the influence of alternative forest management scenarios on the investigated services.

2. Materials and methods

2.1 Study area

The Molise region in Central Italy covering 443,758 ha was chosen as the study area (Figure 1). The elevation on the eastern Adriatic Sea coast reaches 2050 m a.s.l. (Matese massif). The climate is temperate, which is typical of the Mediterranean region (Rivas-Martinez, 2004). Forests and other wooded lands cover 32.8% of the study area (Vizzarri et al., 2015). Turkey oak (*Quercus cerris* L.) (40% of the total forest area), downy oak (*Q. pubescens* Willd.) (22% of the total forest area), and European beech (*Fagus sylvatica* L.) (9.5% of the total forest area) are the most widespread Forest Categories (FCs). Coppice systems account for 76% of the total forest area, while high forests and coppices converting to high forest cover the remaining 24% (Garfì and Marchetti, 2011). In Molise, as well as in other Italian regions, the coppices frequently exceed the customary rotation age due to the post-WWII coppice crisis when firewood and charcoal were replaced by fossil fuels (Ciancio and Nocentini, 2004; Ciancio et al., 2006).

Figure 1. Study area (Molise region).

2.2 Data

The Forest Management Unit (FMU) in the study area was based on a FCs map (Chirici et al., 2014) created at a 1:10,000 scale level where each polygon is homogenous in terms of FCs and forest management systems (Vizzarri et al., 2015). For the purposes of this study, the original map was modified to reduce the size of FMUs larger than 15 ha to obtain forest compartments with a size similar to that implemented by forest management practices according to local forest regulations. The new FMU map contains 54,049 forest management units ranging from 0.5 to 15 ha, with an average size of 2.7 ha (SD=±2.8). Table 1 summarizes the main quantitative forest variables available for each FMU.

Table 1. Summary of the main quantitative forest variables available for each forest management unit (*See Federici et al., 2008 for calculations).

Forest variable	Measurement unit	Average value (±SD)
Altitude	m a.s.l.	686 (±294)
Slope	%	26.8 (±14.7)
Age	years	27.3 (±20)
Standing volume	m^3 ha^{-1}	117 (±86.9)
Above-ground biomass*	Mg C ha^{-1}	48.3 (±36.9)
Below-ground biomass*	Mg C ha^{-1}	8.9 (±8.3)
Dead Organic Matter*	Mg C ha^{-1}	10.2 (±4.5)
Soil Organic Carbon*	Mg C ha^{-1}	49.9 (±26.2)

2.3 Forest management scenarios

Three alternative forest management scenarios were considered, hereinafter named A, B and C. Scenario A is based on the continuation of the current forest management approach. It was designed according to the local forest regulations (regional forest management guidelines) to represent the current Business as Usual (BaU) forestry approach. Two alternative scenarios were designed to describe management approaches directed more towards nature protection (scenario B) or wood production (scenario C) compared to scenario A. In B, forest management is focused on biodiversity and nature conservation to contribute to mitigation and adaptation to climate change by preserving higher carbon stocks in forests. This is pursued by increasing the share of unmanaged forests (i.e., forests left to natural evolution) and by promoting species mixtures, natural regeneration, re-naturalization of coniferous plantations and structural diversity, for instance, by favoring continuous cover and multilayered stands (e.g., Seidl et al., 2007; Wagner et al., 2014). In

C, wood production is favoured by increasing the coppice area and reducing the rotation age. Scenarios B and C were compared to scenario A to calculate their potential costs and benefits in relation to the continuation of the present forest management approach. For scenarios A and C, we assumed that all the forests were managed as even-aged systems, while for scenario B we assumed that all high forests were managed with the aim of transitioning towards uneven-aged systems.
The main criteria for management scenarios A, B and C are shown in Table 2.

Table 2. Main criteria for management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production).

Management system	Criteria	Scenario A	Scenario B	Scenario C
Coppice	Restrictions	Coppices older than 30 years must be converted to high forest	Inside protected areas all coppices must be converted to high forest. Outside protected areas coppices older than 30 years must be converted to high forest	Coppices older than 50 years must be converted to high forest
	Silvicultural system	Simple coppice Coppice with standards	Coppice with standards	Simple coppice Coppice with standards
	Harvesting age	Min age: 20 years Max age: 30 years	Min age: 20 years Max age: 30 years	Min age: 15 years Max age: 50 years
Coppice in conversion to high forest	Restrictions	No thinning for stands from 30 to 50 years of age	No thinning for stands from 30 to 50 years of age	None
	Silvicultural system	Shelterwood system	Selection system	Shelterwood system
	Regeneration cutting age	100 years	Not considered	80 years
High forest	Restrictions	No thinning for stands from 70 to 100 years of age	No thinning for broadleaved stands from 70 to 100 years of age	No thinning for stands from 70 to 80 years of age
	Silvicultural system	Shelterwood system for broadleaves Clearcut system for conifers	Selection system	Shelterwood system for broadleaves Clearcut system for conifers
	Rotation age	100 years	Not considered	80 years

In each scenario some limitations were considered on the basis of terrain slope and the presence of strict nature reserves and degraded forests (e.g., infra-opened forests); in these woodlands wood harvesting was forbidden and the forest was left to natural evolution. For terrain slope, the forest was considered not available for wood supply when slopes were greater than 60%, 70% and 80% for scenarios B, A, and C, respectively.
For broadleaved forests, three management systems were considered: coppice (including simple coppice, i.e. coppice without standards, and coppice with standards), coppice in conversion to high forest, and high forest. For the coppice system some restrictions were taken into account on the basis of forest age and presence of protected areas: the coppice system was restricted to forests younger than 30 and 50 years in scenarios A and C, respectively, while it was forbidden inside protected areas in scenario B where coppice stands were converted to high forest. Depending on forest age, simple coppice was restricted to riparian forests and some minor FCs, as for example false-acacia (*Robinia pseudoacacia* L.) and ailanthus (*Ailanthus altissima* Mill.) stands in scenarios A and C, and chestnut (*Castanea sativa* Mill.) forests in scenario C. Coppice in conversion to high forest was based on gradual thinning of shoots to prepare the stand for the regeneration cut (Ciancio and Nocentini, 2004). Thinning was not allowed in stands from 30 to 50 years of age in scenarios A

and B, while no restriction was considered in scenario C. For the high forest system (coniferous or broadleaves), the restriction referred to the years that separate thinning and regeneration cutting: thinning was allowed in stands younger than 70 years in all scenarios, while regeneration cutting was allowed in stands older than 80 (scenario C) and 100 years (scenarios A and B).

Regarding the silvicultural systems, clearcutting was used for coppices in all scenarios and for coniferous forests in scenarios A and C. For coppices the cutting age ranges between 20 and 30 years in scenarios A and B and between 15 and 50 years in scenario C. For coniferous forests, a 100- and 80-year rotation age with artificial regeneration was used in scenarios A and C, respectively. According to local forest regulations, we considered that each clearcut area should not exceed 15 ha in coppices in all scenarios, and 3 ha in high forests in scenario A. The uniform shelterwood system was used for broadleaved high forests with a 100- and 80-year rotation age in scenarios A and C, respectively. In A and C, the shelterwood system was used as regeneration cutting for coppices in conversion to high forests. In scenario B, the selection system was used both for coppices in conversion to high forests and for high forests.

Finally, harvesting intensity and percentage of removal of forest residues were defined (Table 3). Harvesting intensity was based on national literature (Piussi, 1994; Ciancio, 2009, 2011) and in cooperation with local forest managers. For coppices with standards, the volume of standards left after clearcutting for the different FCs was derived from inventory data (MAF/ISAFA, 1998). Forest residues were estimated taking into account that approximately 77% of wood volume (m^3) is available for wood supply and the remaining as residuals (Bernetti and Fagarazzi, 2003). The share of removed residues was set to 75% and 90% of total residues for scenarios A and C, respectively. For scenario B, the share of removed residues varied from 0% to 75% depending on terrain slope, as reported in EEA (2006).

Table 3. Harvesting intensity (percentage of total growing stock) and percentage of the removal of forest residues for management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production).

Management system	Scenario A	Scenario B	Scenario C
Simple coppice	100%	-	100%
Coppice with standards	60-70% depending on FCs	40-65% depending on FCs	70-80% depending on FCs
Coppice in conversion to high forest	Thinning: 50% for the first thinning, then 20% Seed cut: 30% Removal cuts: 30%	Thinning: 50% for the first thinning, then 15% Selection cut: 10-30% based on the concept of the minimum growing stock (Ciancio, 2011)	Thinning: 50% for the first thinning, then 25% Seed cut: 30% Removal cuts: 30%
High forest	Thinning: 20% Broadleaved species: Seed cut: 30% Removal cuts: 30% Coniferous species: Clear cut: 100% (max 3 ha)	Thinning: 15% Selection cut: 10-30% based on the concept of the minimum growing stock (Ciancio, 2011)	Thinning: 25% Broadleaved species: Seed cut: 30% Removal cuts: 30% Coniferous species: Clear cut: 100%
Share of removed residues in % of total residuals	75%	0-75% depending on terrain slope (EEA, 2006)	90%

FC = Forest Category.

