IMPACTS OF LAND USE CHANGE ON LANDSCAPE STRUCTURE AND ECOSYSTEM SERVICES AT LOCAL SCALE: A CASE STUDY IN CENTRAL PORTUGAL

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Abstract: This study aims to evaluate land use and land cover (LULC) changes and associated impacts in the landscape structure and ecosystem services (ES) value in Lousã municipality, in Central Portugal. The results show that significant changes in LULC were recorded over the study period (1974–2018). Agricultural abandonment, expansion of woodland due to the promotion of the eucalyptus (EU) monoculture (chiefly *Eucalyptus globulus*), the invasion of exotic species (mainly *Acacia dealbata*) and the increase of built-up areas are the most significant changes that have shaped the landscapes of the study area. The analysis also revealed that the mean patch size has decreased whereas the number of patches increased. The results showed that although the EU and deciduous forest (DF) increased the ES value, the overall total ES value fell around 10% between 1974 and 2018, mainly due to the decrease in the supply of agricultural goods. Studies of this kind on local rural landscapes are vital when it comes to devising appropriate land management policies for the landscape level by considering the interaction between each element for sustainable development.

Key words: local land use/cover changes, metrics, fragmentation, ecosystem services, Lousã municipality, Portugal

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Introduction

Significant changes have occurred in land use and land cover (LULC) since the second half of the past century both in Portugal and in the countries in the Mediterranean Basin (Nunes 2008, Almeida et al. 2009, Diogo, Koomen 2012). These changes resulted not only from the application of specific agricultural policies (e.g. conversion from cultivation land into areas of pasture, forest or set aside), but also from the abandonment of traditional activities of the territory, based on agrosilvopastoral systems (Nunes 2008, Almeida et al. 2009, Nunes et al. 2011, Diogo, Koomen 2012, Meneses et al. 2017). These changes are related to the rural exodus (Nunes 2008, Diogo, Koomen 2012) and to socio-economic and regional agricultural policies, with particular emphasis on the Common Agricultural Policy (Nunes 2008, Pôças et al. 2011, Almeida et al. 2012, Kuemmerle et al. 2016). The occurrence and recurrence of disturbances such as forest fires have also been responsible for sudden changes in the landscapes structure and compositions (Almeida et al. 2012, 2013, Nunes et al. 2013). Various authors consider that the evaluation of composition and configuration changes is crucial to understanding a range of phenomena related to landscape

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fragmentation (Grimm et al. 2008, Mitchell et al. 2013, Nurwanda et al. 2016), biodiversity and ecosystem services (ES) (Reidsma et al. 2006, Moreira et al. 2012, Nunes et al. 2016, Song and Deng 2017, Rodríguez-Echeverry et al. 2018), environmental perturbations such as soil degradation and wildfires (Nunes et al. 2011, 2016, Borrelli et al. 2017, Brevik et al. 2017), climate change (Li et al. 2017, Tasser et al. 2017) and human health impacts (Patz et al. 2004). LULC changes commonly influence ES provided by forests, shrubs and cultivated land (CL) across landscapes both on spatial and temporal scales (Midha, Mathur 2010, Pinto-Ledezma, Rivero Mamani 2014, Wang, Yang 2012, Zipperer et al. 2012, Berhane et al. 2013, Cuke, Srivastava 2016). The increase in the number of patches (NumP) and isolation can alter ES such as biodiversity conservation, pollination, carbon sequestration and seed dispersal (Debuse et al. 2007, Kremen et al. 2007, Çakir et al. 2008, Hartter, Southworth 2009, Herrera, Garcia 2010, Wang, Yang 2012, Zipperer et al. 2012, Putz et al. 2014, Qi et al. 2014). Therefore, LULC analysis, both spatially and temporally, is increasingly important in the context of the sustainable management of the landscapes (Salvati et al. 2016). The sustainability of regional/municipal service ecosystems depends upon the support of the ecosystems (land use, forests, water bodies [WB], soils, etc.) and social systems (population, local organisations) that provide and manage these resources. If municipal service systems undermine these resources, then they will ultimately fail. To understand such complex interactions, it is crucial to understand the system in which they occur, integrating different scales and disciplines (Turner et al. 1989, Grau et al. 2013, Nesheim et al. 2014). Turner et al. (1989) state that qualitative and quantitative changes in measurements across spatial scales will differ depending on how scale is defined. In this sense, scale has been identified as one of the important topics in LULC changes (Turner et al. 1989, Holling 1992) and upscaling of local understandings is key to many studies of environmental management (Thrush et al. 1997, Gibson et al. 2000, Liu, Taylor 2002, Liu, Weng 2013). Also, there is a growing consensus that sustainability must be achieved at the local level (United Nations 1992); it cannot be a policy only at higher levels of governance or a corporate commitment (Ostrom 2009).

Considering the abovementioned issues, the objectives of this study were: 1) to evaluate the changes in LULC from 1974 to 2018, in three selected areas with different biophysical characteristics and distinct human occupation in the Lousã municipality, in the central region of Portugal, 2) to evaluate the effect of LULC change on landscape composition and configuration, with direct impacts on landscape fragmentation and 3) to estimate the potential changes on the provision of ES, such as soil sediment retention, carbon sequestration or agricultural goods. Understanding the potential impacts of these patterns of change on the provided ES is also of great importance as it offers a rationale for formulating rural development policies which can address sustainable livelihoods by integrating appropriate land management strategies.

Materials and methods

Study area

The study was conducted in Lousã municipality, in the central region of Portugal. Three selected areas are representative of the geodiversity of Lousã municipality (Fig. 1).

The municipality of Lousã is relatively heterogeneous in terms of climate, geology and topography, so it was considered suitable for studying the effects of spatial and structural features on the patterns of landscape, biodiversity and forest patches. Thus, the selected areas of study include different types of landscape (LULC) because of their biophysical characteristics and human occupation. The first selected area, on the 'western slope of Serra da Lousã', includes a landscape of mountains marked by a strong altimetric gradient that varies between 348 m and 1134 m as well as quite steep slopes with an average of 20% (Table 1). The dominant bedrock type is schist, which generates nutrient poor soils. The second area of study is mostly drained by the Ceira River and occupies a lower altitude, with a mean of 200 m and an average slope of 8%. The third area is located in the Lousã Basin and it partly covers the urban perimeter of the municipality's parishes. The mean altitude is 82 m and the average slope is 3%. North and west aspects are dominant in the study area.

