Trade-offs and Synergies Between Provisioning and Regulating Ecosystem Services in a Mountain Area in Portugal Affected by Landscape Change

Authors: Sil, Ângelo, Rodrigues, Ana Paula, Carvalho-Santos, Cláudia, Nunes, João Pedro, Honrado, João, et al.

Trade-offs and Synergies Between Provisioning and Regulating Ecosystem Services in a Mountain Area in Portugal Affected by Landscape Change

Ângelo Sil[1], Ana Paula Rodrigues[2], Cláudia Carvalho-Santos[3,4], João Pedro Nunes[3], João Honrado[4], Joaquim Alonso[5], Cristina Marta-Pedroso[6], and João C. Azevedo[1]*

* Corresponding author: jazevedo@ipb.pt

[1] *Mountain Research Centre (CIMO) and School of Agriculture, Polytechnic Institute of Bragança, Campus de Santa Apolónia, 5300-253 Bragança, Portugal*

[2] *Institute for Nature Conservation and Forests (ICNF), Parque Florestal, 5300-000 Bragança, Portugal*

[3] *Centre for Environmental and Marine Studies (CESAM) and Department of Environment and Planning, University of Aveiro, Campus Universitário de Santiago, 3810-193 Aveiro, Portugal*

[4] *Research Centre in Biodiversity and Genetic Resources (InBIO/CBIO) and Department of Biology, Faculty of Sciences University of Porto (FCUP) Campus Agrário de Vairão, 4485-601 Vairão, Portugal*

[5] *School of Agriculture, Polytechnic Institute of Viana do Castelo, 4990-706 Ponte de Lima, Portugal*

[6] *Maretec, Instituto Superior Técnico, University of Lisbon, Avenida Ruvisco Pais 1, 1049-001 Lisbon, Portugal*

This study explored the effects of landscape change on the provision of ecosystem services in a mountain area in northern Portugal, in particular the trade-offs and synergies between services in 2 categories: provisioning and regulating. Services were assessed for 1990 and 2006 and projected for 2020 under 3 scenarios, both biophysically and economically, based on modeling and published and unpublished statistics. We found that landscape changes in the 16-year period under study increased the total supply of ecosystem services, measured both biophysically and monetarily, but that agriculture production dropped dramatically. Both regulating and provisioning services increased in value, but only regulating services increased in biophysical units. Projections under 2 of our 3 scenarios indicated that both types of ecosystem services will continue to increase in both amount and monetary value and will function in synergy, whereas the third scenario predicted a decrease in services and trade-offs between the 2 categories. Because land use has a major impact on ecosystem service supply, an understanding of the changes and trade-offs described in this article can support planning and management, in particular in mountain areas and other regions with limited alternatives for income generation. Our findings suggest that regional development plans should include incentives to maximize regulating and provisioning ecosystem services.

Keywords: Ecosystem services; trade-offs; synergies; regulating; provisioning; abandonment; landscape change.

Introduction

Covering no more than one fifth of the Earth's surface, mountain areas supply essential ecosystem services to half of its population (Körner et al 2005). Globally, mountain ranges rank among the systems supplying the most ecosystem services (Grêt-Regamey et al 2012), a pattern that is also observed at the continental scale such as in Europe (Maes et al 2011). Major ecosystem services provided by mountains include water supply, climate regulation, support for biodiversity (both wild and cultivated), contributions to cultural heritage, and support for tourism and recreation. Mountains are also vulnerable to many drivers of socioenvironmental change, in particular climate and land use change, which can promote severe ecosystem degradation and losses in biodiversity (Thuiller et al 2005). Degradation in mountain areas is often difficult and slow to reverse because of the low intrinsic resilience of these systems, given their terrain- and soil-related constraints (Körner et al 2005).

Mountain landscapes contain a high diversity of ecosystems—natural, seminatural, and cultivated—and a remarkable diversity of economic activities, which have sustained a range of ecosystem services (Körner et al 2005). Many mountain areas have been classified as heritage sites or protected landscapes. These landscapes are also dynamic and continuously affected by drivers such as habitat change and overexploitation, climate change, invasive species, and pollution, which affect the supply of ecosystem services (MEA 2005; Schroter et al

2005). Such drivers of change affect different mountain ranges differently, and effective governance requires careful consideration of these differences.

The study of how changes, especially those related to human activity, affect mountain regions' capacity to supply ecosystem services is highly relevant to science and to sustainable development. One approach to this study is ecosystem service trade-off analysis, which uses both quantitative and qualitative methods to analyze *synergies* (which occur when changes in ecosystem services are synchronous—Li et al 2012) and *trade-offs* (which occur when the provision of one ecosystem service is reduced by the increasing use of another—Rodríguez et al 2006). This approach explores complex social–ecological systems by comparing changes in dynamics among particular components of those systems. It can support management and policy-making, avoiding ecosystem disservices and promoting "winning more and losing less" solutions (Elmqvist et al 2011: 5). At the same time, the approach responds to the societal research need for a description of changes in systems as well as for understanding and predicting trends in processes and their interactions under changing conditions (Future Earth 2014).

The lack of detailed knowledge of relationships between provisioning and regulating ecosystem services has been identified as one of the reasons for the slow application of the concept of ecosystem services in land use planning and local decision-making (Elmqvist et al 2011). Several studies have addressed trade-offs (eg Goldstein et al 2012; Felipe-Lucia et al 2014; Jia et al 2014; Pan et al 2014; Yang and Yang 2014; Häyhä et al 2015; Gonzalez-Redin et al 2016). Although some such studies focusing on mountains exist (eg Pan et al 2014; Häyhä et al 2015; Gonzalez-Redin et al 2016), more are needed (Howe et al 2014).

In Portugal, mountains represent more than 39% of the land area and are home to 26% of the population. They provide essential ecosystem services, including support of biodiversity and many provisioning, regulating, and cultural services (Aguiar et al 2009; Lomba et al 2013; Madureira et al 2013; Carvalho-Santos, Nunes, et al 2016; Carvalho-Santos, Sousa-Silva, et al 2016). Mountains have gone through rapid and significant landscape change over the past decades, mainly driven by depopulation, aging, and land abandonment, affecting the provision of ecosystem services (Moreira and Russo 2007; Aguiar et al 2009; Azevedo et al 2011). The effect of these trends on Portuguese mountain areas has been only partially addressed (eg Lomba et al 2013; Madureira et al 2013).

This study focused on the ways that recent changes have affected the provision of ecosystem services in a mountainous area in the north of Portugal. It analyzed trade-offs and synergies between ecosystem service categories and projected how these relationships may change in the future under different scenarios. With this

study we also aimed to overcome the existing gap in the literature concerning trade-offs and synergies between ecosystem services in mountain areas (Howe et al 2014) and interactions between factors involved in landscape change and their effects on the supply of ecosystem services at different scales (Future Earth 2014).

Study area

This study was conducted in the Sabor River's upper basin (Alto Sabor), a 30,650-ha area in the Bragança district of northeastern Portugal (41.9893–41.7691°N, 6.5747–6.82292°W) (Figure 1). The terrain is complex, with a mountain range in the west (elevation 1486 m), a plateau at an average elevation of 900 m in the east, and a flat depression in the middle (less than 600 m at the basin's outlet). Average annual precipitation varies from 1262 mm at the highest altitude (Montesinho) to 806 mm in the Lombada plateau, and average annual temperature varies from 8.5°C at Montesinho to 12.8°C at Lombada.

The study area is part of Montesinho Natural Park, established in 1979, and of sites established under the European Union's Birds Directive (CEC 1979) and Habitats Directive (CEC 1992). The heterogeneity of the area results in part from a high diversity of land uses, including temporary and permanent crops, meadows and pastureland, agroforestry (*Castanea sativa*), natural forests (*Quercus pyrenaica and Q. rotundifolia)*, riparian forests, forest stands of maritime pine (*Pinus pinaster*), and seminatural areas (shrubland) supporting high species richness.

Administratively, the study area encompasses 6 parishes and parts of 13 others (Figure 1) and includes the city of Bragança (population 24,000). Parishes have undergone rapid socioeconomic change in the last decades, in particular depopulation and partial agricultural abandonment, which have impacted the landscape (Azevedo et al 2011; Pinheiro et al 2014). The Alto Sabor basin exhibited a relatively high degree of change from 1990 to 2006. Excluding the urban parishes of Bragança and parishes only marginally included in the study area, population has decreased from about 4130 inhabitants in 1991 to 3554 in 2001 (–14% in 10 years) and 3104 in 2011 (–13% in 10 years and –25% in 20 years). Migration of residents in rural areas to the city of Bragança partially explains this reduction.