The forest management scenarios were simulated at the scale of FMUs over a period of 20 years (2015-2035) by using the model based on the area control method and the current annual increment (CAI, $m^3\ ha^{-1}\ year^{-1}$), as reported by the Italian NFI (Gasparini and Tabacchi 2011) for the different FCs and forest management systems in the study area. During the simulation period under study, thinning operations were performed twice, once every 10 years, depending on forest management and silvicultural systems. Flow diagrams for management scenarios A, B and C are shown in Figures 2, 3 and 4.

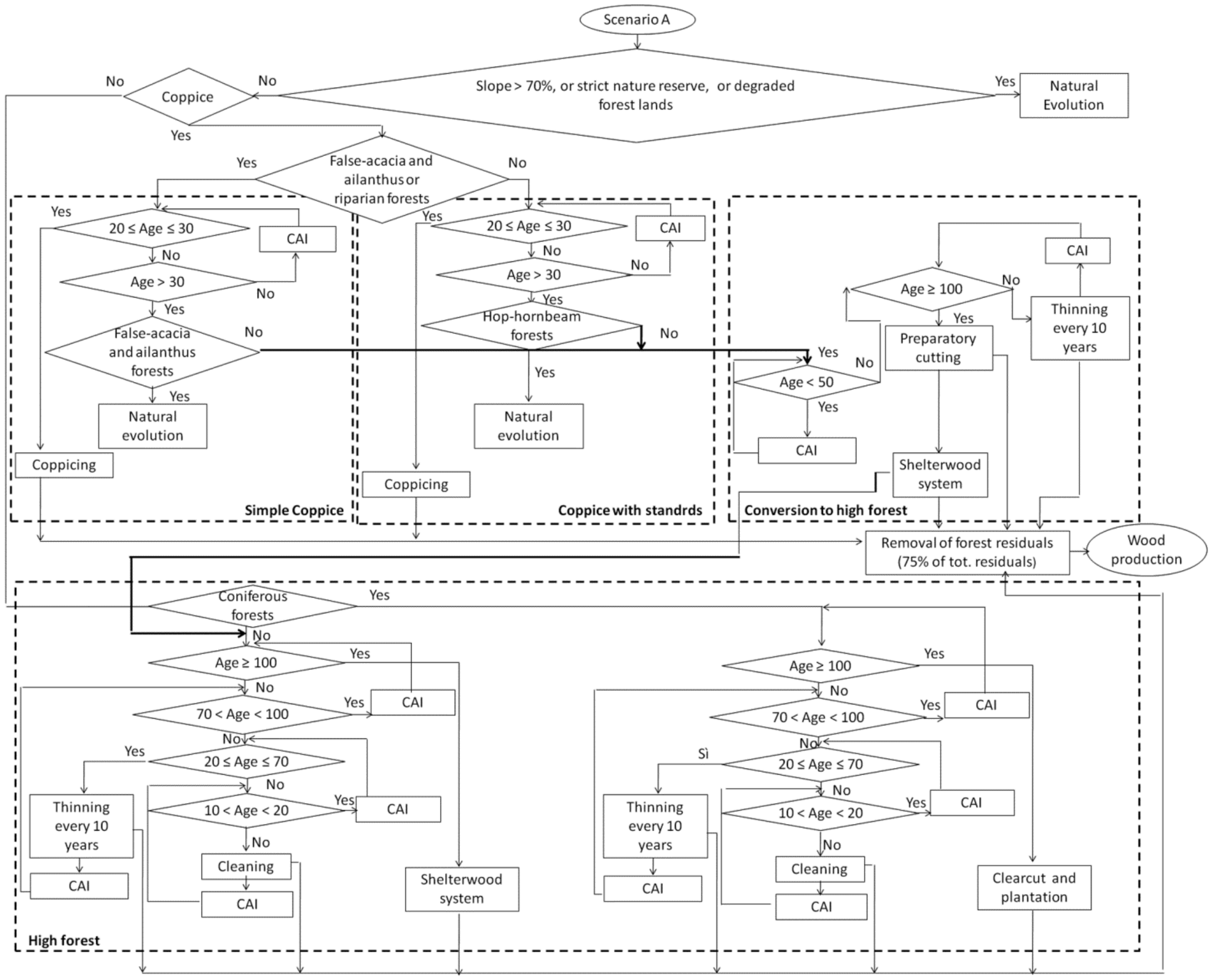

Figure 2. Flow diagram for management scenario A (Business-as-Usual) (CAI = current annual increment).

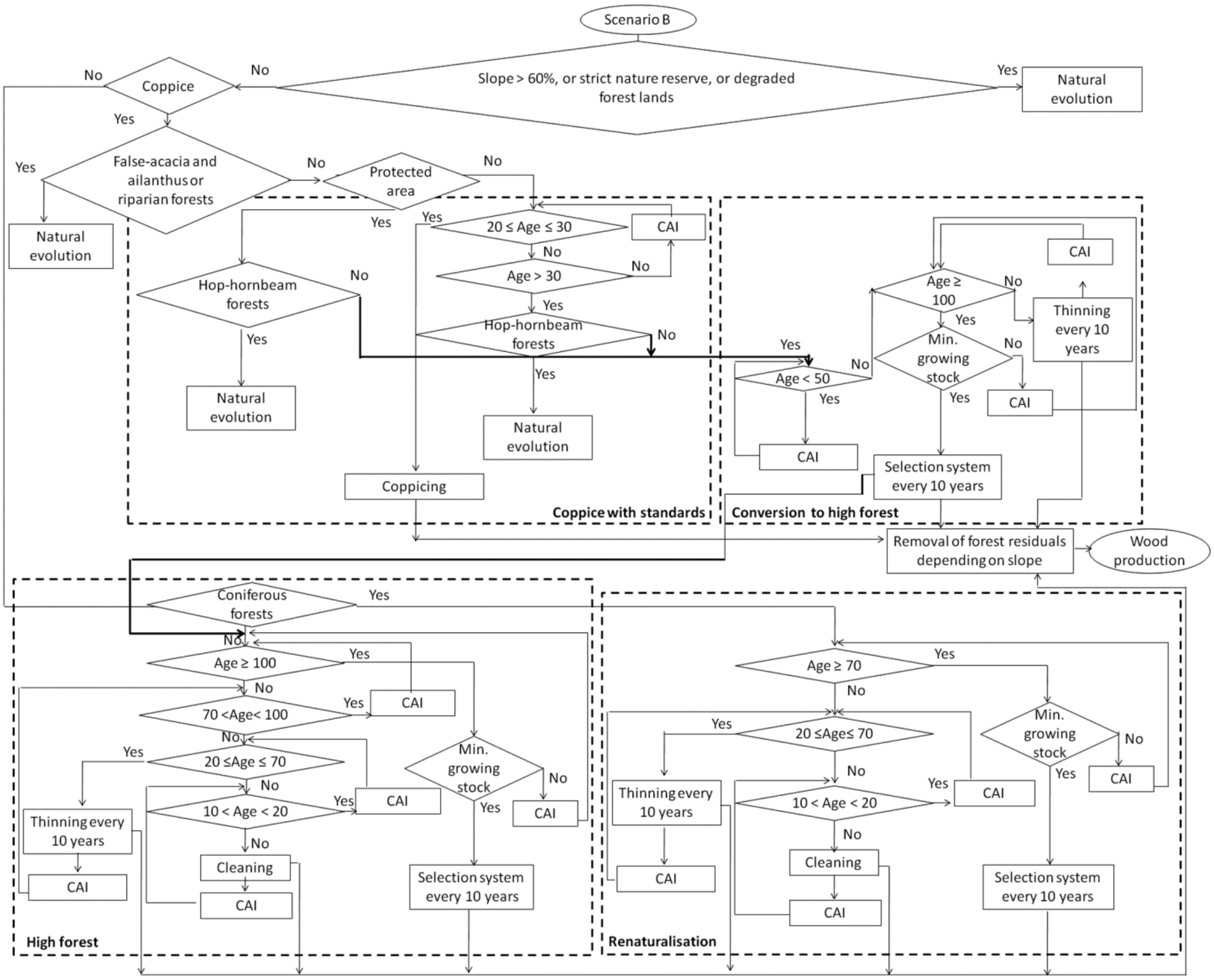

Figure 3. Flow diagram for management scenario B (nature conservation) (CAI = current annual increment).