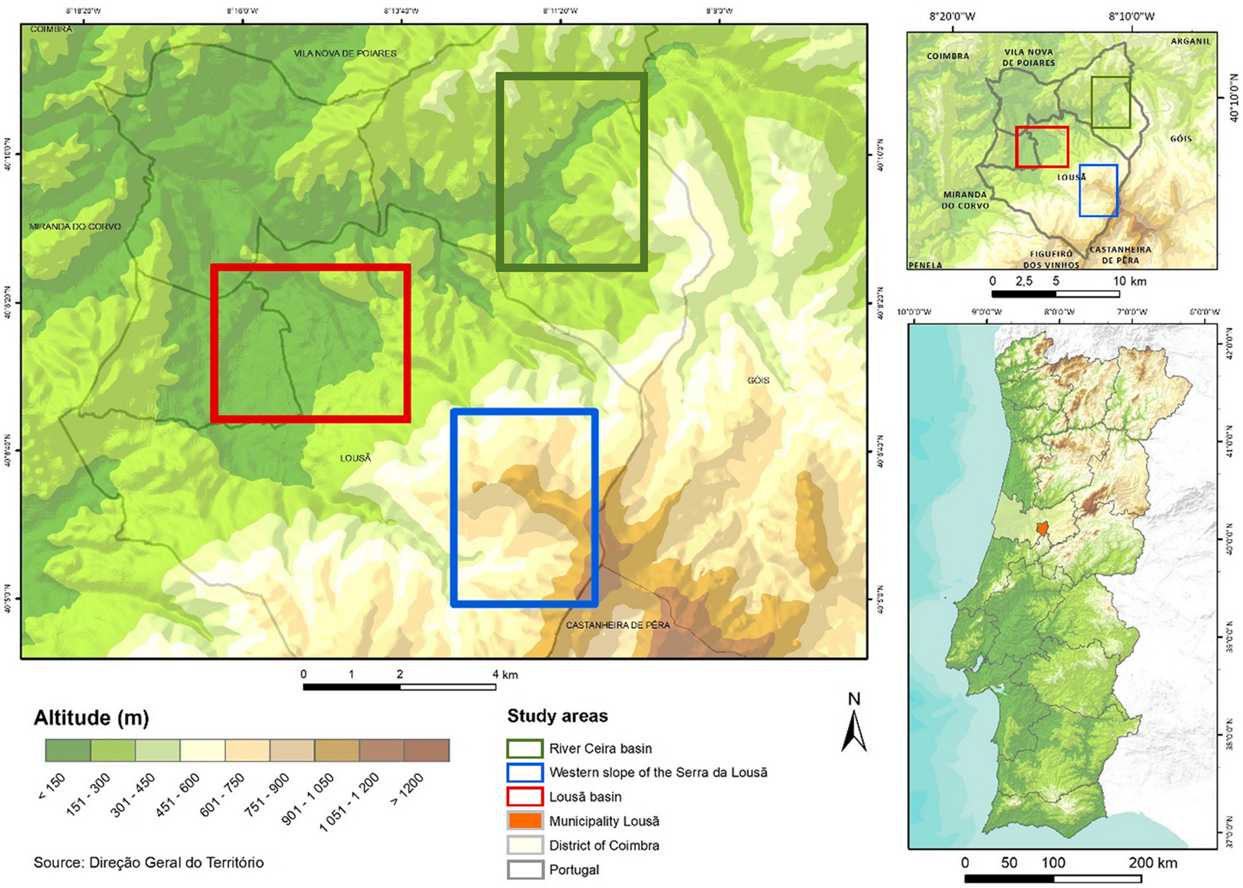

Fig. 1. Location of study areas.

Table 1. Main physical characteristics of the municipality of Lousã and studied areas.

Parameter	Units	Western slope of the Serra da Lousã	River Ceira Basin	Lousã Basin	Municipality of Lousã
Maximum altitude	[m]	1134	438	232	1195
Minimum altitude		348	105	82	60
Mean altitude		748	197	135	326
Mean slope	[%]	20	8	3	11

Methodology

The use of time series of historical maps and aerial photography is a common practice in historical geography and has proven to be very useful (Ihse 1996, Skånes, Bunce 1997, Vuorela 2000). In this study, land use and cover change were based on maps produced by the Agrarian Survey and Management Service for 1974, and the LULC Map of Mainland Portugal for 2018 from the Directorate-General for Territorial Development (both at 1:25,000 scale). The main difficulty with the simultaneous use of these documentary sources has been the different classes of inventorying. Thus, we had to carry out a process of unification through classes which could effectively and immediately demonstrate the significant uses conferred on the territory and the changes carried out in the past few decades (Table 2).

For this purpose, the different classes of land occupation/cover were combined to facilitate the quantification of changes and the application of landscape metrics, which were implemented through the use of geographical information systems (GIS). All spatial analyses and procedures related to LULC change and landscape metrics were implemented through the ArcGIS software, using ArcMap version 10.5.1 and the extension Patch Analyst. The analysis of the landscape metrics used the variables of the *Patch Analyst* that were best suited to our objective (Table 3), related

Table 2. Harmonisation of land use/land cover classification systems.

Agrarian Survey and Management Service, 1974	Used classes	Land use and cover map of continental Portugal, 2018
Social areas	SA	Level 1 – Artificialised territories
Temporary and permanent crops, except olive groves (cultivated land)	CL	Level 2 – Temporary and permanent crops, except olive groves Level 2 – Heterogeneous agricultural areas
Olive grove	OG	Level 3 – Olive groves
Pinus pinaster and *Pinus pinea*	PW	Level 4 – Pine woods
Eucalyptus	EU	Level 5 – Eucalyptus
Deciduous forests (oak, chestnut and other deciduous trees)	DF	Level 5 – Deciduous forests minus Eucalyptus and invasive forests
Shrubland	SL	Level 2 – Open forests, shrubland and herbaceous vegetation + Uncovered areas with slight vegetation
Invasive species (termed 'acacias')	IS	Level 5 – invasive species (classified as invasive forest species, according to Portuguese legislation, i.e. *Acacia dealbata, Ailanthus altissima, etc.)*
Water bodies	WB	Level 1 – water bodies

Table 3. Metrics and their units.

Metrics	Units	Variable
Class area	[ha]	Class area is a measure of landscape composition, specifically, how much of the landscape is composed of a particular patch type.
Percentage of landscape	[%]	The percentage of the landscape comprised of a particular patch type/land use.
Number of patches	[No.]	Number of patches of corresponding patch type (class).
Median patch size	[ha]	Average area of patches of corresponding patch type (class).
Patch density	[No./100 ha]	Number of patches of corresponding patch type (class) per unit area.
Total edge	[m]	The sum of the lengths (m) of all edge segments in the landscape.
Edge density	[m ha^{-1}]	The sum of the lengths (m) of all edge segments in the landscape, divided by the total landscape area (ha).
Mean patch edge	[m/Patch]	Average patch edge in each class, expressed in meters per ha.
Shannon diversity index	[–]	Shannon diversity index refers to the variety and abundance of different land cover types within a landscape. A low value means that the landscape is dominated by one land cover type. The value of the index tends to 1 when the land cover types present have roughly equal proportion or a high number of cover/use types actually being present.
Simpson's diversity index	[–]	Simpson's diversity index is a measure of diversity which takes into account the number of categories present as well as the relative abundance in each category. Ranges between 0 (no diversity) and 1 (infinite diversity).

Table 4. Estimate of the economic value for the different ecosystem services by land use category; mean economic value per ecosystem services adapted from Marta-Pedroso et al. (2014).

Land use and land cover	Mean economic value per ecosystem services [€ ha^{-1} a^{-1}]				
	Sediment retention	Carbon sequestration	Agricultural goods	Fibre	Biodiversity
Social areas	2	0	0	0	0
Cultivated land	130	−54	196	0	0
Olive grove	399	−50	585	0	0
Pine woods	740	111	0	596	0
Eucalyptus	496	−50	0	123	0
Deciduous forest	499	162	0	0	74
Shrubland	599	122	0	0	0
Invasive spc.	n.a.	n.a.	n.a.	n.a.	n.a.

n.a. Not available.

to the landscape composition, configuration and diversity.

The configuration metrics are mainly used to describe the spatial characteristics of individual patches or the spatial relationships between the multiple patches (Botequilha Leitão et al. 2002, Couto 2004). This group of metrics also integrates measurements of the landscape configuration, such as area and edge metrics.

In order to estimate the change in ES flow, the ES mean economic value, adapted from Marta-Pedroso et al. (2014), was used to describe both spatial and temporal changes of ES. The estimated economic values are systematised in Table 4, according to the reference values achieved for a pilot area, located in the Alentejo region, Portugal. The obtained results are included in the first The Economics of Ecosystems and Biodiversity (TEEB) study in Portugal, in which ES were identified based on the literature, expert judgement, stakeholders' engagement and field work. The economic valuation of these ES was performed using available information and relied mainly on the use of avoided costs (carbon sequestration and soil protection), willingness to pay (biodiversity) and market prices (crop and fibre production) methods (Table 4). According to the authors, the economic value of the different services was based on estimates or proxies of values, namely data published in scientific journals, or information available in the public administration that allowed an estimate of the economic value. Thus, the economic coefficients used for this study should be interpreted as proxy values as the estimates of these values of ES can be biased for the study area.

The Ecosystem Services Total Value (ESV) at time T is estimated by multiplying the area of each land-cover category by the coefficient associated to each land use/land cover, using the following relationship:

$$SV = (Ak \times VCk) \qquad (1)$$

where:

- Ak – the area in hectares of land-cover category 'k',
- VCk – the value coefficient (€ ha^{-1} a^{-1}) (Table 4), which we have assumed constant during the temporal range under study.