Approach and research framework

We assessed the simultaneous provision of a set of ecosystem services in the Sabor River's upper basin for 2 past years (1990 and 2006) and simulated the provision of the same services, based on 3 scenarios, for a future year (2020). We considered ecosystem services in terms of potential benefits, independently of demand. Despite its limited applicability to policy-making, this approach is useful (Brouwer et al 2013; Spangenberg et al 2014) and is

FIGURE 1 The Alto Sabor study area in Portugal. (Map by Ângelo Sil)
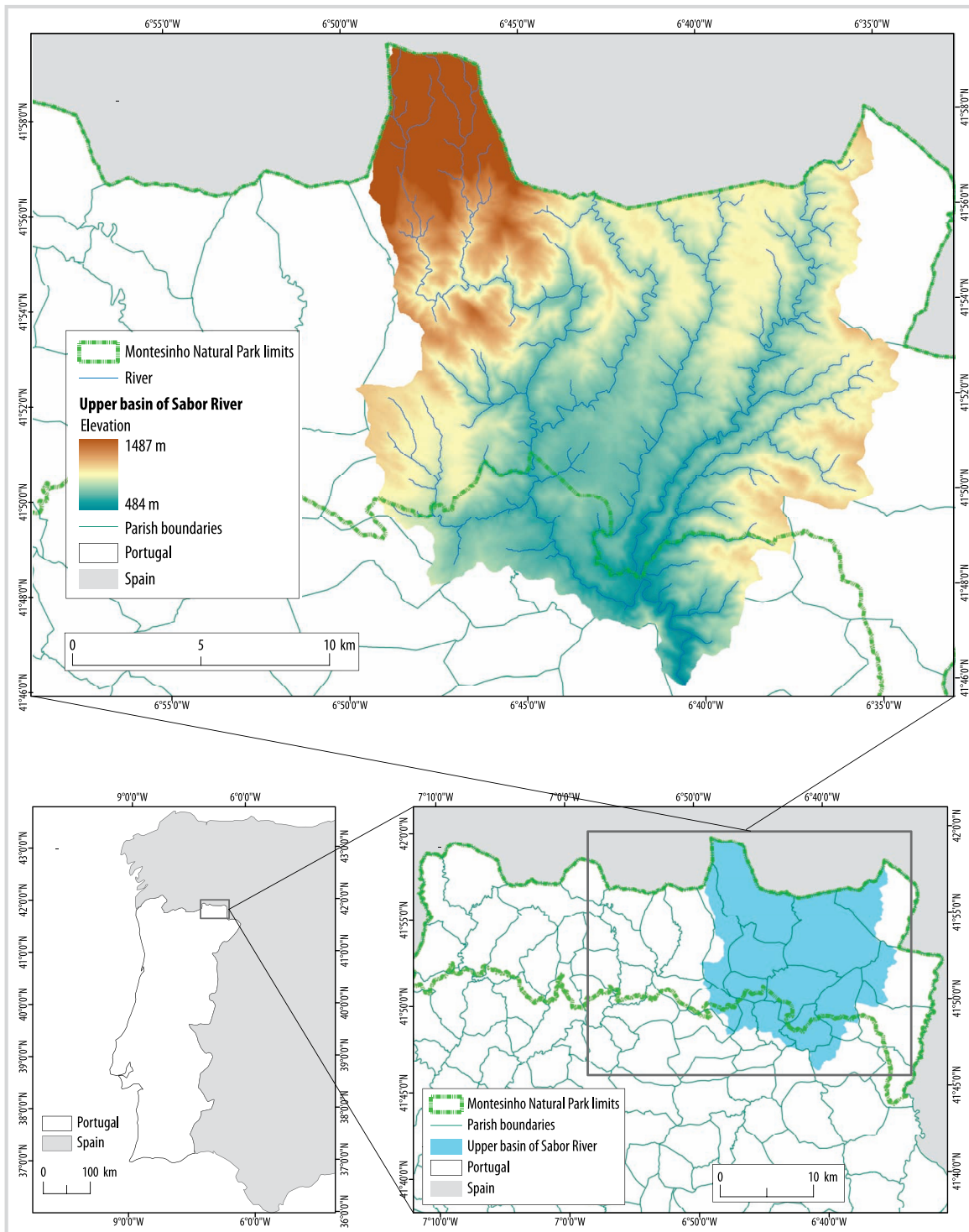
common in research (eg Maes et al 2011; Haines-Young et al 2012; Carvalho-Santos, Nunes, et al 2016). Two remarks regarding limited applicability are important in our context: (1) In the case of one ecosystem service—the provision of agricultural goods—supply is inherently based on demand, and thus it cannot be considered independently of demand; but this does not mean that it cannot be considered in trade-off analysis. (2) Ecosystem

FIGURE 2 Historic and projected LULC in the study area. (Maps by Ângelo Sil)
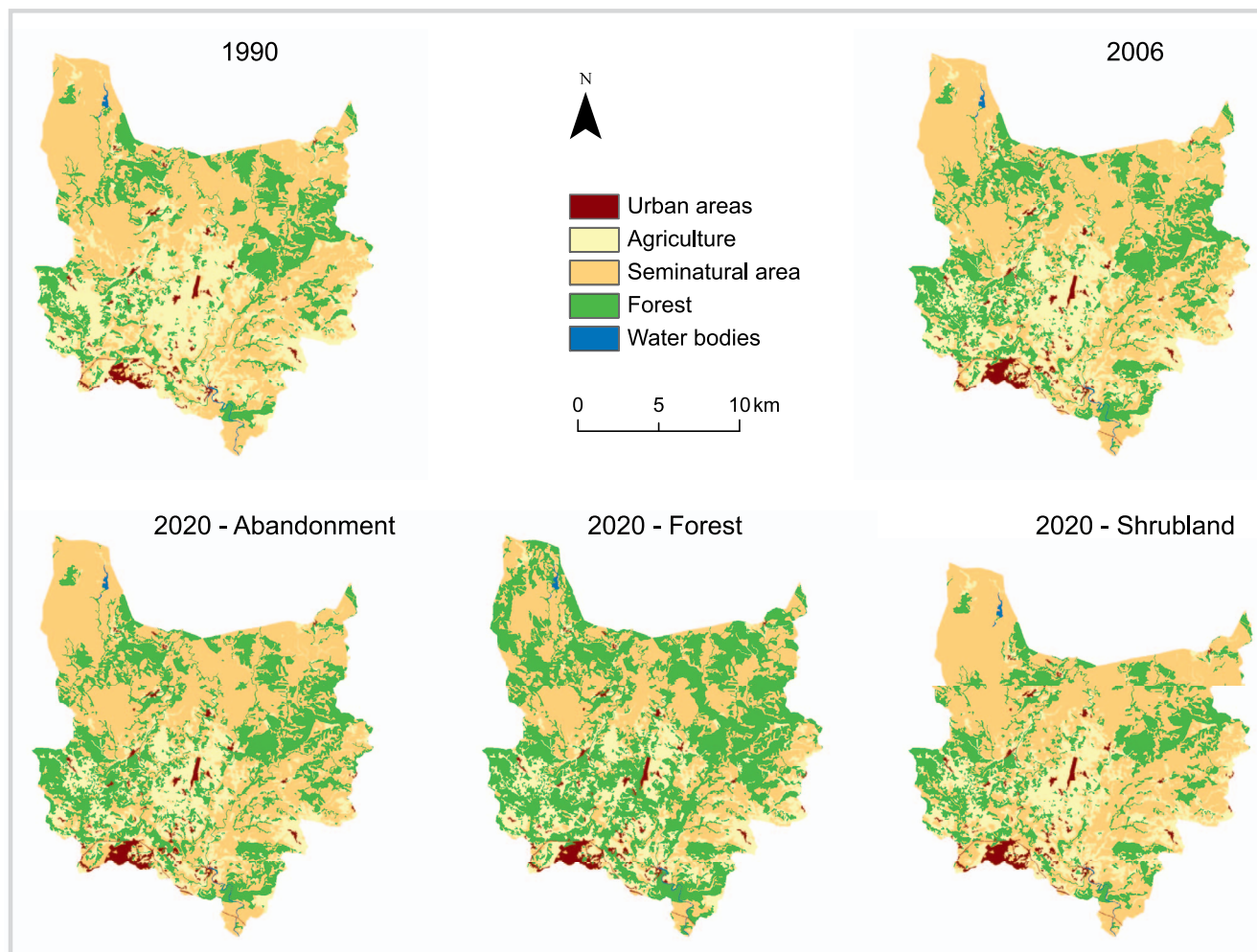
services in future scenarios are not mutually exclusive and their potential provision should be considered only as broad trends.