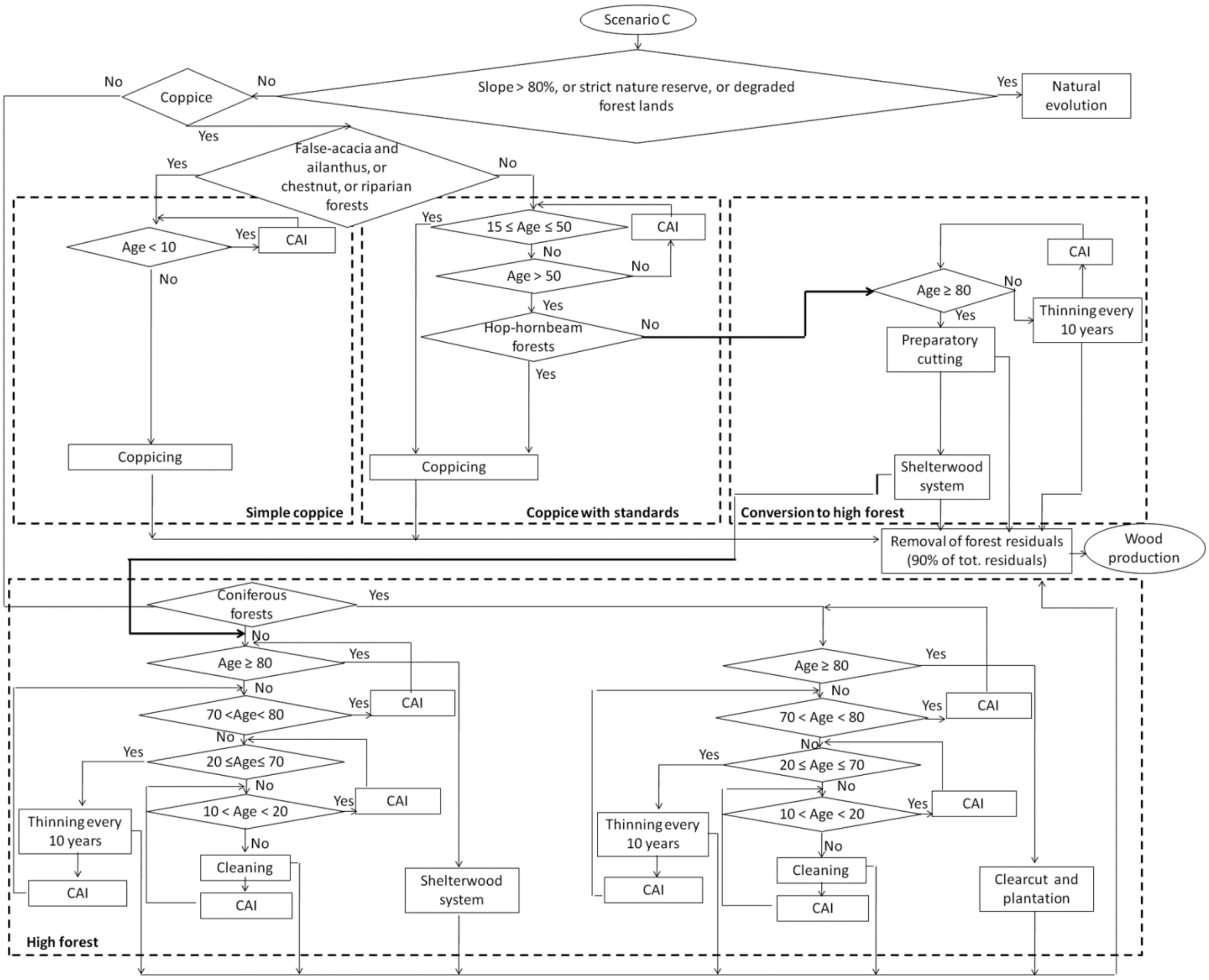

Figure 4. Flow diagram for management scenario C (wood production) (CAI = current annual increment).

2.4 Model description and parameterization

We used the InVEST model to assess timber provision and carbon stock in both biophysical and economic terms for each FMU. The InVEST model was chosen for the following reasons: (i) the model is freely available and suitable for a wide range of environmental assets; (ii) the model is structured into tiers characterized by different input data requirements; and (iii) the model implements 'production functions', which are useful to provide more accurate and policy-relevant results. The FMU map was used as the main input stratum.

2.4.1 Timber production

To simulate the amount of timber harvested from 2015 to 2035, the InVEST Managed Timber Production model was used. This model quantified the volume of harvested timber according to harvest intensity and frequency, rotation periods, and the associated monetary values based on market prices. The model was partially modified by adopting different harvest intensity and frequency thresholds based on alternative management scenarios, as implemented in each FMU.
To calculate the Net Present Value (NPV) for each FMU (in terms of € ha^{-1}), a set of equations was used. The net value of a harvest during a harvest period in the *x*-th FMU was estimated using equation (1):

$$VH_x = Harv\_mass_x \cdot (Price_x - Harv\_cost_x)[\text{€ ha}^{-1}] \quad (1)$$

where VH_x is the monetary value generated during a period of harvest in the x-th FMU (€ ha^{-1}); $Harv\_mass_x$ is the quantity of wood removed from the x-th FMU during a harvest period based on the harvest intensity indicated by the management scenario (Mg ha^{-1}); $Price_x$ is the market price of timber extracted from the x-th FMU (€ Mg^{-1}); and $Harv\_cost_x$ is the cost of removing and delivering $Harv\_mass_x$ to a processing facility or transaction point (€ Mg^{-1}).
We used a market price of 95€ Mg^{-1} for hardwood and of 65€ Mg^{-1} for softwood based on technical literature (e.g., http://www.rivistasherwood.it/tecniko-e-pratiko.html) and by consulting several experts of the local timber market. According to ISTAT (2003) and other local available data, we assumed that the whole amount of timber harvested is used as firewood, because other uses of wood can be regarded as negligible in the study area. Considering the difficulties in estimating harvest costs and their variability at regional scale (due to the impact of many factors such as slope, distance from roads, forest accessibility, etc.), we decided to use 45 and 60€ Mg^{-1} as harvest costs for coppices and high forests, respectively. As a consequence, the frequency/intensity of interventions (i.e. differences in adopted silvicultural systems) is the only criterion for the simulation of different management scenarios.
The stream of net harvest revenues of the x-th FMU was aggregated and appropriately discounted using equation (2):

$$NPV_x = \sum_0^t \frac{VH_x}{(1+r)^n} \quad [€\ ha^{-1}] \tag{2}$$

where r is the market discount rate and n is the number of years until cutting.
Finally, the Total Net Present Value (TNPV) was calculated for each FMU using equation (3):

$$TNPV_x = compartment\_area_x \cdot NPV_x \ [€] \tag{3}$$

where $compartment\_area_x$ is the area of the x-th FMU and NPV_x is the TNPV of the x-th FMU.

2.4.2 Carbon sequestration

To assess forest carbon sequestration from 2015 to 2035, the InVEST Carbon Storage and Sequestration model (Daily et al., 2009; Tallis et al., 2013) was used. This model estimates the net amount of carbon stored in a forest compartment over a given period, the total biomass removed from a harvested area of the FMU, and the social values of the carbon sequestered in the remaining stock. For each class, the model requires an estimate of the amount of carbon stored by each of the fundamental carbon pools according to the Good Practices Guidance for Land Use, Land Use Change and Forestry (GPG-LULUCF) classification and definition: living biomass, both above ground and below ground, dead organic matter, including dead wood and litter, and soil organic matter (Penman et al., 2003). To estimate carbon storage in 2035 we subtracted the harvested biomass, as estimated by the scenario, and added the CAI (Gasparini and Tabacchi, 2011) to the current growing stock volume to simulate forest growth. To estimate the value of carbon sequestration [€] equation (4) was used:

$$value\_seq_x = V \frac{sequest_x}{year\_fut - yr\_cur} \cdot \sum_{t=0}^{yr\_fut - yr\_cur - 1} \frac{1}{(1+\frac{r}{100})^t} \ [€] \tag{4}$$

where $sequest_x$ is the carbon amount (C) sequestered by each pool in the x-th FMU; $year\_fut – yr\_cur$ is the simulation period between current and future dates; and r is the market discount rate.
The Social Cost of Carbon (SCC) was used to calculate the economic value of carbon sequestration. Also known as the marginal damage cost of carbon dioxide, SCC represents the total value of the incremental damage due to a small increase in carbon dioxide emissions (e.g., Tol, 2009). We used a SCC of 37$ Mg^{-1} of CO_2 (about 109 € Mg^{-1} of elemental C) (OIRA, 2013) and assumed this value

to remain stable for the entire simulation period. Finally, the Total Social Cost of Carbon (TSCC) was calculated for each FMU by adopting the same approach illustrated by the equation (3).

2.5 Analyses

2.5.1 Sensitivity analysis

Environmental analyses are characterized by different degrees of uncertainty (e.g., Malczewsky, 2004). Because of the uncertainty usually affecting the choice of the market discount rate (see Adger et al., 1995) for timber production (Krieger, 2001; Ciancio et al., 2007) and carbon (Emission Trading Scheme, http://ec.europa.eu/clima/policies/ets/index_en.htm), we carried out a sensitivity analysis to understand the influence of discount rates on TNPV and TSCC. In addition, we used the sensitivity analysis to compare forest management scenarios among each other. To do this, we used discount rates ranging from 1% to 8% as suggested by Krieger (2001) and Ciancio et al. (2007) for timber production, and by Nordhaus (2007) for carbon sequestration.