Results

Land use and land cover changes

Changes in LULC in the period from 1974 to 2018 are presented in Figure 2 and summarised

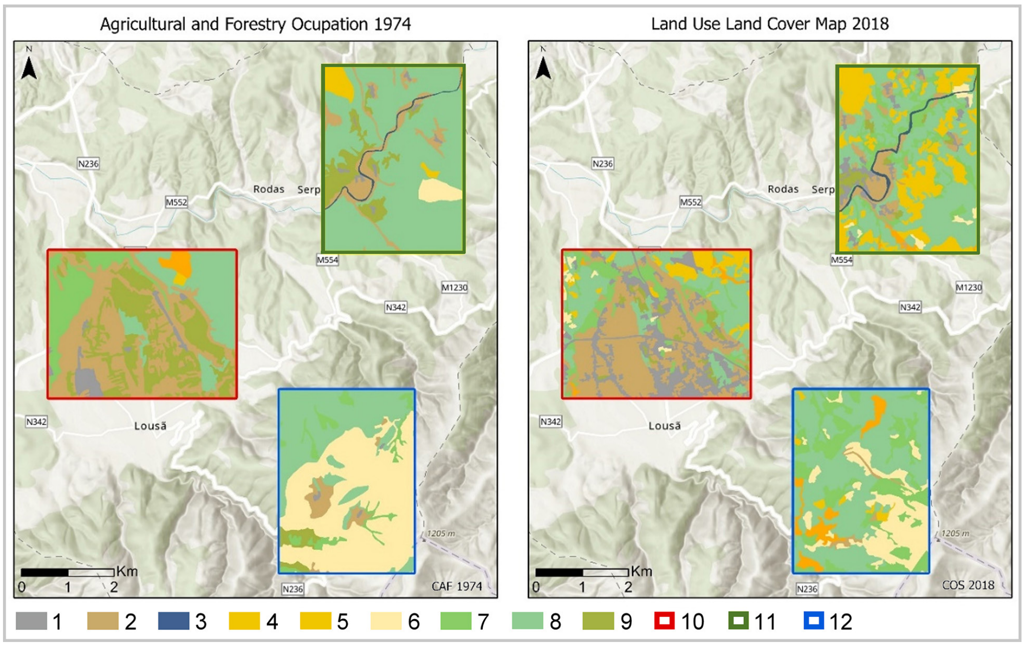

Fig. 2. Land use and land cover in the Lousã municipality in 1974 and 2018. 1 – social areas, 2 – cultivated land, 3 – water bodies, 4 – eucalyptus, 5 – invasive species, 6 – shrubland, 7 – deciduous forest, 8 – pine woods, 9 – olive grove, 10 – Lousã Basin, 11 – River Ceira Basin, 12 – Western slope of the Serra da Lousã.

Table 5. Synthesis of changes in land use and land cover in the Lousã municipality (1974–2018, by class in ha and (%).

Land use and land cover	Social areas	Cultivated land	Olive grove	Deciduous forest	Pine woods	Eucalyptus	Invasive species	Shrubland	Water bodies	Total area in ha (%) 2018
Social area	24.2 (0.7)	112.7 (3.1)	117.9 (3.3)	8.5 (0.2)	35.0 (1.0)	–	8.2 (0.2)	1.7 (0.0)	0.6 (0.0)	308.9 (8.6)
Cultivated land	33.2 (0.9)	314.6 (8.7)	165.0 (4.6)	12.8 (0.4)	28.0 (0.8)	–	–	10.1 (0.1)	4.9 (0.1)	568.6 (15.8)
Olive grove	1.7 (0.0)	10.7 (0.3)	46.2 (1.3)	–	2.4 (0.1)	–	–	–	–	61.0 (0.7)
Deciduous forest	6.1 (0.2)	111.3 (3.1)	52.5 (1.5)	61.1 (1.7)	211.1 (5.9)	1.0 (0.0)	–	93.7 (2.6)	1.3 (0.0)	538.2 (14.9)
Pine woods	2.2 (0.1)	44.3 8 (1.2)	41.1 (1.1)	110.1 (3.1)	809.4 (22.5)	5.6 (0.2)	–	340.2 (9.5)	0.3 (0.0)	1353.3 (37.6)
Eucalyptus	1.7 (0.0)	12.2 (0.3)	0.3 (0.0)	13.1 (0.4)	314.3 (8.7)	33.9 (0.9)	9.9 (0.3)	10.4 (0.3)	–	395.9 (11.0)
Invasive species.	–	10.0 (0.3)	20.3 8 (0.6)	5.0 (0.1)	35.2 (1.0)	–	–	21.3 (0.6)	–	91.8 (2.6)
Shrubland	–	2.3 (0.1)	6.0 (0.2)	17.9 (0.9)	32.2 (0.9)	–	–	198.8 (5.5)	0.3 (0.0)	257.6 (7.2)
Water bodies	–	10.4 (0.3)	1.5 (0.0)	–	4.3 (0.1)	–	–	–	8.4 (0.2)	24.6 (0.7)
Total area in ha (%), 1974	69.2 (1.9)	628.6 (17.5)	450.9 (12.5)	228.5 (6.3)	1471.9 (40.9)	40.5 (1.1)	18.2 (0.5)	676.3 (18.8)	15.9 (0.4)	3600 (100.0)
LULC change (%)	346.5	−9.5	−86.5	135.5	−8.0	877.8	405.6	−96.4	54.4	–

in Table 5. The main changes observed result from a substantial increase of areas occupied by EU (+878%), IS (+405%) and the social areas (SA) (+346%). Conversely, the areas devoted to agriculture and occupied by OGs saw a decrease in their areas, of 10 and 87%, respectively. Between 1974 and 2018, the pine forest remained the dominant land cover in the territory, suffering only a slight reduction.

The conversion from pine forest, shrubland and CL into EU forest explains its relevant increment

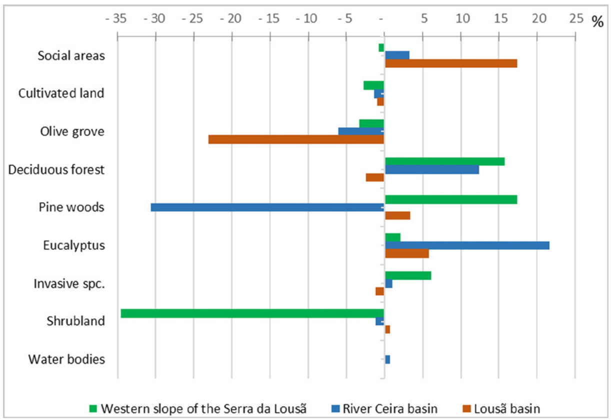

Fig. 3. Land use and land cover changes in the selected local areas, 1974–2018.

while the IS expanded mostly by soils previously occupied by pines, OGs, shrubs and CL. Social facilities have expanded into areas which had previously been cultivated or contained OGs. As for DFs, they increased in area chiefly due to the decline of pine forests (35%) and cultivated areas (26%). The abandonment of agriculture has led to part of the cultivated area now being occupied by DF (20.6%) and SA (16.6%).

Comparing the LULC dynamics in the selected areas (Fig. 3) we can conclude that agricultural land and OGs decreased in the three areas, while PW and shrubland also show a significant decrease in the River Ceira Basin and in the western slope of Serra da Lousã, respectively.

At the same time, the EU enjoyed a general expansion, whilst DF and IS register an increase in the River Ceira Basin and on the western slope of Serra da Lousã. SA have increased significantly in the Lousã Basin, coinciding with the main town in the municipality.

Landscape metrics changes

Figure 4 systematises the metrics of the landscape, considering the two periods under analysis.

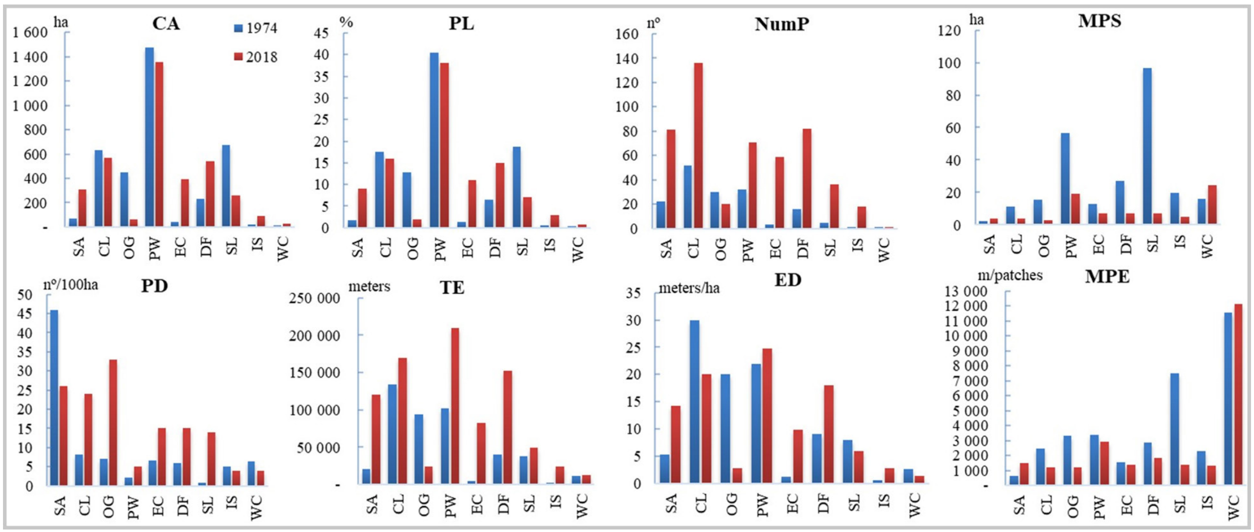

Fig. 4. Changes in landscape composition and configuration in the Lousã Municipality (1974–2018). CA – class area; CL – cultivated land; DF – deciduous forest; EC – eucalyptus; ED – edge density; IS – Invasive species; MPE – mean patch edge; MPS – median patch size; NumP – number of patches; OG – olive grove; PD – patch density; PL – % of landscape; PW – pine woods; SA – social areas; SL: Shrubland – TE – total edge; WC – water bodies.