Trade-off analyses were conducted to evaluate the impact of changes in the landscape on the balance of ecosystem services in the study area. Because there was only 1 study area both for the past and for our scenarios for the future (no replication) and the models used are deterministic (no variability in resulting outputs), the trade-off analyses were done nonstatistically by graphically and numerically comparing provisioning and regulating services, together and separately, in physical and monetary units. We focused on the expected change in landscape structure and associated ecosystem services rather than on the time frame in which changes occur.

Landscape change and scenarios

Vectorial spatial databases (25-m resolution) for land use and land cover (LULC) in the study area were created for 1990 and 2006, following the spatial, thematic, temporal, topologic, and completeness quality requisites of the national land classification system (Carta de Ocupação do Solo) (Carrão et al 2008), based on interpretation of orthophotomaps (Amorim 2015). For 1990 and 2006, LULC maps included 4 hierarchical levels with 70 and 76 unique classes, respectively, for the most detailed level.

Scenarios for 2020 were based on potential trends described for the study area. Scenario maps were created by randomly reclassifying the 2006 LULC classes that are expected to change, using conversion rates estimated in previous research in the study area (Azevedo et al 2011; Pinheiro et al 2014). The resulting maps for the 3 scenarios considered are shown along with the 1990 and 2006 LULC maps in Figure 2.

- The *expansion of forest areas (forest)* scenario represents natural regeneration as well as planned afforestation and forest conservation on formerly seminatural areas and abandoned agricultural areas (conversion of 11% of

TABLE 1 Ecosystem services assessed in the study.

Service class	Service	Indicator	Unit	Description	Biophysical assessment method	Economic assessment method
Regulating	Climate regulation	Carbon sequestration	Gg	Average annual amount of carbon sequestered in the landscape (vegetation, litter, and soil)	Modeling	Avoided cost
	Water quality regulation	Nitrogen retention	Mg	Average annual retention of nitrogen in the watershed, calculated by subtracting simulated nitrogen exported in each date/scenario from maximum simulated export; averages refer to a 10-year period of simulations	Modeling	Unit value transfer
	Erosion regulation	Sediment retention	Mg	Average annual sediment retention in the watershed, calculated by subtracting simulated sediment exported in each date/scenario from maximum simulated export; averages refer to a 10-year period of simulations	Modeling	Unit value transfer
Provisioning	Agricultural goods	Production of agricultural goods	Gg	Annual production of agricultural goods in the study area (including animals and plant crops)	Statistics	Market prices
	Mushrooms	Production of mushrooms	Gg	Potential annual production of wild mushrooms in the study area based on per habitat productivity	Statistics	Market prices
	Timber	Harvested/harvestable timber	dam^3	Timber harvested (1990 and 2006) or harvestable (2020) in the study area annually	Statistics	Market prices
	Water supply	Water discharge	hm^3	Average annual water yield at the outlet of the watershed averaged over 10 years	Modeling	Market prices

agricultural area to forest and 24% of shrubland to forest).

- The *abandonment of agricultural areas (abandonment)* scenario represents a continuation of the recent decrease in agricultural areas due to depopulation and rural abandonment and their replacement by shrubland and forest (conversion of 11% of agriculture to forest and 14% of agriculture to shrubland).
- The *expansion of shrubland (shrubland)* scenario represents shrubland replacing abandoned agricultural areas and forest areas disturbed by fire (conversion of 14% of agriculture to shrubland and 17% of forest to shrubland).

Assessment of ecosystem services

The 2 ecosystem service categories that were the focus of this study, regulation and provisioning, were broken down into specific services and their indicators, as shown in

Table 1. Services and indicators were chosen by the research team based on the Driver-Pressure-State-Impact-Response framework (Gabrielson and Bosch 2003).

Biophysical assessment of ecosystem services supplied in 1990, 2006, and 2020 were based on modeling (eg hydrology and forest growth and yield) and on published and unpublished statistics (agriculture, markets, and harvesting permits) (Table 1). In some cases, estimates were made by multiplying production values by an observed or predicted LULC area.

Ecosystem services were added up in regulation and provisioning classes and compared after normalization of outcomes in physical units using the expression

$$\text{ES score} = (X - X\text{min})/(X\text{max} - X\text{min})$$

where X is the value of a particular service in a particular year and Xmin and Xmax are minimum and maximum values, respectively, within the range of observations (Maes et al 2011).

The monetary value of each ecosystem service was calculated based on market prices (for provisioning services) and on avoided cost and unit value transfer (for regulating services). Monetary values were adjusted using 2014 as the base year, based on the Portuguese consumer price index (INE 2015), and converted to US dollars.

Regulating services

Climate regulation: The carbon storage and sequestration module (Tallis et al 2013) of the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) model (Sharp et al 2015) was used to estimate average annual carbon sequestration in the landscape and its corresponding monetary value. Carbon content was estimated for 4 pools (aboveground and belowground biomass, soil, and litter) using published data from the area and other comparable areas (eg Silva et al 2006; Ramos 2008; Fonseca and Figueiredo 2012; Pinheiro et al 2014) and a local forest growth and yield model (Pérez-Rodríguez et al 2015). To allow estimation of sequestration for 1990, a LULC map for 1970 was built based on the finding that no coniferous forest existed in the area at this date. The InVEST carbon module assumes that carbon pools are in a steady state. However, because biomass growth in forest LULC classes is a relevant process in the study area, we integrated forest growth in the modeling process by subdividing LULC classes according to tree age in a much finer carbon pools matrix. Carbon stocks in the landscape scenarios for 2020 were estimated using 2006 carbon content averages for agricultural and seminatural areas and growth and yield data.

For economic valuation, we used an avoidance of damage costs approach (Pascual et al 2010), applying the value of US\$ 43 per Mg of carbon sequestered (Tol 2005) for the social costs of carbon, a market discount rate of 1.4% (Stern 2007), and an annual rate of change in the price of carbon of 5% (Nelson et al 2009).

Hydrological regulation: Water quality and soil erosion regulation services were derived using the physically based semidistributed Soil and Water Assessment Tool (SWAT) model (Arnold et al 1998; Neitsch et al 2011). The model used homogeneous simulation units called hydrologic response units, each consisting of a unique combination of land use, soil type, and slope (Gassman et al 2007). SWAT uses elevation, LULC, and soil data to parameterize the watershed and daily climate data to run the model (precipitation, maximum and minimum temperature, solar radiation, relative humidity, and wind speed). Detailed description of input data and model setup for the Alto Sabor watershed can be found in Carvalho-Santos, Monteiro et al (2016). The headwaters of the Sabor watershed in Spain were considered in the modeling.

To reduce uncertainty in hydrological model predictions, parameters were calibrated against observed values. Here, a split-sample approach was applied for daily discharge, using part of the dataset for calibration (1998–2007) and an independent dataset for validation (1987–1996), without further parameter adjustment (Figure 3). Discharge predictions can be considered good taking into account the concurrence of simulated and observed values. Suspended sediments and nitrates (NO_3) were also calibrated against field observations, but the lack of observed data prevented the validation exercise. Model predictions were considered adequate despite the limited amount of observed data, as 3 years of measured data are considered sufficient for representing climatic and hydrological variability.

Additionally, simulations using different LULC maps describing the 3 scenarios under analysis were carried out and compared to the area in 2006 (calibration) and 1990 (validation). The SWAT outputs used for the biophysical quantification of hydrological services were nitrogen export (kg/ha/y) for water quality and sediment export (Mg/ha/y) for soil erosion regulation. Values were calculated for the entire basin (40,300 ha). The services were considered as the benefits of retaining nitrogen and sediment and were calculated as differences between scenarios and the baseline value of 1990 (the year with the highest sediment and nitrogen exports).

The benefit transfer method—the application of a monetary value estimated in a location to a similar site (Plummer 2009)—was used in the economic valuation of ecosystem services. For soil retention, the value of € 4.75/Mg, estimated by Marta-Pedroso et al (2007), was used as a proxy for the benefit of soil retention. The final value was calculated for sediment retention in the entire watershed.