2.5.2 Trade-off analysis

We assessed trade-offs between timber and carbon using the BaU scenario (scenario A) as baseline. We considered that the best scenario should be one those providing more economic benefits and increased timber removal without affecting the balance or equilibrium (E) among the examined ES. Although the concept of equilibrium is dynamic, we used E as a means to understand the interdependence of ES (Costanza et al., 1997; Tschirhart, 2000; Finnoff and Tschirhar, 2008). To evaluate the threshold value of timber removed (m^3 ha^{-1}), we calculated the Total Ecosystem Services Value (TESV) as the sum of TNPV and TSCC. TESV represents the capacity of a given area to provide multiple services (Maes et al., 2012). With a TESV of zero there are no added benefits to society, only additional costs.

For the trade-off analysis, we (i) evaluated the E values between TNPV (€ ha^{-1}) and TSCC (€ ha^{-1}) and (ii) analyzed the supply of the examined ES when the TESV was equal to zero (Farber et al., 2002). To facilitate decision making and to visualize the different service provision areas, three TESV maps were created where each FMU represents the capacity to provide multiple services (de Groot et al., 2010; Maes et al., 2012).

For each scenario we used a regression analysis approach to describe the relationship between TNPV, TSCC and TESV as a function of timber removed. TSCC and TESV presented a linear relationship when plotted against the timber removed, thus the model was expressed in the form: $Y=\beta_0+ \beta_1X$. Because TNPV showed heteroscedasticity when plotted against the timber removed, a multiplicative model in the form $Y=\beta_0 X^{\beta1}$ was used (Baskerville, 1972). Model parameters were estimated in linear form using logarithmic transformation: $\ln(Y) = \ln(\beta_0)+ \beta_1\ln(X)$. To facilitate the comparison between scenarios, it was necessary to convert the TNPV model back to the untransformed values. The back-conversion introduced a bias, which was corrected by adding half of the residual variance to the intercept before conversion (Goldberger, 1968).

3. Results

3.1. Forest management scenarios and services provision

The forest area divided into forest management systems for each FC is shown in Figure 5. Overall, the coppice area amounted to 39%, 25% and 62% of the total forest area in scenarios A, B and C, respectively. Coppice in conversion to high forest covered 31%, 38% and 10% of the total forest area in scenarios A, B and C, respectively. High forest amounted to 10% of the total forest area in all scenarios. Unmanaged forests amounted to 20%, 27% and 18% of the total forest area in scenarios A, B and C, respectively.

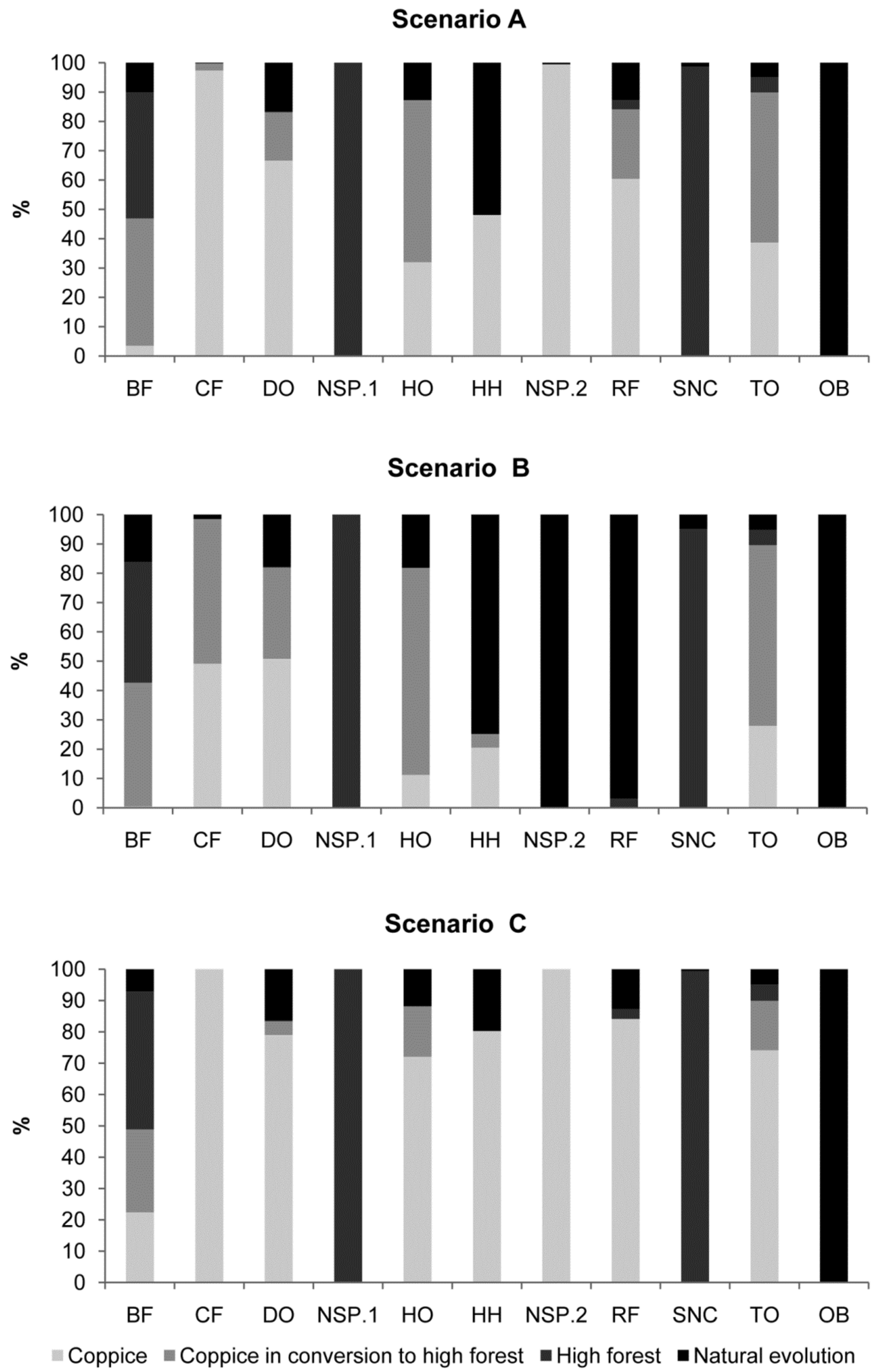

Figure 5. Forest area (expressed as percentage) divided into forest management systems for each FC in scenario A (Business-as-Usual), B (nature conservation), and C (wood production). BF (Beech forest), CF (Chestnut forest), DO (Downy oak forest), NSP.1 (Eucalyptus plantation), HO (Holm oak forest), HH (Hop-hornbeam forest), OB (Other broadleaved forest), NSP.2 (Plantation of False-acacia and ailanthus forest), RF (Riparian forest), SNC (Site native coniferous forest), TO (Turkey oak forest).

Our results show that a total wood production of about 8.5 million m^3 and a total residuals production of about 1.9 million m^3 can be obtained in the next 20 years (2015-2035) when the BaU scenario (scenario A) is considered. When scenario B was considered, wood and residuals production decreased by 28% and 81%, respectively, compared to scenario A. When scenario C was used, wood and removals production increased by 45% and 74%, respectively, compared to scenario A.

Overall, the percentage of the initial growing stock that was removed by forest utilization amounts to 46%, 28%, and 65% for scenarios A, B, and C, respectively. The average value of wood removal (both wood and residuals) computed on the harvested forest area over the 20-year time period amounts to 89 m^3ha^{-1}, 70 m^3ha^{-1}, and 130 m^3ha^{-1} for scenarios A, B, and C, respectively.

Carbon sequestration varied depending on harvest intensity, passing from a net value of 3,306,309 Mg for scenario B to 39,301 Mg for the BaU approach and to a net decrease (carbon source) of 3,435,231 Mg for scenario C. A similar trend was highlighted for the carbon stored by FMUs in 2035, with a total value ranging from 22,982,058 and 16,240,486 Mg for scenarios B and C, respectively, compared to the value estimated for scenario A (19,675,718 Mg).

An example of the spatial distribution of the TESV values computed for each FMU is shown in Figure 6. Scenario A showed positive values of TNPV, except in areas where timber removal is null. Positive values of TNPV are associated with negative values of TSCC to a maximum of -8,000 € ha^{-1} in the TESV. In scenario B, the total benefit supply increased compared to A due to the less intensive forest management approach, therefore in B there were no TESV values lower than -4,000 € ha^{-1}. Scenario C showed an opposite trend compared to scenario B, and the results produced negative values up to -24,000 € ha^{-1}. In addition, in scenarios A and B the TNPV and TSCC supplies were fairly balanced compared to scenario C, since in this latter scenario the increased timber production lead to less total social benefits.