The results show an almost triple increase in the NumP in all land uses/covers and a decrease in the average size, from 30 ha to 7 ha, meaning that the landscape has experienced a strong fragmentation process. Pine remains the species with the greatest spatial relevance, occupying around 40% of the total area, and has the largest NumP (71 No./P) and the highest average size, despite the significant decrease observed, that is, from 57 ha to 19 ha. The highest increase in the NumP is recorded by the EU and DF, although the respective mean patch area decreased from 13 ha to 7 ha and from 27 ha to 7 ha. The same trends are noted for shrubland and IS. The CL records a significant increase in the NumP between 1974 and 2018; however, the relevant size fell by about one-third. The social area is the only LULC to register an increase in the mean patch size. Consequently, the patch density (PD) has also expanded in all LULC classes, whilst total edge (TE) and edge density (ED) only decreased for the CL and OG.

Both the Shannon Diversity Index (SDI) and the Simpson diversity index (SEI), which express the degree of landscape diversity given by the number of classes and proportion of the landscape area occupied by each class, indicate an increment from 1.2 to 1.6 and 0.6 to 0.7, respectively, from 1974 to 2018.

In the western slope of the Serra da Lousã (Table 6), the landscape matrix was dominated by shrubland in 1974, and it changed to PW by 2018. SA, CL and OGs almost disappeared in 2018, whereas IS and EU forest record the highest increases. With the lowest NumP among the selected areas, the western slope of the Serra da Lousã recorded an increase from 48 to 74 (+41%), a decrease in the median path size, from 36 ha to 13.6 ha, and in the PD, from 61 ha to 59/100 ha. Only the PW doubled their patch size, from 1974 to 2018. Both indexes of diversity – SDI and SEI – have recorded an increase from 1.59 to 1.78 and 0.72 to 0.81, respectively.

In the River Ceira Basin (Table 6), PW stands as the dominant landscape matrix in both periods, in spite of its significant decline in the total area recorded in recent decades. Conversely, EU and DFs have recorded a relevant increase. Overall, the NumP has more than doubled, and the respective average area has fallen from 30 ha to 7.5 ha. The SDI and SEI also recorded an increase, indicating that the landscape heterogeneity and evenness have slightly increased. Table 6 also shows that although CL has lost a significant part of its area in the Lousã Basin, in 2018 it is still the most important land use, with a value significantly above PW, which increased around 3%. SA and EU forest recorded the highest increment (around 600%). Conversely, OGs decreased by 70%. In general, the total NumP tripled, whereas the mean area per patch has fallen, from 37 ha to

Table 6. Changes in landscape composition and configuration in the three selected areas.

Types of land cover and land use	Percentage of landscape		Number of Patches		Median Patch Size [ha]	
	1974	2018	1974	2018	1974	2018
Western slope of Serra da Lousã						
Social areas	0.9	0.1	3	1	3.6	17.0
Cultivated land	4.9	2.2	7	4	8.3	6.6
Olive grove	3.3	–	2	–	20.0	–
Deciduous forest	6.9	22.7	13	26	6.4	10.5
Pine wood	30.6	47.9	19	12	19.3	47.9
Eucalyptus	–	2.1	–	6	–	4.3
Shrubland	53.4	18.8	4	19	160.2	11.9
Invasive species	–	6.1	–	6	–	12.3
River Ceira Basin						
Social areas	1.3	4.6	9	21	1.7	2.6
Cultivated land	9.7	8.4	22	36	5.3	2.8
Olive grove	8.5	2.4	6	6	17.0	4.7
Deciduous forest	–	12.5	–	30	–	5.0
Pine wood	72.9	42.3	7	36	125.0	14.1
Eucalyptus	3.3	24.9	2	34	20.0	8.8
Shrubland	3.0	1.8	1	7	35.6	3.1
Invasive species	–	1.0	–	7	–	1.9
Water bodies	0.44	0.68	1	1	15.9	24.5
Lousã Basin						
Social areas	3.6	21.0	20	59	2.2	4.3
Cultivated land	38.0	36.0	20	94	15.5	4.6
Olive grove	26.0	2.7	20	14	14.0	2.3
Deciduous forest	12.0	9.7	1	26	145.0	4.5
Pine wood	19.2	22.5	4	23	57.6	11.8
Eucalyptus	–	5.9	1	19	0.5	3.7
Shrubland	–	–	–	–	–	–
Invasive species	1.5	0.4	1	5	18.2	1.0

5 ha. Considering all the LULC, the NumP has increased, whereas the mean area per patch has only grown in the SA (from 2.2 ha to 4.3 ha).

Impacts on ecosystem services providing

By using the value coefficients and areas of the main LULC categories changes (Tables 4 and 5, respectively), the ecosystem service value of land use category was calculated for both years (1974 *vs* 2018). The results showed that although the eucalyptus and deciduous forest increased the ecosystem service value, the overall total ecosystem service value fell around 10% between 1974 and 2018, as a consequence of the decline recorded in the olive groves (−74%), cultivated land (−44%) and shrubland (−54%) (Fig. 5).

The estimated ecosystem service value for the three selected areas demonstrates different dynamics as we can see in Figure 6. On the western slope of Serra da Lousã an overall increase in the ecosystem service value mainly related to the growth observed in the pine woods area. Conversely, in the Ceira River and Lousã Basin the registered decrease in the overall ecosystem services value are mainly related to the significant decrease in the pine woods and agricultural area, respectively.

Pine woods and olive groves are the LULC that contribute the most to the variability

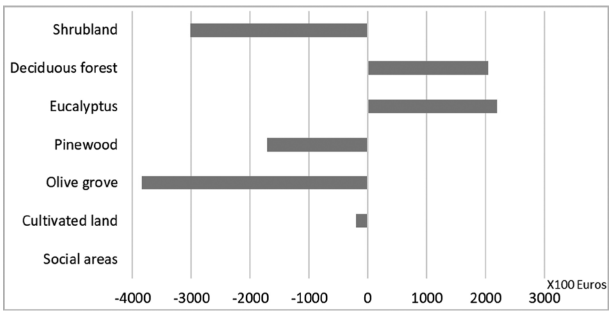

Fig. 5. Changes in the ecosystem service value (€ a^{-1}) by land use category between 1974 and 2018.

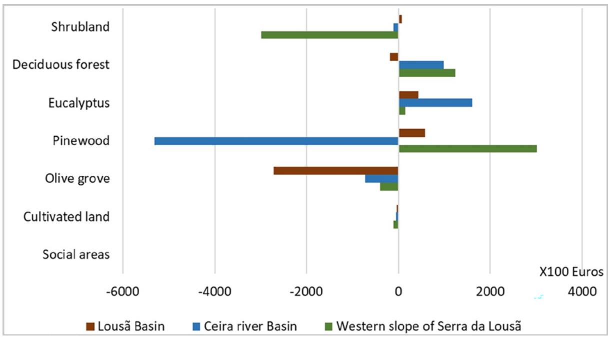

Fig. 6. Changes in the ecosystem service value (€ a^{-1}) by land use category, between 1974 and 2018, for the three selected areas.