Given the complexity of assessing and valuing water-quality-related services (Keeler et al 2012), multiple criteria should be taken into account when conducting a valuation study, including the specific goals of the study and the availability of data and resources (Wang et al 2011). Based on the biophysical outputs of the SWAT model, we estimated nitrogen retention across land use scenarios using the unit value transfer approach (Brander 2013). Hernández-Sancho et al (2010) estimated the environmental benefit (avoided cost) associated with undischarged pollution of nitrogen as ranging from € 4.6 to 65.2/kg and € 16.35/kg when rivers were the destination of pollutants. This value is a proxy for the environmental benefit generated by water quality regulation within the basin and should be regarded as the minimum value of the benefit. The unit value (€ 16.35/kg) was multiplied by the change units (kg of nitrogen retention) in the basin for each assessment date.

Provisioning services

Agricultural products: Agriculture production was calculated based on national statistics, reported at the parish level (INE 1989, 2009), and productivity data from several sources (Rodrigues 2015). It included permanent crops (orchards and vineyards), temporary crops (cereals

FIGURE 3 Observed and simulated discharge rates and nitrate and sediment export, after parameter calibration.
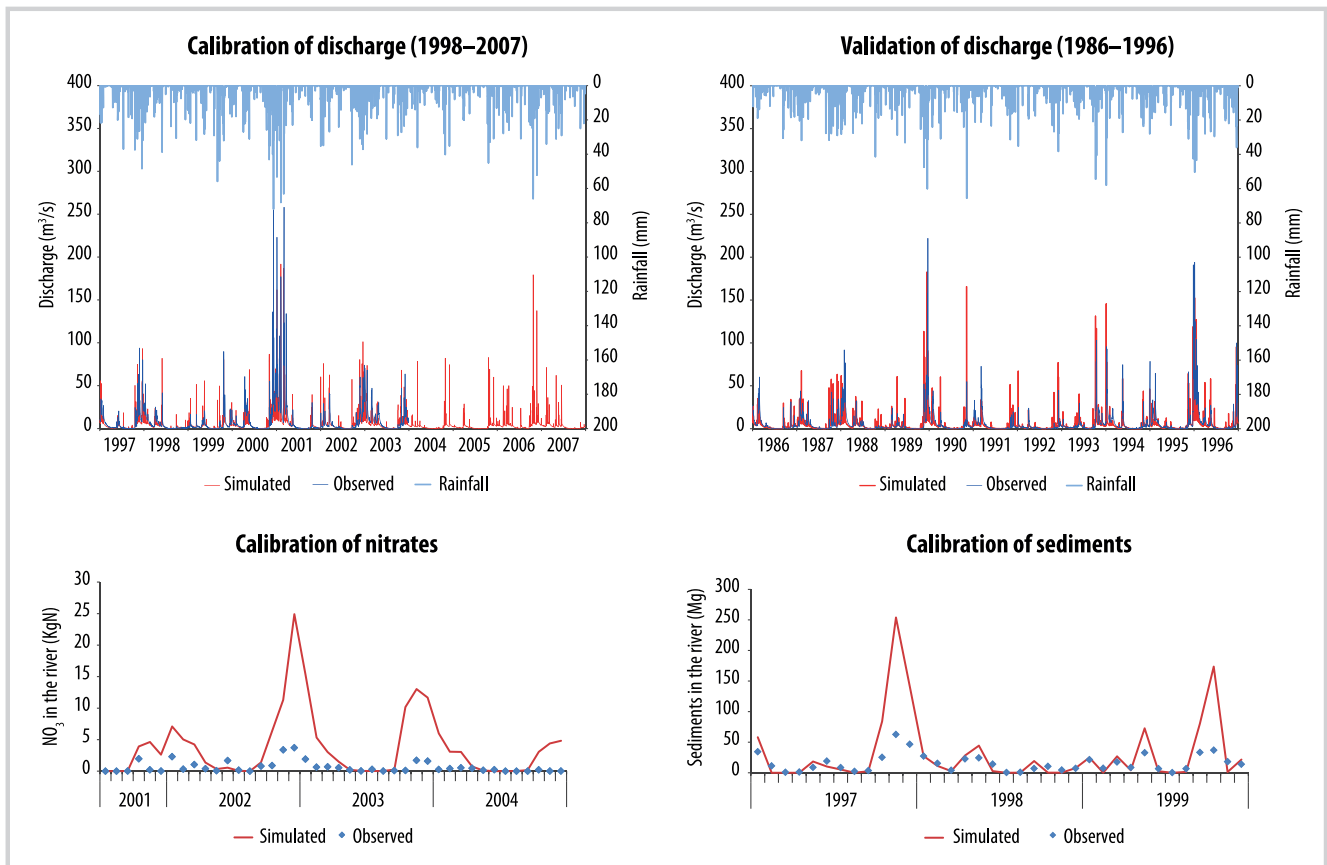
and legumes, forage, potatoes, industrial and horticultural crops), and pastureland, as well as livestock production (milk and meat) and honey and beeswax.

Annual production in Mg was calculated for each parish by multiplying areas or existing stock by production values. Parish-level production data were adjusted based on the percentage of the parish within the study area. Monetary values were calculated based on production values in the study area (GPP 2011). These express the value of primary and secondary products in a region, derived by multiplying production by sale prices at the farm. These values are based on average agricultural activity in a particular region at a particular time and do not include subsidies and taxes. We used 2005–2009 prices (GPP 2011).

Mushrooms: Production of wild mushrooms was estimated per ha of mushroom habitat per year for each of the species known to be collected for home consumption or commercial sale in the region (Garcia et al 2006). Estimates were based on LULC coverages for 1990, 2006, and 2020 (projected) and on productivity data reported for the area (Rodrigues 2015). Productivity expresses the potential supply of mushrooms in each habitat, independently of collection. Economic valuation was done

by multiplying production per species and habitat by average market prices reported for the northeast of Portugal (Garcia et al 2006), including the study area.

Timber: Timber was estimated based on the volume of harvested or harvestable softwood available in each year according to field data and forest stand growth, yield projections, and data on harvests that took place in communal areas managed by the Forest Service (Instituto da Conservação da Natureza e das Florestas) within the study area. Data provided by Instituto da Conservação da Natureza e das Florestas (available from the corresponding author) on the volume of softwoods harvested in the 2 forest perimeters in the area (Deilão and Serra de Montesinho) allowed the estimation of the volume of timber harvested in 2006. We found no data on harvests for 1990; considering that softwood stands in the area in 1990 were mostly younger than 20 years and small in diameter (<14 cm), we assumed the 1990 timber harvest to be null. Timber was estimated for the 3 scenarios for 2020 based on harvestable volume.

The monetary value of timber in 2006 was obtained directly from the revenue of the public sale of timber (internal data from Instituto da Conservação da Natureza e das Florestas), whereas the value for 2020 was projected

TABLE 2 Historic and projected ecosystem services in the study area in biophysical units.

Service class	Service indicator	1990	2006	2020 Abandonment	2020 Forest	2020 Shrubland
Regulating	Carbon sequestration (Gg/y)	5.93	20.56	27.19	38.07	14.12
	Nitrogen retention (Mg/y)	0.00	23.78	26.20	13.70	15.72
	Sediment retention (Mg/y)	0.00	2055.30	2498.60	2675.92	2675.92
Provisioning	Water supply (hm^3/y)	146.75	170.96	172.16	183.50	172.16
	Agricultural production (Gg/y)	41.17	27.69	20.77	24.65	23.81
	Mushroom production (Gg/y)	485.94	483.74	515.24	588.45	474.68
	Timber production (dam^3/y)	0	4.88	345.67	345.67	298.88

by multiplying the volume of harvestable timber (≥25 cm diameter) by 0.5, representing the percentage of wood that can be used in sawmills (Rodríguez Soalleiro et al 1997), and by the average timber market price per diameter class (Duro 2008).

Water supply: Freshwater supply was modeled with SWAT, as described above (hydrological regulation services), and measured as annual average river discharge in m^3, which refers to all the water that is exported downstream and available in the watershed for domestic and industrial use (agricultural use was already integrated in the modeling exercise). To estimate the value of the freshwater supply, we assumed a constant water price of € 0.46/m^3, the price of water for consumers in the municipality of Bragança (CMB 2009), and assumed that the price increase would be negligible until 2020.

Results

Past changes in ecosystem services

There were considerable changes in the provision of ecosystem services in the study area during 1990–2006 (Table 2; Figure 4). Normalized values for regulating ecosystem services as a whole increased from 1990 to 2006, as did those for each service viewed separately. Carbon sequestration per year increased by a factor of 3.5. Water-quality-regulating services all depart from 0, the reference value adopted for nitrogen and sediment retention. Total provisioning services from 1990 to 2006 remained relatively constant. Among particular provisioning ecosystem services, agricultural production decreased by nearly 33%, and water supply increased by 16.5%. Timber production occurred only in 2006 and mushroom production decreased slightly (0.45%).