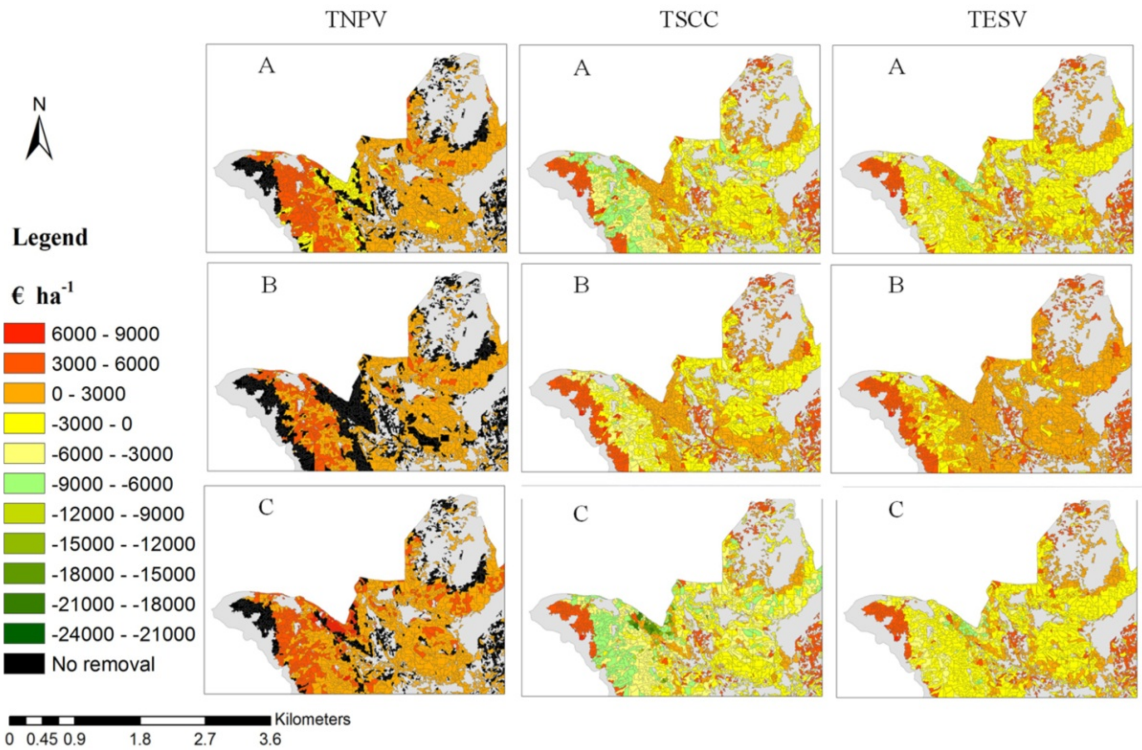

Figure 6. Maps depicting the spatial distribution of Total Net Present Value (TNPV), Total Social Cost of Carbon (TSCC), and Total Ecosystem Services Value (TESV) for scenario A (Business-as-Usual), B (nature conservation), and C (wood production).

3.2 Sensitivity analysis

Table 4 shows the TNPV computed at the regional scale for different scenarios (A, B, and C), discount rates and management systems. With a discount rate of 1%, scenario C has the highest TNPV, which amounts to 322,988,990 € (4,122 € ha^{-1}) for coppices and high forests. Using an 8% discount rate, the TNPV decreased by 48% (167,147,132 € and 2190 € ha^{-1}). In scenario A, the TNPV decreased by 45% going from a discount rate of 1% (226,578,897 € and 2,618 € ha^{-1}) to 8% (125,337,415 € and 1,444 € ha^{-1}). In scenario B, the reduction of the TNPV was smaller (44%) compared to scenarios A and C, ranging between 159,685,452 € (1,857 € ha^{-1}) with a discount rate of 1% and 90,148,659 € (1,038 € ha^{-1}) with a discount rate of 8%.

Figure 7 shows that in all scenarios the TNPV in coppice forests decreased as the discount rate increased. Similar results were obtained for high forest systems (data not shown).

Comparing the different scenarios, our results show that using a 4% discount rate, which falls within the range suggested by Ciancio et al. (2007) and Krieger (2001), scenario C yielded the highest TNPVs. In fact, for coppices the TNPV was equal to 2011 € ha^{-1}, which is 25% greater than in scenario A (1492 € ha^{-1}) and double compared to scenario B (1060 € ha^{-1}). The differences between alternative scenarios are emphasized when high forests were considered. In this case, scenario C produced a mean TNPV of 1062 € ha^{-1}, which is more than double of the TNPV for scenarios A (496 € ha^{-1}) and B (358 € ha^{-1}). The TNPV for the whole region for a 4% discount rate amounted to 171,986,957.59 €, 122,329,564.94 € and 237,823,040.23 € for scenarios A, B and C, respectively.

It is worth noting that increasing discount rates provide different results among scenarios, as indicated in Figure 8 where the percentage variation of economic value for scenario C (the scenario with the highest TNPVs) is plotted against the other two scenarios. In coppice forests, an increasing discount rate produced decreasing percentage variation between scenarios C-B and C-A, while in high forests an increasing discount rate determined an increasing percentage variation between scenarios C-B and C-A. Despite the fact that the lines in Figure 8 indicate slight slopes, the trend observed for high forest encourages selecting scenario C, while the choice becomes uncertain for coppice forest.

The sensitivity analysis was also performed to understand the impact of discount rates on the TSCC and related implications from an economic (market) perspective. Table 5 shows the value of the TSCC computed at the regional scale compared to different scenarios, discount rates and management systems. Overall, scenarios A and C yielded negative TSCC values, while scenario B produced positive values. However, in scenarios A and C, the TSCC increased up to 42% going from a discount rate of 1% to 8%, whereas in scenario B, the TSCC decreased up to 42% going from 1% to 8%.

Figure 9 shows that in scenarios A and C the TSCC in coppice forest increased as the discount rate increased, while in scenario B the trend was reversed. In high forests in all scenarios the TSCC decreased as the discount rate increased (Figure 10).

Using a discount rate of 7%, which is within the range suggested by Nordhaus (2007), it is possible to note that scenario C produced the lowest TSCC values (Table 5). In coppice forests the TSCC amounted to -285,209,073 € (-2587 € ha^{-1}), which is lower than half compared to scenario A (-111,084,094 € and 1007 € ha^{-1}). Contrarily, scenario B showed positive TSCC values, which amounted to 71,411,381 € (647 € ha^{-1}). In the case of high forests, it is possible to note positive values in all scenarios. Scenario B yielded the highest TSCC value (56,672,738 € and 3734 € ha^{-1}). Scenario A showed a value similar to scenario B (42,066,562 € and 2771 € ha^{-1}), which is six-fold the value estimated for scenario C (6,580,110 and 433 € ha^{-1}) using a 7% discount rate.

Figure 11 shows TESVs obtained using a 1% discount rate (with a discount rate >1%, the trend did not change). From this figure it is possible to note that scenario B had the highest TESVs (coppice and high forest), while scenario A had average values, and scenario C negative values.

Finally, if the TNPV and TSCC were considered with a discount rate of 1% (Figure 12), it is possible to note that in scenario A the negative TSCC values were compensated by high TNPVs. On the other hand, in scenario C negative TSCC values were not compensated by positive TNPVs.

Table 4. Total Net Present Value (TNPV, in millions of €) in relation to forest management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production) and discount rates and forest management systems.

Scenario	Management system	Discount rate							
		1%	2%	3%	4%	5%	6%	7%	8%
A	Coppice	216.7	196.9	179.6	164.5	151.2	139.5	129.1	119.9
	High forest	9.9	9.0	8.2	7.5	6.9	6.3	5.8	5.4
B	Coppice	152.5	139.0	127.2	116.9	107.8	99.7	92.6	86.3
	High forest	7.2	6.5	5.9	5.4	5.0	4.6	4.2	3.9
C	Coppice	302.0	271.3	244.7	221.7	201.6	184.1	168.8	155.3
	High forest	21.0	19.2	17.5	16.1	14.9	13.7	12.8	11.9

Table 5. Total Social Cost of Carbon (TSCC, in millions of €) in relation to forest management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production) and discount rates and forest management systems.

Scenario	Management system	Discount rate							
		1%	2%	3%	4%	5%	6%	7%	8%
A	Coppice	-178.7	-163.5	-150.2	-138.6	-128.3	-119.2	-111.1	-104.0
	High forest	67.6	61.9	56.9	52.5	48.6	45.1	42.1	39.3
B	Coppice	114.7	105.0	96.5	89.0	82.4	76.5	71.4	66.8
	High forest	91.1	83.4	76.6	70.7	65.4	60.8	56.7	53.0
C	Coppice	-458.7	-419.7	-385.6	-355.7	-329.3	-306.0	-285.2	-266.9
	High forest	10.6	9.7	8.9	8.2	7.6	7.1	6.6	6.2

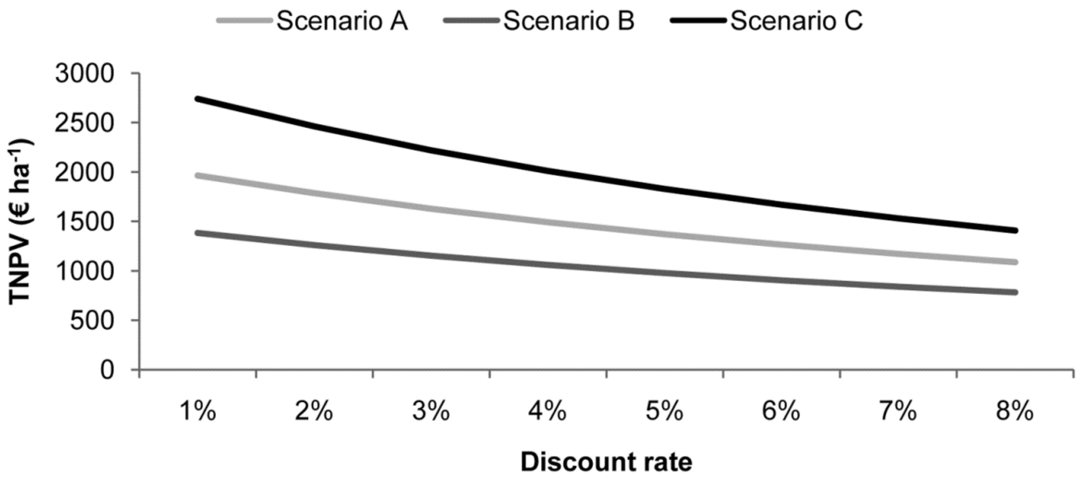

Figure 7. Variation of Total Net Present Value (TNPV, € ha^{-1}) in coppice forest with increasing discount rates.