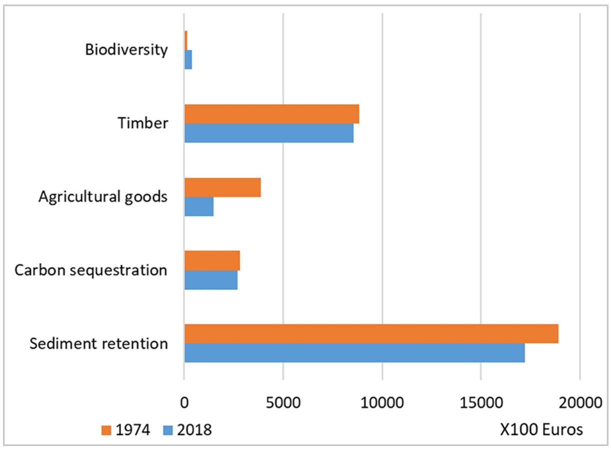

Fig. 7. Changes in the ecosystem services assessed (€ a^{-1}) between 1974 and 2018.

observed in the ecosystem services value, whilst the regulation services (sediment retention and timber production) are the ones with the highest value. Considering the various ES evaluated, the greatest losses occurred in the production of agricultural goods (−64% of the annual value) while the most evident gains occurred in biodiversity, where the value more than doubled (Fig. 7).

Discussion

The current study integrated LULC and landscape metrics to expose landscape composition and configuration changes over the past four decades. The LULC analysis was based on eight classes in the study areas. The results showed that conversion from one land use to the other is dynamic, and it did not follow a spatial linear pattern over a 40-year period in the study area. This means that certain land use types such as PW or SA showed increase and decrease patterns in the three selected areas, although integrated in the same municipality. On the other hand, the rates of change associated with specific LULC varies significantly between areas, although an overall decrease/increase is recorded for the whole study period and at municipal level. Such changes in local LULC are anthropogenic in nature and influenced by complex interactions between environmental, socioeconomic, political and social factors (Echeverria et al. 2008, Verburg et al. 2010, Pôças et al. 2011, Temesgen et al. 2013, Turner, Gardner 2015). Despite this local variability, we can say that five principal phenomena shaped the landscapes of the study area with consequences on the value of the ES provided.

Agricultural abandonment

If we take the three selected areas, a general trend in agricultural abandonment was observed that was more significant in the mountain area of Lousã municipality, which can be mainly attributed to rural exodus. In fact, villages such as Cerdeira and Candal, on the western slope of the Serra da Lousã, had suffered a truly significant reduction in the population (280 inhabitants in 1940 to 10 inhabitants by 2011, i.e., −96%) (INE 1947, 2012). River Ceira Basin also saw its population decrease between 1940 and 2011 (from 2400 to 1800 inhabitants, i.e., −25%) however, the rates of conversion were significantly lower. This agricultural abandonment has negatively impacted the supply of agricultural goods, which have been partially counterbalanced by an increase in fibre production and biodiversity, mainly associated with an expansion of native forest species. In fact, several authors (Lasanta et al. 2015, Nadal Romero et al. 2016, Gashaw et al. 2018, Perpiña Castillo et al. 2020) view agricultural abandonment as an opportunity to reverse the long-term decline of forests, provision of ES, and habitat enhancement that re-naturalisation of landscape provides. Other authors (Qin et al. 2013, Ramankutty et al. 2018, Zhang et al. 2018, Chaudhary et al. 2020, Crawford et al. 2022), however, regard agricultural abandonment as negative process with impacts on the local population and on society as a whole in terms of the production of goods (e.g. foodstuffs, livestock feed, fibre), biodiversity, as well as services provided by the multi-functionality (e.g. socio-cultural practices, values, and norms) of the agricultural landscape.

Forestland increase

Forests (evergreen plantation and deciduous woodland) occupy the major proportions of land use in the study area and also recorded the highest increase over the analysed period. Plantations of *Pinus pinaster* constitute the dominant element of the landscape matrix. The high percentage of PW is related to several afforestation campaigns promoted by the Portuguese government, mostly involving mountain common lands and the planting of *Pinus pinaster*. Since joining the European Union in 1986, new forestry policies have been adopted by Portugal. These prioritise the restoration of mixed woodlands and closer collaboration with private owners of woodlands and forests (Canaveira et al. 1998), explaining the increase observed in both forest plantation (mainly *Eucalyptus* species, due to given their fast growth) and deciduous. In the study area, as well as in other parts of the Mediterranean region, the increase of DF mainly results from agricultural abandonment that enhanced natural secondary succession and the spread of woodland (Nunes 2008, Almeida et al. 2009, Lasanta et al. 2009).

It is widely accepted that forest ecosystems provide diverse critical services and benefits to human society (Jenkins, Schaap 2018). Although the increase of EU monoculture plantations have a positive impact in the timber production, various authors consider that monoculture plantations could affect several regulating ES, related to soil, water and biodiversity. Tree harvesting by machines can promote soil erosion and compaction (Boltodano 2000, Bowyer 2006, Affek et al. 2017) which will adversely affect the growth of understory. Single-species plantations are also not efficient in trapping nutrients, because there are fewer roots near the surface, which may further lead to significant loss of nutrients from the harvest sites. In addition, some species, such as Eucalyptus, can acidify the soil and release specific substances that inhibit the growth of other plant species (Bowyer 2006), thus affecting the biodiversity. Some researchers (Morris et al. 2004, Bowyer 2006) have observed that Eucalyptus consumes more water than other species in natural forests, which may draw down the water table in some regions.

On the other hand, the impressive afforestation measures implemented on common and private land in the twentieth century went ahead without proper forest management, silviculture and fuel-hazard reduction, especially after the 1970s (Mateus, Fernandes 2014), have increased the wildfires hazard.

Alien species in expansion

The municipality of Lousã has registered a very important increase in alien species, especially *Acacia dealbata,* mainly as a consequence of land abandonment and the recurrence of wildfires. The *Acacia dealbata* is considered as a problematic and widespread invasive plant; it has negative impacts on the ecosystem's structure and functioning, triggering ecological homogenisation and reducing biodiversity (Binggeli 1996, Williamson 1999, Chapin et al. 2000, Pysek, Richardson 2010, Correia 2012, Marchante et al. 2015, Gil 2017). Therefore, biological invasions by alien species have been recognised as one of the most important drivers of biodiversity loss and ES changes worldwide. Also, they have negative effects on socioeconomic, cultural and human health aspects by affecting all four categories of ES: supporting, provisioning, regulating and cultural services (Vilá, Weiner 2004, Marchante et al. 2015, Gil 2017). In this context, urgent preventative, eradication and control actions are required to impede their entry and establishment or minimise their long-term impacts (Robertson et al. 2020).

Urban sprawl

Urban expansion in the study area occurs mainly in the Lousã Basin, where the municipality's main town is located, as it responds to pressures on the territory to accommodate new residential, service or industrial areas. This process of transforming agricultural and forest land uses into urban land cause substantial losses of natural habitats and ES (e.g. food production, freshwater provision, and carbon storage) due to the loss of vegetation and increase in impervious surfaces

(Zhang et al. 2017). Several authors (Schröter et al. 2005, Wade et al. 2009, Wu et al. 2014) also consider that it indirectly influences the delivery of ES, such as water retention, climate regulation and nutrient retention by altering the hydrologic cycling, atmospheric circulation and nutrient cycling processes.

Landscape fragmentation

Fragmentation was the most obvious characteristic of landscape change in all the areas studied, even though the obtained results might be partially overestimated by different details in the mapping between 1974 and 2018. The results indicate the threefold increase in the NumP and the decrease by half in the mean patch area.

Although both Diversity Indexes used denoted an increase in the study area, several authors consider that the habitat fragmentation commonly influence ES provided by forests, shrubs and grasslands across landscapes on both spatial and temporal scales (Kremen et al. 2006, Laurance et al. 2007, Midha, Mathur 2010, Berhane et al. 2013, Pinto-Ledezma, Rivero Mamani . 2014, Cuke, Srivastava 2016). As mentioned by many authors (Debuse et al. 2007, Çakir et al. 2008, Hartter, Herrera 2009, Zipperer et al. 2012, Qi et al. 2014), the high increase in the NumP, the shrinking mean patch size as well as increasing ED at the landscape level corresponded to the declining quality of ES such as biodiversity conservation, carbon sequestration and seed dispersal.

Conclusions

The dynamics of LULC changes at local scale in the municipality of Lousã show different spatial patterns with respect to land abandonment, increase in the areas occupied by planted versus deciduous forests and in the urban built-up area. The combining of LULC data with fragmentation analysis improves our understanding of the level of landscape transformation, the nature of such changes and how each land use type became aggregated with or dispersed from others. The analysis of fragmentation at class level provided detailed information in relation to the size of each patch, NumP, percentage of land use within the landscape and other important variables that can be helpful to understand how the different land uses could determine changes in ES such as erosion control, carbon sequestration and production of agricultural goods and fibre in the study area. However, it is important to emphasise that the analysis of ES values used as reference in this work is rather a kind of scientific exercise and aims to highlight the problem of changing ES along with LULC changes. In fact, ES values are highly arbitrary and only some specific aspects of particular ES were taken into account, and only a few of the many possible ES were considered.