Projected trends in ecosystem services

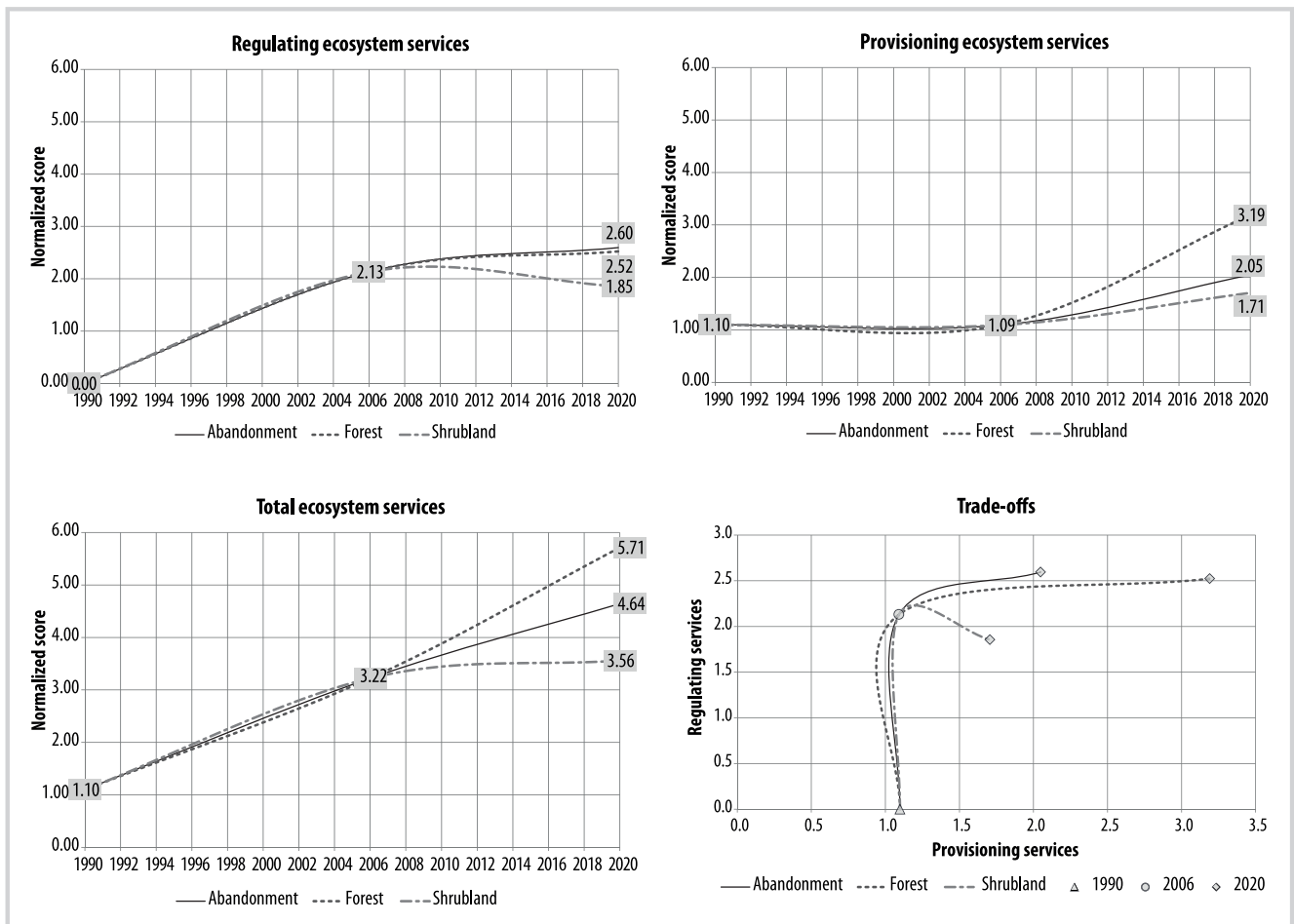No change was projected by 2020 in the supply of regulating services under the forest and abandonment scenarios, but a decrease was projected under the shrubland scenario. Provisioning services increased under all scenarios, in particular the forest scenario. Changes observed from 1990 to 2006 were projected to persist under the forest and abandonment scenarios. The highest ecosystem service score (5.71 out of 7) was reached under the forest scenario. Under the shrubland scenario, ecosystem services were projected to maintain almost the same level in 2020 as in 2006. A trade-off appeared between regulating and provisioning ecosystem services under the shrubland scenario, but the relationship between these 2 categories was synergic under the remaining scenarios. Historic and projected trends measured in biophysical terms are summarized in Figure 4.

Monetary value of ecosystem services

From 1990 to 2006 there was an increase in the monetary value of both regulating and provisioning services in the study area (Table 3). Provisioning services had a monetary value much higher than regulating services in any of the years considered, which resulted from the value of the water supply. As occurred in the biophysical measurement, the monetary value of agriculture dropped significantly (46%) during this time.

By 2020, the monetary value of regulating services is projected to increase under the forest and abandonment scenarios but to decrease to lower than 2006 levels under the shrubland scenario. The monetary value of provisioning services is projected to increase at a rate above that observed for 1990–2006 under all 3 scenarios and to be highest under the forest scenario at almost US$ 116 million. Given the weight of provisioning services in the overall value, the trend for total value of services was similar to that for value of provisioning services. There was synergy between the monetary value of regulating and provisioning ecosystem services from 1990 to 2006. The same type of relationship was projected between 2006 and 2020 for the abandonment and forest scenarios. There was, however, a trade-off between the 2 categories in the

FIGURE 4 Ecosystem services and trade-offs, historic and projected—normalized scores derived from biophysical units.

Discussion

Historic changes

The Alto Sabor basin experienced significant changes over a short period of time (16 years)—not only in the landscape structure but also in ecosystem services. Forests increased 20.7% and agriculture decreased 13.8% in area. Seminatural habitats (shrubland), the most abundant class in the area at all times during the study period, decreased 3.3%. There was a general increase in landscape heterogeneity (larger edge extension, a higher number of patches, higher landscape diversity, smaller patch size, shorter distances between patches, and less dominance of the largest patches). Overall, landscape changes in the area increased the supply of ecosystem services, measured both biophysically and economically.

Agriculture production, although not as important as in other areas covered in the literature (it represented just

same period under the shrubland scenario. Historic and projected monetary values are summarized in Figure 5.

2.6 to 8.4% of the total value of the ecosystem services in the 2 years considered), dropped dramatically from 1990 to 2006, when all other ecosystem services increased in supply and value. This trade-off has occurred in many mountain regions of Europe, where agricultural abandonment positively affected the supply of other ecosystem services (eg Schroter et al 2005; EEA 2010; Maes et al 2011; Haines-Young et al 2012; Briner et al 2013). Other provisioning ecosystem services either increased or remained stable.

At the ecosystem service category level, the change in the supply of provisioning services showed synergy with the change in the supply of regulating services when measured in monetary units, but not when measured in biophysical units. This was caused by an increasing supply of water—which (together with timber) counterbalanced losses in agricultural production, but had a much stronger effect on monetary value than on biophysical totals. This is partially explained by the methodology followed in this study, in which water supply was measured as the capacity of the watershed to supply water. The small variation of water supply across different landscape scenarios in 2020,

TABLE 3 Historic and projected ecosystem services in the study area in monetary value.