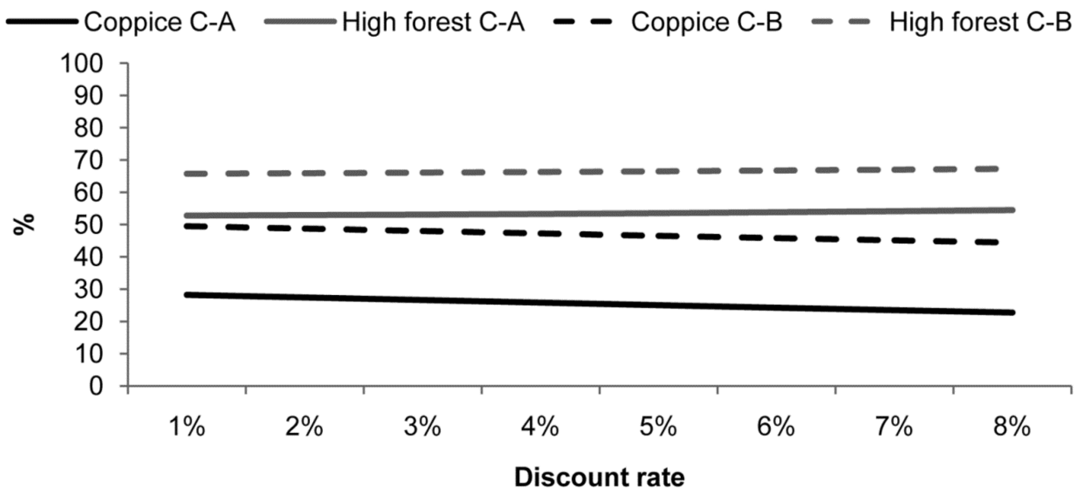

Figure 8. Percentage variation of economic values with increasing discount rates.

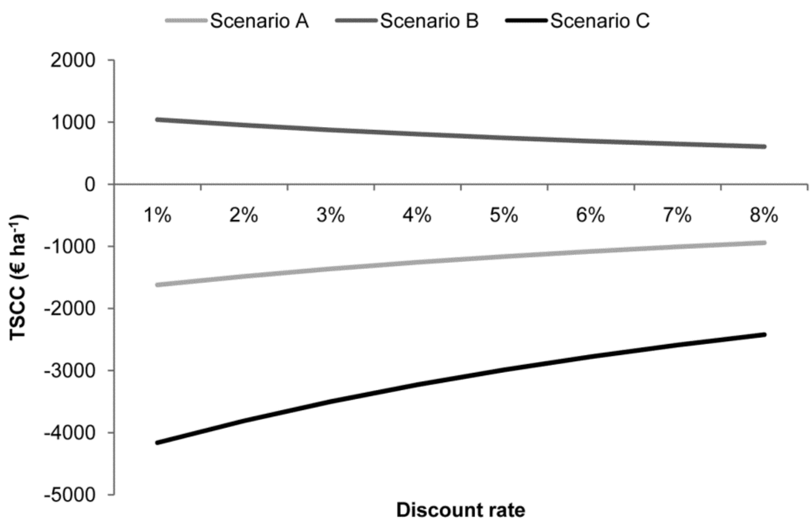

Figure 9. Variation of Total Social Cost of Carbon (TSCC, € ha^{-1}) in coppice forest with increasing discount rates.

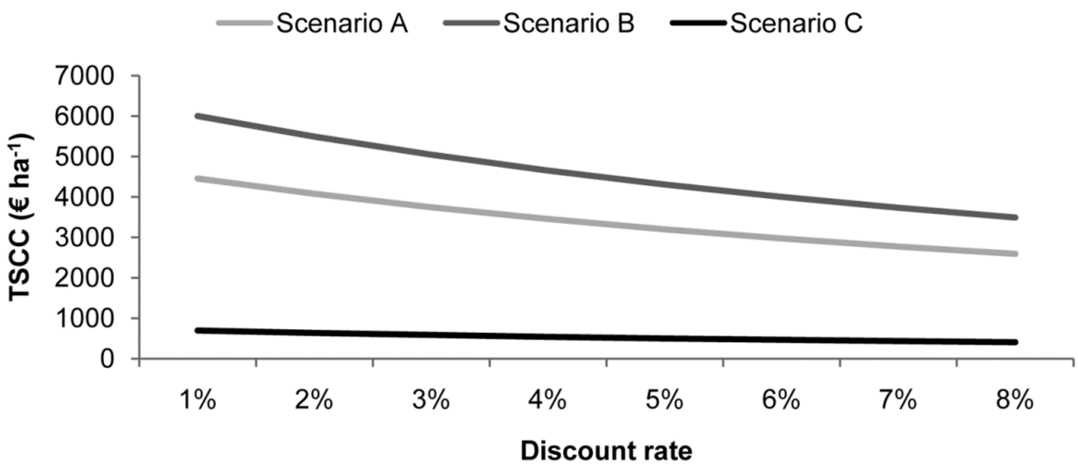

Figure 10. Variation of Total Social Cost of Carbon (TSCC, € ha^{-1}) in high forest with increasing discount rates.

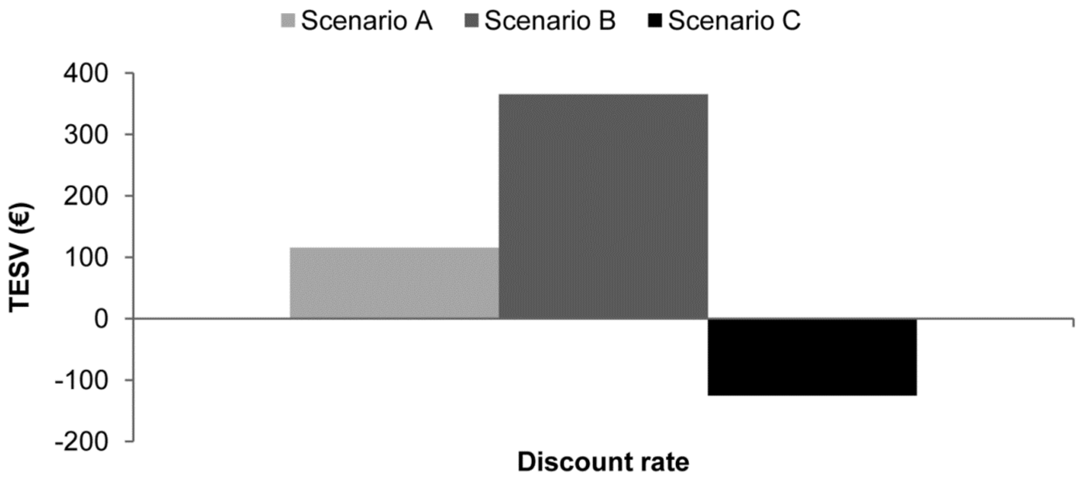

Figure 11. Value of Total Ecosystem Services Value (TESV, in millions of €) using a discount rate of 1%.

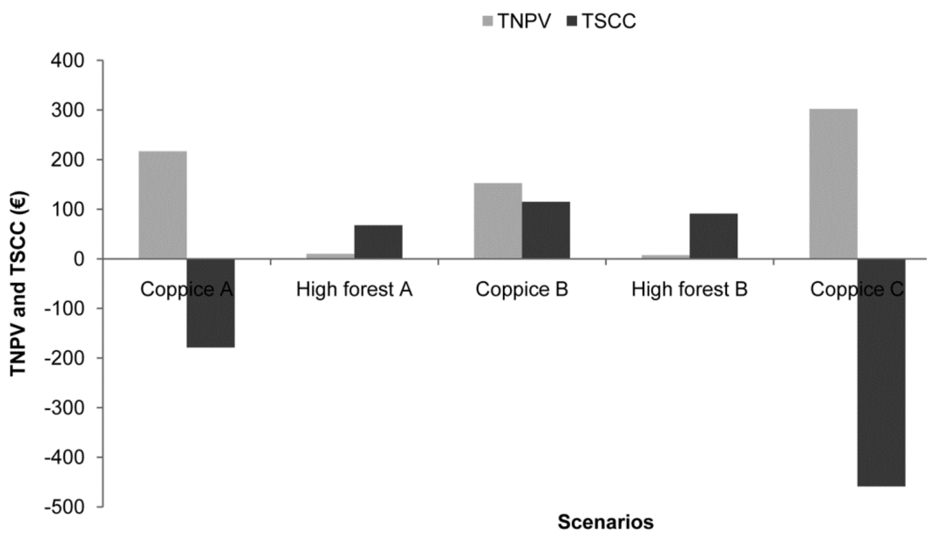

Figure 12. Total Net Present Value (TNPV, in million of €) and Total Social Cost of Carbon (TSCC, in million of €) in forest management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production).