Although the results of this study are strongly dependent on the local scale analysis, governed by combinations of different factors, such as geographical, environmental, socio-economic and political ones, more work is needed with the aim to make clear the relevance of LULC changes and their impacts on ES value. This is the only way to reinforce the need for LULC changes and ES to be considered as a basis for formulating local rural development policies that can address sustainable livelihoods by integrating appropriate land management strategies.

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Author’s contribution

Conceptualisation: A.N., BM and AO; methodology: AN, AO, AD, PV and BM; software: AO, AD and PV; formal analysis: AN, AO, AD, PV and BM.; investigation: AN, AO, AD, PV and BM.; writing—original draft preparation, AN, AO and BM. All authors have read and agreed to the published version of the manuscript.

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References

Affek A.N., Zachwatowicz M, Sosnowska A., Gerlée A., Kiszka K., 2017. Impacts of modern mechanised skidding on the natural and cultural heritage of the Polish Carpathian Mountains. *Forest Ecology and Management* 405: 391–403. DOI 10.1016/j.foreco.2017.09.047.

Almeida A.C., Nunes A., Figueiredo A., 2012. Recent changes in the Portuguese landscape. In: Passos M.M., Cunha L., Jacinto R. (eds), *The new Geographies of Portuguese-speaking countries: landscapes, territories and policies in Brazil and Portugal (II)*. Edit. Outras Expressões, S. Paulo, Brasil: 113–135.

Almeida A.C., Nunes A., Figueiredo A., 2013. Landscape at risk. In: Lourenço L.F., Mateus M.A. (eds.). *Natural, manmade and mixed risks. Homenagem ao Professor Doutor Fernando Rebelo*". Departamento de Geografia. Faculdade de Letras. Universidade de Coimbra: 741–750.

Almeida A.C., Nunes A., Figueiredo A., 2009. *Land use change in the central and northern inland of Portugal*. Coimbra University Press, Portugal.

Baltodano J., 2000. *Monoculture forestry: a critique from an ecological perspective. Tree trouble: a compilation of testimonies on the negative impact of large-scale monoculture tree plantations*. 6th COP of the FCCC, Friends of the Earth International: 2–10.

Berhane A., Totland Ø., Moe S.R., 2013. Woody plant assemblages in isolated forest patches in a semiarid agricultural matrix. *Biodiversity Conservation* 22: 2519–2535. DOI 10.1007/s10531-013-0534-0.

Binggeli P.A., 1996. Taxonomic, biogeographical and ecological overview of invasive woody plants. *Journal of Vegetation Science* 7: 121–124. DOI 10.2307/3236424.

Borrelli P., Robinson D.A., Fleischer L.R., Lugato E., Ballabio C., Alewell C., Meusburger K., Modugno S., Schütt B., Ferro V., Bagarello V., Oost K.V., Montanarella L., Panagos P., 2017. An assessment of the global impact of 21st century land use change on soil erosion. *Nature Communications* 8. DOI 10.1038/s41467-017-02142-7.

Botequilha Leitão A., Ahern J., 2002. Applying landscape ecological concepts and metrics in sustainable landscape planning. *Landscape and Urban Planning* 59(2): 65–93. DOI 10.1016/S0169-2046(02)00005-1.

Bowyer J.L., 2006. Forest plantations threatening or saving natural forests? *Arborvitae* (IUCN/WWF Forest Conservation Newsletter) 31: 8–9.

Brevik E.C., Pereira P., Muñoz-Rojas M., 2017. Historical perspectives on soil mapping and process modeling for sustainable land use management. In: Pereira P., Brevik E., Muñoz-Rojas M., Miller B. (eds), *Soil mapping and process modelling for sustainable land use management*. Elsevier: 3–28. DOI 10.1016/B978-0-12-805200-6.00013-X.

Çakir G., Ün C., Baskent E.Z., Köse S., Sivrikaya F., Keleş, S., 2008. Evaluating urbanization, fragmentation and land use/land cover change pattern in Istanbul city, Turkey from 1971 to 2002. *Land Degradation and Development* 19: 663–675. DOI 10.1002/ldr.859.

Canaveira P., Almeida A.M., Teixeira J.S., Oliveira R., 1998. *Underlying Causes of Deforestation and Forest Degradation (Synthesis Report of the European Regional Meeting – Bonn, Germany): Forests and People in the Iberian Peninsula*. World Rainforest Movement.

Chapin F.S., Zavaleta E.S., Eviner V.T., Naylor R.L., Vitousek P.M., Reynolds H.L., Hooper D.U., Lavorel S., Sala O.E., Hobbie S.E., Mack M.C., Díaz S., 2000. Consequences of changing biodiversity. *Nature* 405: 234–242. DOI 10.1038/35012241.

Chaudhary S., Wang Y., Dixit A.M., Khanal N.R., Xu P., Fu B., Yan K., Liu Q., Lu Y., Li M., 2020. A synopsis of farmland abandonment and its driving factors in Nepal. *Land* 9(3): 84. DOI 10.3390/land9030084.

Correia M.C.L., 2012. *Reproductive biology of Australian acacias in Portugal*. Master Thesis, FCTUC, Portugal.

Couto P., 2004. Factorial analysis applied to landscape metrics defined in FRAGSTATS. *Investigação Operacional* 24(1): 109–137.

Crawford C.L., Yin H., Radeloff V.C., Wilcove D.S., 2022. Rural land abandonment is too ephemeral to provide major benefits for biodiversity and climate. *Science Advances* 8(21): eabm8999. DOI 10.1126/sciadv.abm8999.

Cuke M., Srivastava D.S., 2016. Divergent effects of tropical forest fragmentation and conversion on leaf litter decomposition. *Landscape Ecology* 31(5): 1037–1050. DOI 10.1007/s10980-015-0316-z.

Debuse V.J., King J., House, A.P.N., 2007. Effect of fragmentation, habitat loss and within-patch habitat characteristics on ant assemblages in semi-arid woodlands of eastern Australia. *Landscape Ecology* 22: 731–745. DOI 10.1007/s10980-006-9068-0.

Diogo V., Koomen E., 2012. Land-use change in Portugal, 1990–2006: main processes and underlying factors. *Cartographica: The International Journal for Geographic Information and Geovisualization* 47(4). DOI 10.3138/carto.47.4.1504.

Echeverria C., Coomes D.A., Hall M., Newton A.C., 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. *Ecological Modeling* 212: 439–449. DOI 10.1016/j.ecolmodel.2007.10.045.

Gashaw T., Tulu T., Argaw M., Worqlul A.W., Tolessa T., Kindu M., 2018. Estimating the impacts of land use/land cover changes on ecosystem service values: the case of the Andassa watershed in the Upper Blue Nile basin of Ethiopia. *Ecosystem Services* 31: 219–228. DOI 10.1016/j.tfp.2022.100212.

Gibson C.C., Ostrom E., Anh T.K., 2000. The concept of scale and the human dimensions of global change: a survey. *Ecological Economics* 32: 217–239. DOI 10.1016/S0921-8009(99)00092-0.

Gil A.M.C., 2017. *Controlo de acácias invasoras em sítios de importância comunitária: do planeamento aos desafios da intervenção*. Master Thesis, Coimbra Superior School of Agriculture, Porugal.

Grau R., Kuemmerle T., Macchi L., 2013. Beyond 'land sparing versus land sharing': environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. *Current Opinion in Environmental Sustainability* 5(5): 477–483. DOI 10.1016/j.cosust.2013.06.001.

Grimm N.B., Faeth S.H., Golubiewski N.E., Redman C.L., Wu J., Bai X., Briggs J.M., 2008. Global change and the ecology of cities. *Science* 319: 756–760. DOI 10.1126/science.1150195.

Hartter J., Southworth J., 2009. Dwindling resources and fragmentation of landscapes around parks: Wetlands and forest patches around Kibale National Park, Uganda. *Landscape Ecology* 24: 643–656. DOI 10.1007/s10980-009-9339-7.

Herrera J.M., Garcia D., 2010. Effects of forest fragmentation on seed dispersal and seedling establishment in ornithochorous trees. *Conservation Biology* 24(4): 1089–1098. DOI 10.1111/j.1523-1739.2010.01459.x.

Holling C.S., 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs* 62: 447–502.