Service class	Service indicator	Value (millions of US $/y)				
		1990	2006	2020		
				Abandonment	Forest	Shrubland
Regulating	Carbon sequestration	0.17	0.63	0.87	1.22	0.45
	Nitrogen retention	0.00	0.43	0.48	0.25	0.29
	Sediment retention	0.00	0.01	0.01	0.01	0.08
	Total	*0.17*	*1.07*	*1.36*	*1.48*	*0.82*
Provisioning	Water supply	74.89	87.25	87.87	93.65	87.72
	Agricultural production	7.13	3.87	2.91	3.45	3.33
	Mushroom production	2.91	2.90	3.09	3.46	2.87
	Timber production	0.00	0.10	15.39	15.39	13.43
	Total	*84.93*	*94.13*	*109.25*	*115.96*	*107.35*
Total		85.10	95.20	110.61	117.43	108.17

FIGURE 5 Ecosystem services and trade-offs, historic and projected—monetary value.
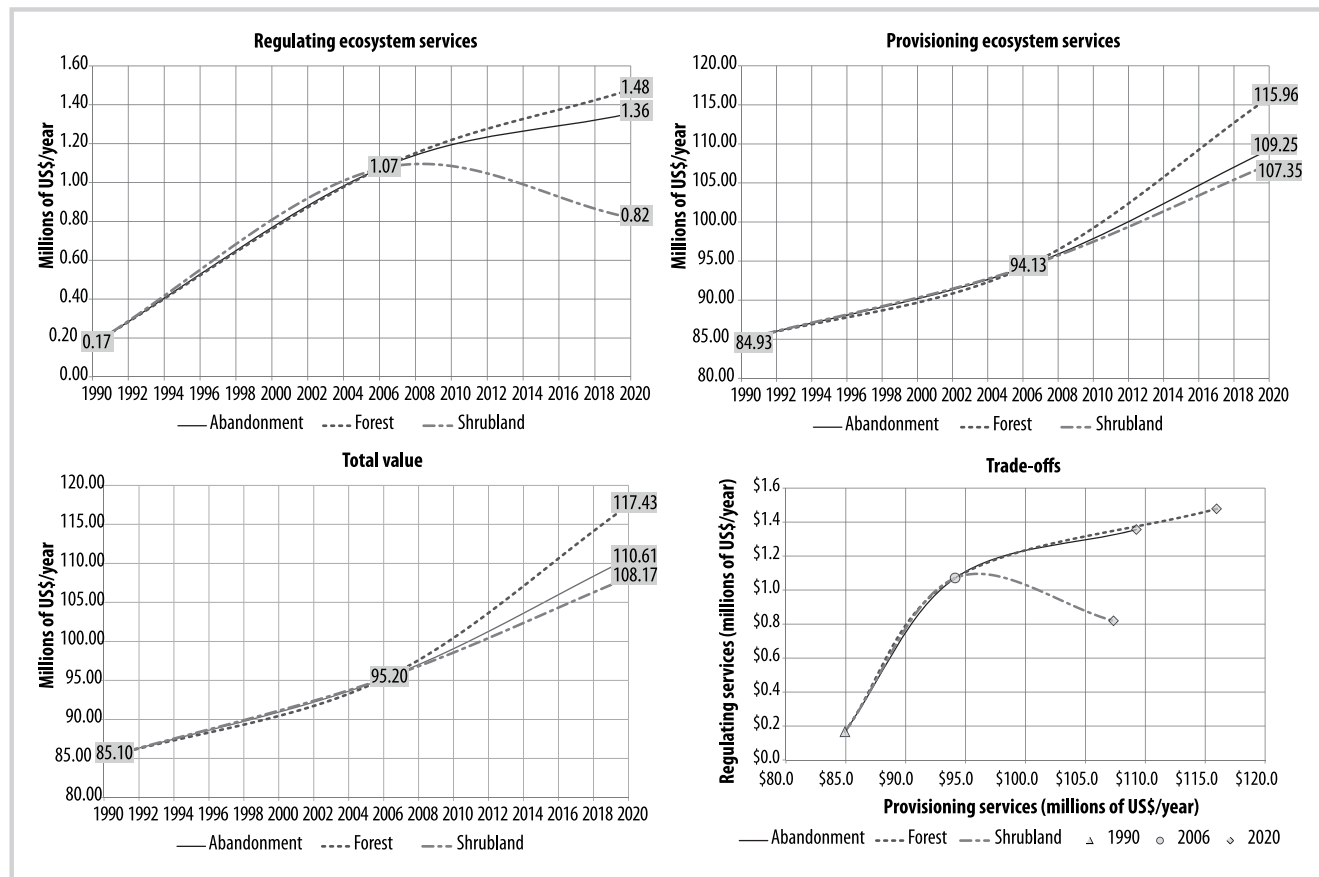
however, indicates that climate can play a larger role than land use in the supply of this ecosystem service, a pattern that has also been observed for similar landscapes and attributed to local climatic and soil conditions constraining water retention by any landscape (Hawtree et al 2015).

Overall, provisioning services showed a much higher value than regulating services. As mentioned above, this results from the method applied. This situation was also observed in other studies where, despite stakeholders giving higher importance to regulating services, these held lower monetary value than provisioning services (Martín-López et al 2014).

Projected changes

Our results suggest that the potential supply of ecosystem services in this area will increase by 2020, especially if forests become dominant, resulting from synergy between regulating and provisioning services. Only in the case of dominance by shrublands is a trade-off between regulating and provisioning services expected, because of a decrease in carbon sequestration and nitrogen retention. The changes in agriculture production from 1990 to 2006, and expected future changes, indicate a trend toward the reduction of the capacity of farmland to produce revenue in the area (Acs et al 2010).

Forests are potential sources of income in the area. They already regularly provide high quantities of mushrooms with well-established markets that generate income for families (Garcia et al 2006). Forestry is likely to become an attractive economic alternative in the near future, when many trees will reach harvestable sizes. Considering the expected effects of climate change and carbon fertilization in forest productivity (Chen et al 2014; Nunes et al 2015), forestry is likely to become a major economic activity in the region. These trends, although uncertain, indicate that traditional farming and human presence in the landscape, and the corresponding management systems and landscape patterns, might be strongly affected by ongoing changes.

Another likely trend in the area is toward a seminatural condition often described as rewilding (Navarro and Pereira 2012). Under this trend, well represented in all of our scenarios for 2020, the area is expected to undergo changes in the composition of plant and animal communities (Lomba et al 2013) as well as in ecological processes such as wildfires (Azevedo et al 2011) and biological invasions fostered by climate change (eg Vicente et al 2013).

Policy relevance of ecosystem service assessment

From decision-making and management perspectives (eg land planning, natural resource management, nature conservation, agriculture, and forest management), the results of our study are particularly relevant for the sustainable development of the area and for avoiding degradation of ecosystems and the landscape. First, the trends observed can be used to establish baseline conditions and expected outcomes for the area. Any development plan or strategy for the area should, therefore, be tested against these references through cost–benefit or other comparative analysis. Second, the projections for 2020 represent likely conditions whose effects on the provision of ecosystem services have been estimated. For example, based on the ecosystem service assessment in this work, the shrubland scenario seems to be less desirable for the area. In addition, shrubland is very prone to fire (Barros and Pereira 2014). Processes that promote expansion of shrubland, such as fire, should therefore receive particular attention from decision-makers, and processes that favor expansion of forests should be promoted instead.

Our study is of particular interest for planning and management of the Parque Natural de Montesinho, a protected area intended to preserve a rich and diverse landscape with moderate levels of human disturbance. Currently, park management is based on a plan established in 2008 (ICN 2007), largely without considering ongoing landscape change processes. Our study demonstrated that landscape change is an important process in the area with likely effects on the supply of different types of ecosystem services. Because the area is currently evolving toward a condition that is very different from the target condition of the natural park assumed in the plan (a complex landscape mosaic driven by human disturbance), park authorities should revisit their plan and management goals, methods, and scales, incorporating causes, trends, magnitudes, spatial variability, and consequences of change, among other components.

The results of this research may also be relevant to rural development policy. Increasing the supply and monetary value of ecosystem services in the Alto Sabor will not necessarily generate revenue for households. In 1990–2006, there was in fact a reduction of income in the area. The value of ecosystem services, however, is very high and is likely to increase in the near future as a result of landscape change. The value of potential ecosystem services in 2006 was almost US$ 3100/ha. Even if demand criteria were applied in valuing water supply, the highest valued of all ecosystem services analyzed here, the monetary value of ecosystem services would still be high. Water generates revenue at the municipal level, the scale at which surface water is managed in Portugal, but not at the estate level—landowners do not receive any income related to water supply. The high value of ecosystem services and the lack of other sources of income in the area suggest that the development and implementation of schemes for payment for ecosystem services could be an interesting way to deal with depopulation, aging, and abandonment in the area. This mechanism should, therefore, based on stakeholders' perceptions and expectations, support regional development policies. As

highlighted in the introduction, many of the ecosystem services provided in mountain areas are in demand elsewhere, and this should be taken into account in future watershed management.

Conclusion

The trade-offs and synergies between ecosystem service delivery under different land use scenarios were analyzed by considering past (1990 and 2006) and projected future (2020) land use scenarios for the Alto Sabor basin in Portugal. Land use change is a major driver of shifts in ecosystem service supply, and hence depicting such changes and associated ecosystem service trade-offs and

synergies is of utmost importance for planning and management. The LULC scenarios used in this study suggested a trend toward increasing supply and value of regulating and provisioning ecosystem services in the Alto Sabor basin. There is simultaneously a trend for agriculture to decrease and forest-related activities to replace farming as the major source of income. For the ecosystem services considered in this study, forest-related land uses can be highly valuable in comparison with agriculture. Although further research would be needed, our findings suggest that a new model for regional development should consider both market and nonmarket benefits of ecosystem services while designing incentives to maximize ecosystem service provision.