3.3 Trade-off analysis

Our results show that for different forest management scenarios both trade-offs and the TESV changed over the simulation period (20 years). In all scenarios, the trade-offs as a function of timber removed (m^3 ha^{-1}) corresponded to an increasing TNPV, while the TSCC decreased (Figure 13). Thus, the increase of an additional unit of removed wood increased the private benefits of the TNPV, while the social benefits of carbon sequestration (TSCC) decreased.

Figure 13 shows that in scenario A the E value is reached at 50.7 m^3 ha^{-1} of removed timber, which corresponds to an economic value of 685 € ha^{-1}. The E value increased up to 51.5 m^3 ha^{-1} (536 € ha^{-1}) and 58.9 m^3 ha^{-1} (901 € ha^{-1}) in scenarios C and B, respectively.

The second important parameter to be considered is the point where the TESV is equal to zero, which can be considered as a maximum threshold value. Figure 13 shows that each scenario had a different sustainability threshold. Considering scenario A, the threshold of the TESV was reached at 131 m^3 ha^{-1} of removed timber. This threshold increased in scenario B (163 m^3 ha^{-1}) and decreased in scenario C (116 m^3 ha^{-1}) compared to scenario A. Accordingly, the potential range of timber removal that can be considered sustainable was wider in scenario B and narrower in scenario C. However, it is worth noting that these results are based on average data computed for all forests without distinguishing between coppices and high forests.

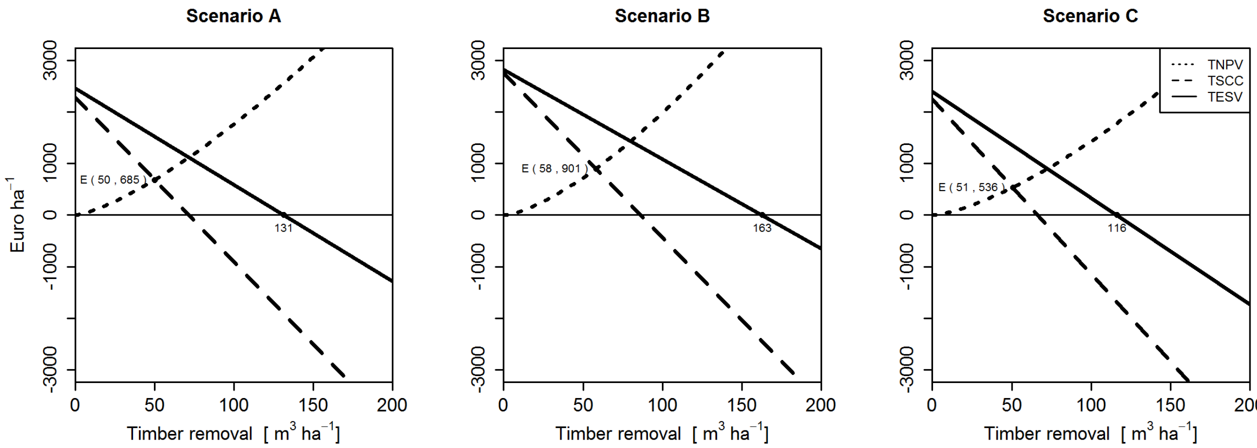

Figure 13. Total Net Present Value (TNPV, in million of €) and Total Social Cost of Carbon (TSCC, in million of €) in forest management scenarios A (Business-as-Usual), B (nature conservation), and C (wood production).

4. Discussion

We developed a spatially explicit model (MiMoSe – Multiscale Mapping of ecoSystem services) to assess how forest management influences the forest potential to provide ES at the regional scale. The model was implemented to assess timber production and carbon sequestration under alternative management scenarios over a 20-year time period. Moreover, for each scenario the trade-offs between the TNPV and TSCC were analyzed, and the average E values between the TNPV (€ ha^{-1}) and TSCC (€ ha^{-1}) were assessed. Herein we discuss the main results concerning ES assessment and their models.

4.1 Management scenarios and service trade-offs

In general, our results reveal that a forest management approach mainly directed at nature conservation and climate change mitigation (scenario B) at the regional scale increases the TESV by approximately 85% in comparison with the BaU approach (scenario A). A forest management approach mainly geared towards maximizing economic incomes from timber production (scenario C) reduces the TESV by approximately 82% compared to scenario A.

The trade-offs analysis showed that by adding units of removable timber the TNPV increases while the TSCC decreases and the E values change, indicating that removing quantities of timber that are higher or lower compared to E values produces an increase of one ES and a decrease of the other. The E values between the TNPV and TSCC in scenarios A and C are similar, with an average of about 51-52 m^3 ha^{-1} of timber removed, whereas the economic values differ (685 € ha^{-1} and 536 € ha^{-1}, respectively). In scenario B, the E value slightly increases to 59 m^3 ha^{-1} of removed timber, with an economic value of 901 € ha^{-1} (Figure 13). More specifically, by removing timber the TNPV increased while the TSCC decreased until equilibrium (E) was reached, indicating that the two ES are at the optimum level, that is the maximization of both private (TNPV) and social (TSCC) benefits. Indeed, while the TNPV represents the economic value associated to forest utilization conducted by a physical or legal authority, the TSCC represents the social benefit associated to the removal of additional quantities of CO_2. In addition, in scenario B the threshold TESV was reached with a quantity of timber removal larger than those of scenarios A and C (Figure 13). This finding indicates that scenario B has the potential of providing more timber compared to scenarios A and C before the ecosystem ceases to provide benefits, that is, the amount of timber removal where the TESV is equal to zero, probably because of a compensation effect due to the unmanaged forest area and a less intensive management approach in scenario B.

Other studies at the regional scale in Europe investigated the impacts of forest management alternatives (e.g., Seidl et al., 2007; Duncker et al., 2012) and land use strategies (e.g., Fürst et al., 2013) on forest ES provision. Results from these studies mainly revealed that by adopting low-intervention forestry, or by implementing nature conservation scenarios, benefits associated with carbon sequestration mostly increase, while those associated with timber production generally decrease over time.

Results from our simulations mainly indicate that forest management, in terms of harvesting intensity and frequency, strongly influences ES provision and associated benefits. In fact, alternative management scenarios generate a certain variability among simulated services. In particular, the results demonstrate that establishing management restrictions, prolonging rotation periods, reducing the harvesting intensity (amount of removals), and adopting close-to-nature forestry interventions increase carbon sequestration (and associated TSCC) and reduce timber production (and associated TNPV) (scenario B in comparison with scenario A), despite similar trends for the TESV (scenarios B and C in comparison with scenario A). This aspect of course depends on FCs, forest management systems, and stand age. The most widespread FCs in the Molise region, namely Turkey oak and Downy oak forests (Vizzarri et al., 2015), positively contribute to the TESV in scenarios A and B and negatively in scenario C (due to higher negative TSCC values) (see Table 6). Other broadleaved and riparian forests, which are less important for wood production in the scenarios under study, positively contribute to the TESV in all scenarios (due to higher positive TSCC values) (see Table 6). In the case of forest management systems, although coppice forests contribute to increasing timber production incomes at the regional scale (TNPVs), high-forest and infra-opened stands positively contribute to the TESV in such a way that the losses of TSCC for coppice forests, especially in scenario A, are compensated (see Table 7). Stand age also influences simulations. Considering that the largest part of forest stands in the Molise region belongs to the 30-60 (i.e. ageing coppice forests) and 60-90 (i.e. mature high forests) age classes, scenario C causes a reduction of the TSCC and subsequently of the TESV mainly in these age classes (see Table 8) in comparison with the current state (BaU). Higher TESVs (due to higher TSCC values) for ageing coppice forests in scenario B in comparison with scenario A are justified by the fact that most of these formations are converted to high forests and thus managed using less intensive approaches.

Table 6. Total Net Present Value (TNPV), Total Social Cost of Carbon (TSCC), Total Ecosystem Services Value (TESV) (in millions of €) for each forest category and management scenario. A (Business-as-Usual), B (nature conservation), C (wood production), BF (Beech forest), CF (Chestnut forest), DO (Downy oak forest), NSP.1 (Eucalyptus plantation), HO (Holm oak forest), HH (Hop-hornbeam forest), OB (Other broadleaved forest), NSP.2 (Plantation of False-acacia and ailanthus forest), RF (Riparian forest), SNC (Site native coniferous forest), TO (Turkey oak forest).