Ihse M., 1996. *Landscape analysis in the Nordic countries*. Swedish Council for Planning and Coordination of Research, Stockholm.

INE [Instituto Naciona de Estatistica], 1947. *VIII Recenseamento geral da população no continente e ilhas adjacentes em 12 de dezembro de 1940*. Imprensa Nacional de Lisboa, Lisboa: 1947. Online: www.ine.pt/xurl/pub/66774710 (accessed 28 July 2022).

INE [Instituto Nacional de Estatistica], 2012. *Censos 2011. XV Recenseamento Geral da População: V Recenseamento Geral da Habitação*. Resultados definitivos. Lisboa, Portugal.

Jenkins M., Schaap B., 2018. *Forest ecosystem services. Background analytical study 1. Background study prepared for the thirteenth session of the United Nations Forum on Forests*, 41p. Online: https://www.un.org/esa/forests/wp-content/uploads/2018/05/UNFF13_BkgdStudy_ForestsEcoServices.pdf (accessed 12 August 2020).

Kremen C., Williams N.M., Aizen M.A., Gemmill-Herren B., LeBuhn G., Minckley R., Packer L., Potts S.G., Roulston T., Steffan-Dewenter I., Vázquez D.P., Winfree R., Adams L., Crone E.E., Greenleaf S.S., Keitt T.H., Klein A.M., Regetz J., Ricketts T.H., 2007. Pollination and other ecosystem services produced by mobile organisms: A conceptual framework for the effects of land-use change. *Ecology Letters* 10: 299–314. DOI 10.1111/j.1461-0248.2007.01018.x.Kuemmerle T., Levers C., Erb K., Estel S., Jepsen M.R., Müller D., Plutzar C., Stürck J., Pieter J., Verkerk P.J., Verburg-P. H., Reenberg A., 2016. Hotspots of land use change in Europe. *Environmental Research Letters* 11(6): 1–14. DOI 10.1088/1748-9326/11/6/064020.

Laurance WF, Nascimento HEM, Laurance SG, Andrade A, Ewers RM, Harms KE, Luizão RCC, Ribeiro JE., 2007. Habitat Fragmentation, Variable Edge Effects, and the Landscape-Divergence Hypothesis. *PLoS ONE* 2(10): e1017. DOI 10.1371/journal.pone.0001017.

Lasanta T., Arnáez J., Errea M.P., Ortigosa L., Ruiz-Flaño P., 2009. Mountain pastures, environmental degradation, and landscape remediation: the example of a Mediterranean policy initiative. *Applied Geography* 29(3): 308–319. DOI 10.1016/j.apgeog.2008.09.006.

Lasanta T., Nadal-Romero E., Arnáez J., 2015. Managing abandoned farmland to control the impact of re-vegetation on the environment. The state of the art in Europe. *Environmental Science & Policy* 52: 99–109. DOI 10.1016/j.envsci.2015.05.012.

Li Y., Zhang X.X., Mao R.L., Yang J., Miao C.-Y., Li Z., Qiu Y.-X., 2017. Ten years of landscape genomics: challenges and opportunities. *Frontiers in Plant Science* 8: 2136. DOI 10.3389/fpls.2017.02136.

Liu H., Weng Q., 2013. Landscape metrics for analysing urbanization-induced land use and land cover changes. *Geocarto International* 28(7): 582–593. DOI 10.1080/10106049.2012.752530.

Liu J., Taylor W. (eds), 2002. Integrating landscape ecology into natural resource management (Cambridge Studies in Landscape Ecology). Cambridge University Press, Cambridge. DOI 10.1017/CBO9780511613654.

Marchante H., Marchante E., Freitas H., Hoffmann, J.H., 2015. Temporal changes in the impacts on plant communities of an invasive alien tree: Acacia longifolia. *Plant Ecology* 216: 1481–1498.

Marta-Pedroso C., Gama I., Laporta L., Domingos T., 2014. *Mapping and assessing ecosystem services in Portugal: study of the economics of ecosystems and biodiversity for the S. Mamede Natural Park*. Instituto da Conservação da Natureza e Florestas, I.P. Instituto Superior Técnico, Lisboa, Portugal.

Mateus P., Fernandes P.M., 2014. Forest fires in Portugal: dynamics, causes and policies. In: Reboredo F. (ed.), *Forest context and policies in Portugal, present and future challenges*, Vol. 19. World Forests book series (WFSE, volume 19) Springer: 97–115.

Meneses B.M., Reis E., Pereira S., Vale M.J., Reis R., 2017. Understanding driving forces and implications associated with the land use and land cover changes in Portugal. *Sustainability* 9(351). DOI 10.3390/su9030351.

Midha N., Mathur P.K., 2010. Assessment of forest fragmentation in the conservation priority Dudhwa landscape, India using FRAGSTATS computed class level metrics. *Journal of the Indian Society of Remote Sensing* 38: 487–500. DOI 10.1007/s12524-010-0034-6.

Mitchell M.G.E., Bennett E.M., Gonzalez A., 2013. Linking landscape connectivity and ecosystem service provision: current knowledge and research gaps. *Ecosystems* 16: 894–908. DOI 10.1007/s10021-013-9647-2.

Moreira F., Silva J.P., Estanque B., Palmeirim J.M., Lecoq M., Pinto M, Leitão D., Alonso I, Pedroso R., Santos E., Catry T., Silva P., Henriques H., Delgado A., 2012. Mosaic-level inference of the impact of land cover changes in agricultural landscapes on biodiversity: a case-study with a threatened grassland bird. *PLoS ONE* 7(6): e38876. DOI 10.1371/journal.pone.0038876.

Morris D.W., Diffendorfer J.E., Lundberg P., 2004. Dispersal among habitats varying in fitness: reciprocating migration through ideal habitat selection. *Oikos* 107: 559–575. DOI 10.1111/j.0030-1299.2004.12894.x.

Nadal Romero E., Cammeraat E., Pérez-Cardiel E., Lasanta, T., 2016. Effects of secondary succession and afforestation practices on soil properties after cropland abandonment in humid Mediterranean mountain areas. *Agriculture, Ecosystems & Environment* 228: 91–100. DOI 10.1016/j.agee.2016.05.003.

Nesheim I., Reidsma P., Bezlepkina I., Verburg R., Abdeladhim M.A., Bursztyn M., Chen L., Cissé Y., Feng S., Gicheru P., Jochen König H., Novira N., Purushothaman S., Rodrigues-Filho S., Sghaier M., 2014. Causal chains, policy trade-offs and sustainability: analysing land (mis) use in seven countries in the South. *Land Use Policy* 37: 60–70. DOI 10.1016/j.landusepol.2012.04.024.

Nunes A.N., Almeida A.C., Coelho C.O.A., 2011. Impacts of land use and cover type on runoff and soil erosion in a marginal area of Portugal. *Applied Geography* 31(2): 687–699. DOI 10.1016/j.apgeog.2010.12.006.

Nunes A., 2008. *Land abandonment in the "Beira Transmontana"*. Iberografias 13, Ed. Campo das Letras SA, Portugal.

Nunes A., Lourenço L., Meira A.C., 2016. Exploring spatial patterns and drivers of forest fires in Portugal (1980–2014). *Science of the Total Environment* 573: 1190–1120. DOI 10.1016/j.scitotenv.2016.03.121.

Nunes, A., Figueiredo, A., Almeida, A.C., 2013. Changes in composition and threats to the sustainable management of woodlands in Portugal. In: Manzanares L. (eds), *Wood-*

lands: structure, species diversity and sustainable management. Nova Science Publisher, New York: 1–41.

Nurwanda A., Zain A.F.M., Rustiadi E., 2016. Analysis of land cover changes and landscape fragmentation in Batanghari Regency, Jambi Province. *Procedia – Social and Behavioral Sciences* 227: 87–94. DOI 10.1016/j.sbspro.2016.06.047.

Ostrom E.A., 2009. General framework for analyzing sustainability of social-ecological systems. *Science* 325: 419–422. DOI 10.1126/science.1172133.

Patz J.A., Daszak P., Tabor G.M., Aguirre A.A., Pearl M., Epstein J., Wolfe N.D., Kilpatrick A.M., Foufopoulos J., Molyneux D., Bradley D.J., 2004. Unhealthy landscapes: policy recommendations on land use change and infectious disease emergence. *Environmental Health Perspectives* 112(10): 1047-A582. DOI 10.1289/ehp.6877.