REFERENCES

Acs S, Hanley N, Dallimer M, Gaston KJ, Robertson P, Wilson P, Armsworth PR. 2010. The effect of decoupling on marginal agricultural systems: Implications for farm incomes, land use and upland ecology. *Land Use Policy* 27(2):550–563.

Aguiar C, Rodrigues O, Azevedo J, Domingos T. 2009. Montanha. *In:* Pereira HM, Domingos T, Vicente L, Proença V, editors. *Ecossistemas e Bem-estar Humano Avaliação para Portugal do Millennium Ecosystem Assessment*. Lisbon, Portugal: Escolar Editora, pp 295–339.

Amorim JC. 2015. *Análise e modelação da mudança da ocupação e uso do solo: Caso de estudo da bacia hidrográfica do Rio Vez*. Ponte de Lima, Portugal: Instituto Politécnico de Viana do Castelo.

Arnold JG, Srinivasan R, Muttiah RS, Williams JR. 1998. Large area hydrologic modeling and assessment part I: Model development. *Journal of the American Water Resources Association* 34(1):73–89.

Azevedo JC, Moreira C, Castro JP, Loureiro C. 2011. Agriculture abandonment, land-use change and fire hazard in mountain landscapes in northeastern Portugal. *In:* Li C, Lafortezza R, Chen J, editors. *Landscape Ecology in Forest Management and Conservation: Challenges and Solutions for Global Change*. Beijing, China: HEP-Springer, pp 329–351.

Barros AMG, Pereira JMC. 2014. Wildfire selectivity for land cover type: Does size matter? *PLoS One* 9(1):e84760.

Brander L. 2013. *Guidance Manual on Value Transfer Methods for Ecosystem Services*. Nairobi, Kenya: United Nations Environment Programme.

Briner S, Huber R, Bebi P, Elkin C, Schmatz DR, Grêt-Regamey A. 2013. Trade-offs between ecosystem services in a mountain region. *Ecology and Society* 18(3):35. http://www.ecologyandsociety.org/vol18/iss3/art35/; accessed on 1 January 2016.

Brouwer R, Brander L, Kuik O, Papyrakis E, Bateman I. 2013. *A Synthesis of Approaches to Assess and Value Ecosystem Services in the EU in the Context of TEEB. TEEB Follow-up Study for Europe*. Amsterdam, Netherlands: University Amsterdam.

CEC [Council of the European Communities]. 1979. *Council Directive 79/409/EC of 2 April 1979 on the Conservation of the Wild Birds*. Off J L 103. http://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX:31979L0409; accessed on 1 September 2016.

CEC [Council of the European Communities]. 1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Off J Eur Communities, p 66. http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CONSLEG:1992L0043:20070101:EN:PDF; accessed on 2 September 2016.

Carrão H, Gonçalves P, Caetano M. 2008. Contribution of multispectral and multitemporal information from modis images to land cover classification. *Remote Sensing of Environment* 112(3):986–997.

Carvalho-Santos C, Monteiro AT, Azevedo JC, Nunes JP. 2016. Climate change impacts on water supply: implications for reservoir management in Alto Sabor, northeast Portugal. *In:* CIMO, editor. *International Conference on Research for Sustainable Development in Mountain Regions*. Bragança, Portugal: Instituto Politécnico de Bragança, p 151.

Carvalho-Santos C, Nunes JP, Monteiro AT, Hein L, Honrado JP. 2016. Assessing the effects of land cover and future climate conditions on the provision of hydrological services in a medium-sized watershed of Portugal. *Hydrological Processes* 30:720–738.

Carvalho-Santos C, Sousa-Silva R, Gonçalves J, Honrado JP. 2016. Ecosystem services and biodiversity conservation under forestation scenarios: Options to improve management in the Vez watershed, NW Portugal. *Regional Environmental Change* 16:1557–1570.

Chen J, John R, Sun G, McNulty S, Noormets A, Xiao J, Turner MG, Franklin JF. 2014. Carbon fluxes and storage in forests and landscapes. *In:* Azevedo CJ, Perera HA, Pinto AM, editors. *Forest Landscapes and Global Change: Challenges for Research and Management*. New York, NY: Springer New York, pp 139–166.

CMB [Câmara Municipal de Bragança]. 2009. EDITAL N° 21/2009. Bragança, Portugal: Câmara Municipal de Bragança.

Duro MRG. 2008. Vendas de madeira nos perímetros florestais do Barroso-Padrela – factores relevantes na formação do preço. Vila Real, Portugal: Universidade de Trás-os-Montes e Alto Douro.

EEA [European Environment Agency]. 2010. *Europe's Ecological Backbone: Recognising the True Value of our Mountains*. Copenhagen, Denmark: European Environment Agency.

Elmqvist T, Tuvendal M, Krishnaswamy J, Hylander K. 2011. *Managing Trade-offs in Ecosystem Services*. Stockholm, Sweden: UNEP Division of Environmental Policy Implementation.

Felipe-Lucia MR, Comín FA, Bennett EM. 2014. Interactions among ecosystem services across land uses in a floodplain agroecosystem. *Ecology and Society* 19(1):20. http://dx.doi.org/10.5751/ES-06249-190120.

Fonseca F, Figueiredo, T. 2012. Carbon in soils of Montesinho Natural Park, Northeast Portugal: preliminary map-based estimate of its storage and stability. *Spanish Journal of Rural Development* 3(1):71–78.

Future Earth. 2014. *Future Earth Strategic Research Agenda 2014*. Paris, France: International Council for Science.

Gabrielson P, Bosch P. 2003. *Environmental Indicators: Typology and Use in Reporting*. EEA Internal Working Paper. Copenhagen, Denmark: European Environment Agency.

Garcia MM, Carvalheira M, Azevedo J. 2006. Contribuição para a caracterização da recolha comercial de macrofungos comestíveis no Distrito de Bragança, Portugal. *A Pantorra* 6:141–153.

Gassman PW, Reyes MR, Green CH, Arnold JG. 2007. The soil and water assessment tool: Historical development, applications, and future research directions. *Transactions of the ASABE* 50:1211–1250.

Goldstein JH, Caldarone G, Duarte TK, Ennaanay D, Hannahs N, Mendoza G, Polasky S, Wolny S, Daily GC. 2012. Integrating ecosystem-service tradeoffs into land-use decisions. *Proceedings of the National Academy of Sciences of the United States of America* 109(19):7565–7570.

Gonzalez-Redin J, Luque S, Poggio L, Smith R, Gimona A. 2016. Spatial bayesian belief networks as a planning decision tool for mapping ecosystem services trade-offs on forested landscapes. *Environmental Research* 144(part B):15–26.

GPP [Gabinete de Planeamento e Políticas]. 2011. Valores de produção padrão 2007 (quinquénio 2005–2009). Lisbon, Portugal: Gabinete de Planeamento e Políticas, Direção de Serviços de Estatística Metodologia e Estudos.

Grêt-Regamey A, Brunner SH, Kienast F. 2012. Mountain ecosystem services: Who cares? *Mountain Research and Development* 32(S1):S23–S34.

Haines-Young R, Potschin M, Kienast F. 2012. Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs. *Ecological Indicators* 21:39–53.

Hawtree D, Nunes JP, Keizer JJ, Jacinto R, Santos J, Rial-Rivas ME, Boulet AK, Tavares-Wahren F, Feger KH. 2015. Time series analysis of the long-term hydrologic impacts of afforestation in the Águeda watershed of north-central Portugal. *Hydrology and Earth System Sciences* 19(7):3033–3045.

Häyhä T, Franzese PP, Paletto A, Fath BD. 2015. Assessing, valuing, and mapping ecosystem services in alpine forests. *Ecosystem Services* 14:12–23.

Hernández-Sancho F, Molinos-Senante M, Sala-Garrido R. 2010. Economic valuation of environmental benefits from wastewater treatment processes: An empirical approach for Spain. *Science of the Total Environment* 408(4):953–957.

Howe C, Suich H, Vira B, Mace GM. 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change* 28:263–275.

ICN [Instituto da Conservação da Natureza]. 2007. *Plano de Ordenamento do Parque Natural de Montesinho. Relatório*. Bragança, Portugal: Ministério do Ambiente, do Ordenamento do Território e do Desenvolvimento Regional.

INE [Instituto Nacional de Estatística]. 1989. *Recenseamento geral agrícola 1989*. Lisbon, Portugal: Instituto Nacional de Estatística

INE [Instituto Nacional de Estatística]. 2009. *Recenseamento agrícola 2009*. Lisbon, Portugal: Instituto Nacional de Estatística.