Forest category	Scenario A			Scenario B			Scenario C		
	TNPV	TSCC	TESV	TNPV	TSCC	TESV	TNPV	TSCC	TESV
BF	24.6	-0.8	23.8	19.7	26.5	46.2	34.6	-40.3	-5.7
CF	0.9	-0.4	0.6	0.4	1.1	1.5	1.3	-1.5	-0.2
DO	31.7	-1.8	29.9	20.7	35.2	55.8	41.3	-29.9	11.4
NSP.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
HO	2.7	1.2	3.9	1.8	5.6	7.3	4.0	-3.7	0.3
HH	10.9	4.0	14.9	4.0	19.4	23.5	21.6	-28.6	-7.0
OB	0.0	38.3	38.3	0.0	38.3	38.3	0.0	38.3	38.3
NSP.2	0.9	-1.7	-0.9	0.0	1.5	1.5	0.9	-1.8	-0.9
RF	10.3	4.1	14.4	0.0	40.7	40.8	12.7	-5.8	6.9
SNC	0.3	6.4	6.7	0.2	8.7	8.9	0.3	4.7	5.0
TO	89.8	-46.9	42.9	75.5	22.7	98.2	121.2	-138.5	-17.3
Total	172.0	2.5	174.4	122.3	199.6	321.9	237.8	-207.2	30.6

Table 7. Total Net Present Value (TNPV), Total Social Cost of Carbon (TSCC), Total Ecosystem Services Value (TESV) (in millions of €) for each forest management system and management scenario. A (Business-as-Usual), B (nature conservation), C (wood production), IOS (Infra-opened stand).

Forest management system	Scenario A			Scenario B			Scenario C		
	TNPV	TSCC	TESV	TNPV	TSCC	TESV	TNPV	TSCC	TESV
Coppice	164.4	-111.1	53.4	116.9	71.4	188.3	221.7	-285.2	-63.5
High forest	7.5	42.1	49.6	5.4	56.7	62.1	16.1	6.6	22.7
IOS	0.0	71.5	71.5	0.0	71.5	71.5	0.0	71.5	71.5
Total	172.0	2.5	174.4	122.3	199.6	321.9	237.8	-207.2	30.6

Table 8. Total Net Present Value (TNPV), Total Social Cost of Carbon (TSCC), Total Ecosystem Services Value (TESV) (in millions of €) for each age class and management scenario. A (Business-as-Usual), B (nature conservation), C (wood production).

Age class	Scenario A			Scenario B			Scenario C		
	TNPV	TSCC	TESV	TNPV	TSCC	TESV	TNPV	TSCC	TESV
<30	87.5	56.6	144.1	46.1	176.6	222.6	103.8	-8.7	95.1
30-60	52.5	-30.7	21.8	48.3	19.1	67.5	97.7	-141.1	-43.5
60-90	24.0	-11.0	13.0	21.2	8.7	29.9	28.2	-43.5	-15.3
90-120	6.8	-10.3	-3.6	5.3	-3.0	2.3	6.7	-11.1	-4.4
120-150	0.7	-1.1	-0.4	0.7	-0.8	-0.1	0.8	-1.4	-0.7
>150	0.5	-1.0	-0.5	0.7	-1.0	-0.4	0.6	-1.2	-0.6
Total	172.0	2.5	174.4	122.3	199.6	321.9	237.8	-207.2	30.6

4.2 Model robustness and sensitivity

Although the InVEST model has been used in the Mediterranean context for several purposes (see e.g., Garcìa-Nieto et al., 2013), none of these studies have assessed and mapped the future provision of forest ES at the regional level, either in Europe (e.g., for Switzerland, Grêt-Regamey et al., 2013; for Germany, Frank et al., 2015) or in Italy (e.g., Alpine range, Schirpke et al., 2014 and Häyhä et al., 2015).

Considering the main research outcomes presented herein, the InVEST model was able to simulate alternative management scenarios and related benefits (in terms of ES provided). In comparison with other large-scale simulation models (e.g., ARIES, GISCAME, etc.), InVEST is easier to implement and provides more detailed information for policy-makers. Aside from the original purposes for which it was development (e.g., Nelson et al., 2009), in our case InVEST was forced to correlate management alternatives with forest ES provision, thus unraveling the potentialities for its cross-scale implementation (from FMU to regional scale) in different contexts (i.e. land uses) and for several purposes (i.e. management planning). Our research outcomes (influence of forest management scenarios on ES provision) are consistent with those obtained by other studies implementing InVEST, despite the different contexts (e.g., in Chile, Geneletti, 2013; in Brazil, Garrastazú et al., 2015).

One of the limitations of the InVEST model is the assumption that the amount of harvested timber and the frequency of each harvest period remain constant in each FMU over the considered time-span. This assumption was very restrictive in our case study largely dominated by natural forests rather than by forest plantations. The InVEST model was then partially modified by adopting different harvest intensities and frequencies based on current management practices as implemented in each FMU.

The InVEST model has been extensively used but never applied at the FMU level. Overall, we are confident that the forest simulation model in combination with the regression analysis to develop models that describe the relationship between TNPV, TSCC and TESV for each scenario can provide directions on ES supply under the management strategies analyzed in our study.

Nevertheless, it is important to note that there are considerable uncertainties related to the economic assessment approach (Seidl et al., 2007). The economic indicators chosen to estimate the two services addressed different scales - local for the timber indicator (TNPV) and global for the carbon sequestration indicator (TSCC) (Faber et al., 2002).

The aim of the sensitivity analysis is to guide stakeholders towards an optimal planning decision. Values of timber production and carbon stock, management systems, and market conditions (different discount rates) were implemented in the sensitivity analysis to take into account ecological, environmental and economic issues. Our results (Tables 4 and 5) show that scenario B provides positive TNPV and TSCC values both for coppices and high forests. Whereas, in scenarios A and C, high forests provide positive TNPVs and TSCC values, while coppices provide positive TNPVs and negative TSCC values. The results obtained using discount rates ranging between 1% and 8% show that the TESV is positive but shows a negative trend in scenarios A and B and is negative values but with a positive trend in scenario C. This means that for a discount rate >8% a breakeven point might occur, that is, the discount rate where the TESV becomes positive for scenario C and negative for scenarios A and B. In our simulation, we estimated a breakeven point of 15%, 16%, and 28% for scenarios A, B, and C, respectively. However, these breakeven points are unrealistic. Thus, if present conditions do not change, scenario B is the best management scenario for the provision of the investigated services, as it provides higher TESVs than the values of scenarios A and C for all discount rates in the range of 1% to 8%. However, assuming a change in the current conditions, scenario B might not be the best option.

Conclusions

This study contributes to the ongoing debate on trade-offs and synergies between carbon sequestration and wood production benefits associated with socio-ecological systems. It provides a powerful approach for investigating general relationships between pairs of benefits and between different approaches for integrating services in conservation planning. Through the multi-scale approach for assessing forest ES and related benefits, the study enables decision-makers, stakeholders and landscape planners to better guide management strategies and decisions in the future.

The spatial visualization of ES across the regional area could be useful to identify priority areas for maximizing ES provision and benefits for local communities (i.e., win-win and lose-lose area of intervention). This may assist in differentiating interventions so as to maximize economic incomes over a shorter time period (for coppice forests), or contribute to nature conservation and climate change mitigation (mainly for high forests and infra-opened stands). Moreover, the ES trade-off analysis could provide important information on how to balance economic incomes from alternative services according to management strategies and landscape characteristics. This is particularly useful for forest management (related objectives and purposes), which can be optimized by balancing the TNPV with the TSCC for each FC, individually (i.e. higher and less productive FCs in terms of revenue). To this end, the outcome of this study stresses the critical role of detailed definition and mapping of both forest types and silvicultural systems (e.g., Bottalico et al., 2014); moreover, as distinctively concerns the forest types, the value of shared and integrated typological frameworks (e.g., Barbati et al., 2007, 2014) should be more readily acknowledged.

The approach presented herein (including the implementation of the InVEST model) needs to be further developed by (Nahuelhual et al., 2015): (i) diversifying timber assortments besides fuel wood to improve the effectiveness of forest management alternatives in terms of expected revenues and benefits; (ii) improving data availability (in terms of quality and amount of data) to further enhance the reproducibility of results and the applicability of the tested approaches; (iii) considering the Harvested Wood Products as an additional carbon pool to guide forestry practices towards better mitigation strategies; (iv) including the prediction of land use change and the effects of climate in forest ES assessment to explicitly consider uncertainty in future-oriented landscape simulations; and (v) including the influence of stakeholders (not only forest managers) in forest ES assessment, both at an earlier (scenario design) and later stage (multi-stage model running) of research, thus improving the two-way exchange of knowledge between scientists and local communities.

Despite these limitations, the proposed approach proves to be feasibly applicable to other contexts with similar characteristics. Thus it offers a useful basis for projecting the benefits from forest ecosystems into the future to further improve the effectiveness of management strategies towards the preservation of natural capital from local to regional/national scale, especially in Mediterranean areas where forest ecosystem resilience is severely threatened.

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