Perpiña Castillo C., Coll Aliaga E., Lavalle C., Martínez Llario J.C., 2020. An assessment and spatial modelling of agricultural land abandonment in Spain (2015–2030). *Sustainability* 12(2): 560. DOI 10.3390/su12020560.

Pinto-Ledezma J.N., Rivero Mamani M.L., 2014. Temporal patterns of deforestation and fragmentation in lowland Bolivia: implications for climate change. *Climatic Change* 127: 43–54. DOI 10.1007/s10584-013-0817-1.

Pôças. I., Cunha. M., Pereira L.S., 2011. Remote sensing-based indicators of changes in a mountain rural landscape of Northeast Portugal. *Applied Geography* 31: 871–880. DOI 10.1016/j.apgeog.2011.01.014.

Putz S., Groeneveld J., Henle K., Knogge C., Martensen A.C., Metz, M., Metzger J.P., Ribeiro M.C., de Paula M.D., Huth A., 2014. Long-term carbon loss in fragmented Neotropical forests. *Nature Communications* 5: 5037. DOI 10.1038/ncomms6037.

Pysek P., Richardson M.R., 2010. Invasive species, environmental change and management, and health. *Annual Review of Environment and Resources* 35(1): 25–55. DOI 10.1146/annurev-environ-033009-095548.

Qi Z.F., Ye X.Y., Zhang H., Yu Z. L., 2014. Land fragmentation and variation of ecosystem services in the context of rapid urbanization: The case of Taizhou city, China. *Stochastic Environmental Research and Risk Assessment* 28: 843–855. DOI 10.1007/s00477-013-0721-2.

Qin Y., Yan H., Liu J., Dong J., Chen J., Xiao X., 2013. Impacts of ecological restoration projects on agricultural productivity in China. *Journal of Geographical Sciences* 23: 404–416. DOI 10.1007/s11442-013-1018-6.

Ramankutty N., Mehrabi Z., Waha K., Jarvis L., Kremen C., Herrero M., Rieseberg L.H., 2018. Trends in global agricultural land use: Implications for environmental health and food security. *Annual Review of Plant Biology* 69: 789–815. DOI 10.1146/annurev-arplant-042817-040256.

Reidsma P., Tekelenburg T., van den Berg M., Alkemade R., 2006. Impacts of land-use change on biodiversity: an assessment of agricultural biodiversity in the European Union. *Agriculture, Ecosystems & Environment* 114(1): 86–102. DOI 10.1016/j.agee.2005.11.026.

Robertson P.A., Mill A., Novoa A., Jeschke J.M., Essl F., Gallardo, B., Geist J., Jarić I., Lambin X., Musseau C., Pergl J., Pyšek P., Rabitsch W., von Schmalensee M., Shirley M., Strayer D.L., Stefansson R.A., Smith K., Booy O., 2020. A proposed unified framework to describe the management of biological invasion. *Biological Invasions* 22: 2633–2645. DOI 10.1007/s10530-020-02298-2.

Rodríguez-Echeverry J., Echeverría C., Oyarzún, C., Morales L., 2018. Impact of land-use change on biodiversity and ecosystem services in the Chilean temperate forests. *Landscape Ecology* 33: 439–453. DOI 10.1007/s10980-018-0612-5.

Salvati L., Ferrara C., Mavrakis A., Colantoni A., 2016. Toward forest “sprawl”: monitoring and planning a changing landscape for urban sustainability. *Journal of Forestry Research* 27: 175–184. DOI 10.1007/s11676-015-0144-7.

Schröter D., Cramer W., Leemans R., Prentice I.C., Araújo M.B., Arnell N.W., Bondeau, A., Bugmann H., Carter T.R., Gracia C.A., de la Vega-Leinert A.C., Erhard M., Ewert F., Glendining M., House J. I., Kankaanpää S., Klein R.J.T., Lavorel S., Lindner M., Metzger M.J., Meyer J., Mitchell T.D,, Reginster I., Rounsevell M., Sabaté S., Sitch S., Smith B., Smith J., Smith P., Sykes M.T., Thonicke K., Thuiller W., Tuck G., Sönke Zaehle S., Zierl B., 2005. Ecosystem service supply and vulnerability to global change in Europe. *Science* 310(5752): 1333–1337. DOI 10.1126/science.1115233.

Skånes H.M., Bunce R.G.H., 1997. Directions of landscape change (1741–1993) in Virestad, Sweden — characterised by multivariate analysis. *Landscape and Urban Planning* 38(1–2): 61–75. DOI 10.1016/S0169-2046(97)00019-4.

Song W., Deng X., 2017. Land-use/land-cover change and ecosystem service provision in China. *Science of the Total Environment* 576: 705–719. DOI 10.1016/j.scitotenv.2016.07.078.

Tasser E., Georg Leitinger G., Tappeiner U., 2017. Climate change versus land-use change — what affects the mountain landscapes more? *Land Use Policy* 60: 60–72. DOI 10.1016/j.landusepol.2016.10.019.

Temesgen H., Nyssen J., Zenebe A., Haregeweyn N., Kindu M., Lemenih M., Haile M., 2013. Ecological succession and land use changes in a lake retreat area (Main Ethiopian Rift Valley). *Journal of Arid Environments* 91: 53–60. DOI 10.1016/j.jaridenv.2012.12.001.

Thrush, S.F., Schneider, D.C., Legendre, P., Whitlatch, R.B., Dayton, P.K., Hewitt, J.E., Hines, A.H., Cummings, V.J., Lawrie, S.M., Grant, J., Pridmore, R.D., Turner, S.J., McArdle B.H., 1997. Scaling-up from experiments to complex ecological systems: where to next? *Journal of Experimental Marine Biology and Ecology* 216: 234–254.

Turner M.G., Virginia H.D., Gardner R.H., 1989. Predicting across scales: theory development and testing. *Landscape Ecology* 3(3/4): 245–252. DOI 10.1007/BF00131542.

Turner M.G., Gardner R.H., 2015. *Landscape ecology in theory and practice*. Pattern and Process. Springer-Verlag, New York.

United Nations, 1992. *Agenda 21*. United Nations Conference on Environment & Development Rio de Janerio, Brazil. Online: sustainabledevelopment.un.org/content/documents/Agenda21.pdf (accessed 6 October 2022).

Verburg P.H., van Berkel D.B., van Doorn A.M., van Eupen M., van den Heiligenberg H.A. R.M., 2010. Trajectories of land use change in Europe: a model-based exploration of rural futures. *Landscape Ecology* 25: 217–232. DOI 10.1007/s10980-009-9347-7.

Vilá M., Weiner J., 2004. Are invasive plant species better competitors than native plant species? – Evidence from pair-wise experiments. *Oikos* 105: 229–238. DOI 10.1111/j.0030-1299.2004.12682.x.

Vuorela N., 2000. Can data combination help to explain the existence of diverse landscapes? *Fennia* 178: 55–80.

Wade T.G., Wickham J.D., Zacarelli N., Riitters K.H., 2009. A multi-scale method of mapping urban influence. *En-*

vironmental Modelling & Software 24(10): 1252–1256. DOI 10.1016/j.envsoft.2009.03.006.

Wang J., Yang X. A., 2012. A hierarchical approach to forest landscape pattern characterization. *Environmental Management* 49: 64–81. DOI 10.1007/s00267-011-9762-9.

Williamson M., 1999. Invasions. *Ecography* 22: 5–12. DOI 10.1111/j.1600-0587.1999.tb00449.x.

Wu J.G., Guo X.C., Yang J., Qian G.X., Niu J.M., Liang C.Z., Zhang Q., Li A., 2014. What is sustainability science? *Chinese Journal of Applied Ecology* 25(1): 1–11.

Zhang D., Qingxu H., Chunyanghe Q., Wu J., 2017. Impacts of urban expansion on ecosystem services in the Beijing-Tianjin-Hebei urban agglomeration, China: a scenario analysis based on the shared socioeconomic pathways. *Resources, Conservation and Recycling* 125: 115–130. DOI 10.1016/j.resconrec.2017.06.003.

Zhang L., Lu D., Li Q., Lu S., 2018. Impacts of socioeconomic factors on cropland transition and its adaptation in Beijing, China. *Environmental Earth Sciences* 77: 575. DOI 10.1007/s12665-018-7740-8.

Zipperer W.C., Foresman T.W., Walker S.P., Daniel C.T., 2012. Ecological consequences of fragmentation and deforestation in an urban landscape: a case study. *Urban Ecosystem* 15(3): 533–544. DOI 10.1007/s11252-012-0238-3.