INE [Instituto Nacional de Estatística]. 2015. *Índice de Preços do Consumidor IPC*. Lisbon, Portugal: Instituto Nacional de Estatística.

Jia XQ, Fu BJ, Feng XM, Hou GH, Liu Y, Wang XF. 2014. The tradeoff and synergy between ecosystem services in the grain-for-green areas in northern Shaanxi, China. *Ecological Indicators* 43:103–113.

Keeler BL, Polasky S, Brauman KA, Johnson KA, Finlay JC, O'Neill A, Kovacs K, Dalzell B. 2012. Linking water quality and well-being for improved assessment and valuation of ecosystem services. *Proceedings of the National Academy of Sciences of the United States of America* 109(45):18619–18624.

Körner C, Ohsawa M, Authors CL. 2005. Mountain systems. *In:* Hassan R, Scholes R, Ash N, editors. *Ecosystems and Human Well-being: Current State and Trends*. Washington, DC: Island Press, pp 681–716.

Li P, Jiang L, Feng Z, Yu X. 2012. Research progress on trade-offs and synergies of ecosystem services: An overview. *Shengtai Xuebao/Acta Ecologica Sinica* 32(16):5219–5229.

Lomba A, Gonçalves J, Moreira F, Honrado J. 2013. Simulating long-term effects of abandonment on plant diversity in Mediterranean mountain farmland. *Plant Biosystems: An International Journal Dealing With All Aspects of Plant Biology* 147(2):328–342.

Madureira L, Magalhães P, Silva PG, Marinho C, Oliveira R. 2013. *Economia dos serviços de ecossistema. Um guia para conhecer e valorizar serviços de agroecossistemas em áreas protegidas de montanha*. Lisbon, Portugal: Quercus.

Maes J, Paracchini ML, Zulian G. 2011. *A European Assessment of the Provision of Ecosystem Services: Towards an Atlas of Ecosystem Services*. Luxembourg: Publications Office of the European Union.

Marta-Pedroso C, Domingos T, Freitas H, de Groot RS. 2007. Cost–benefit analysis of the zonal program of Castro Verde (Portugal): Highlighting the trade-off between biodiversity and soil conservation. *Soil and Tillage Research* 97(1):79–90.

Martín-López B, Gómez-Baggethun E, García-Llorente M, Montes C. 2014. Trade-offs across value-domains in ecosystem services assessment. *Ecological Indicators* 37(part A):220–228.

MEA [Millennium Ecosystem Assessment]. 2005. *Ecosystems and Human Well-being: Synthesis*. Washington, DC: Island Press.

Moreira F, Russo D. 2007. Modelling the impact of agricultural abandonment and wildfires on vertebrate diversity in Mediterranean Europe. *Landscape Ecology* 22(10):1461–1476.

Navarro L, Pereira H. 2012. Rewilding abandoned landscapes in Europe. *Ecosystems* 15(6):900–912.

Neitsch SL, Arnold JG, Kiriny JR, Williams JR. 2011. *Soil & Water Assessment Tool. Theoretical Documentation*. Version 2009. College Station, TX: Texas A&M University System.

Nelson E, Mendoza G, Regetz J, Polasky S, Tallis H, Cameron D, Chan KMA, Daily GC, Goldstein J, Kareiva PM, Lonsdorf E, Naidoo R, Rickets TH, Shaw MR, et al. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* 7(1):4–11.

Nunes L, Gower ST, Peckham SD, Magalhães M, Lopes D, Rego FC. 2015. Estimation of productivity in pine and oak forests in northern Portugal using Biome-BGC. *Forestry* 88(2):200–212.

Pan Y, Wu J, Xu Z. 2014. Analysis of the tradeoffs between provisioning and regulating services from the perspective of varied share of net primary production in an alpine grassland ecosystem. *Ecological Complexity* 17(1):79–86.

Pascual U, Muradian R, Brander L, Gómez-Baggethun E, Martín-López M, Verman M, Armsworth P, Christie M, Cornelissen H, Eppink F, et al. 2010. The economics of valuing ecosystem services and biodiversity *In:* Kumar P, editor. *The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations*. London, United Kingdom: Routledge, pp 183–256.

Pérez-Rodríguez F, Azevedo J, Nunes L, Sil A. 2015. *Flornext® 1.0. User's Manual*. Bragança, Portugal: Instituto Politécnico de Bragança.

Pinheiro H, Castro JPM, Azevedo JC. 2014. Alterações na paisagem e sequestro de carbono na freguesia de Deilão, nordeste de Portugal. *Revista Árvore* 38:41–52.

Plummer ML. 2009. Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment* 7(1):38–45.

Ramos MAB. 2008. *Matos do Parque Natural de Montesinho - Erosão hídrica e dinâmica do carbono: Um estudo à micro-escala com simulação de chuva* [MS thesis]. Terceira, Portugal: Universidade dos Açores.

Rodrigues APP. 2015. *Quantificação, valoração e mapeamento de serviços de ecossistema na bacia superior do Rio Sabor (concelho de Bragança)* [MS thesis]: Bragança, Portugal: Instituto Politécnico de Bragança.

Rodríguez JP, Beard JTD, Bennett EM, Cumming GS, Cork SJ, Agard J, Dobson AP, Peterson GD. 2006. Trade-offs across space, time, and ecosystem services. *Ecology and Society* 11(1):28. http://www.ecologyandsociety.org/vol11/iss1/art28/; accessed on 1 September 2016.

Rodríguez Soalleiro R, Álvarez Gozález J, Cela González M, Mansilla Vázquez P, Vega Alonso P, González Rosales M, Ruíz Zorrilla P, Vega Alonso G. 1997. *Manual técnico de selvicultura del pino pinaster*. Santiago de Compostela, Spain: Universidad de Santiago de Compostela, Escuela Politécnica Superior de Lugo.

Schroter D, Cramer W, Leemans R, Prentice IC, Araujo MB, Arnell NW, Bondeau A, Bugmann H, Carter TR, Gracia CA, de la Vega-Leinert AC, Erhard M, Ewert F, Glendining M, House JI, et al. 2005. Ecosystem service supply and vulnerability to global change in Europe. *Science* 310(5752):1333–1337.

Sharp R, Tallis HT, Ricketts T, Guerry AD, Wood SA, Chaplin-Kramer R, Nelson E, Ennaanay D, Wolny S, Olwero N, et al. 2015. *Invest Version 3.2.0 User's Guide*. Stanford, CA: Stanford University, University of Minnesota, Nature Conservancy, and World Wildlife Fund.

Silva T, Pereira J, Paúl J, Santos M, Vasconcelos M. 2006. Estimativa de emissões atmosféricas originadas por fogos rurais em Portugal. *Silva Lusitana* 14(2):239–263

Spangenberg JH, Görg C, Truong DT, Tekken V, Bustamante JV, Settele J. 2014. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. *International Journal of Biodiversity Science, Ecosystem Services & Management* 10(1):40–53.

Stern N. 2007. *The Economics of Climate Change. The Stern Review*. Cambridge, United Kingdom: Cambridge University Press.

Tallis HT, Ricketts T, Guerry AD, Wood SA, Sharp R, Nelson E, Ennaanay D, Wolny S, Olwero N, Vigerstol K, et al. 2013. *Invest 2.5.6 User's Guide*. Stanford, CA: Natural Capital Project.

Thuiller W, Lavorel S, Araújo MB, Sykes MT, Prentice IC. 2005. Climate change threats to plant diversity in Europe. *Proceedings of the National Academy of Sciences of the United States of America* 102(23):8245–8250.

Tol RSJ. 2005. The marginal damage costs of carbon dioxide emissions: An assessment of the uncertainties. *Energy Policy* 33(16):2064–2074.

Vicente JR, Fernandes RF, Randin CF, Broennimann O, Gonçalves J, Marcos B, Pôças I, Alves P, Guisan A, Honrado JP. 2013. Will climate change drive alien invasive plants into areas of high protection value? An improved model-based regional assessment to prioritise the management of invasions. *Journal of Environmental Management* 131:185–195.

Wang Y, Neupane A, Vickers A, Klavins T, Bewer R. 2011. *Ecosystem Services Approach Pilot on Wetlands. Economic Valuation Technical Report*. Edmonton, Canada: Alberta Government.

Yang W, Yang ZF. 2014. Integrating ecosystem-service tradeoffs into environmental flows decisions for Baiyangdian Lake. *Ecological Engineering* 71:539–550.