Long-term bio-cultural heritage: Exploring the intermediate disturbance hypothesis in agro-ecological landscapes (Mallorca, c. 1850-2012)

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Abstract

We applied an intermediate disturbance-complexity approach to the land-use change of cultural landscapes in the island of Mallorca from c. 1850 to the present, which accounts for the joint behaviour of human appropriation of photosynthetic capacity used as a measure of disturbance, and a selection of land metrics at different spatial scales that account for ecological functionality as a proxy of biodiversity. We also delved deeper into local land-use changes in order to identify the main socioeconomic drivers and ruling agencies at stake. A second degree polynomial regression was obtained linking socio-metabolic disturbance and landscape ecological functioning (jointly assessing landscape patterns and processes). The results confirm our intermediate disturbance-complexity hypothesis by showing a hump-shaped relationship where the highest level of landscape complexity (heterogeneity-connectivity) is attained when disturbance peaks at 50-60%. The study proves the usefulness of transferring the concept of intermediate disturbance to Mediterranean cultural landscapes, and suggests that the conservation of heterogeneous and well-connected land-use mosaics with a positive interplay between intermediate level of farming disturbances and land-cover complexity endowed with a rich bio-cultural heritage will preserve a wildlife-friendly agro-ecological matrix that is likely to house high biodiversity.

Keywords

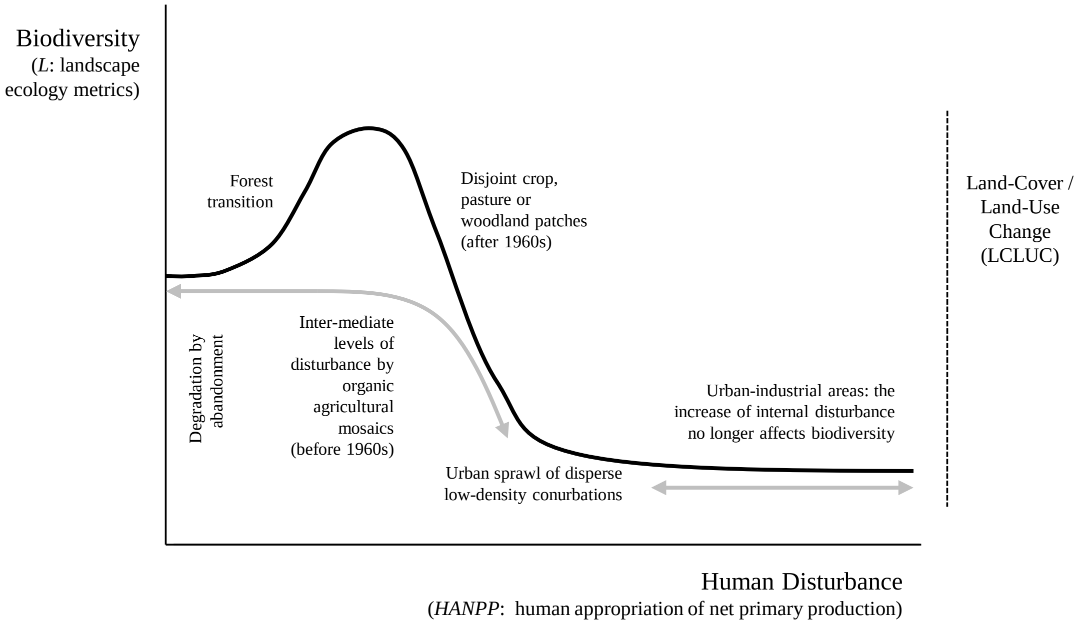
Cultural landscapes · Bio-cultural heritage · Disturbance ecology · Human Appropriation of Net Primary Production · Socio-ecological transition · Biodiversity conservation

1. Introduction

Biodiversity has been related to the existence of intermediate disturbances in ecosystems for a long time. Despite the intense debate raised by its detractors (Wilkinson 1999; Fox 2013; Sheil and Burslem 2013; Pierce 2014; Huston 2014), the intermediate disturbance hypothesis (IDH) is used in a growing number of scientific research (Svensson et al. 2012). Yet, since its introduction (Connell 1978) the IDH has hardly been applied to the socio-natural interplay or to study agricultural landscapes.

Assuming that agro-ecosystems are the result of energy flows and knowledge that farmers invest in a land matrix, the biodiversity associated to cultural landscapes (Altieri 1999) can be related on the one hand to their own complexity, and on the other hand to the degree of disturbance they exert upon natural systems. Traditional agro-ecological landscapes are endowed with an age-old bio-cultural heritage accumulated by rural communities that experienced a long-lasting joint adaptation with nature. Their maintenance are indissolubly tied to the practical knowledge handed down from one generation of farmers, shepherds and lumberjacks to the next, a complex set of ingenious techniques and local know-how that have contributed to historically compound this cultural and biological legacy. As a result, the complexity of cultural landscapes diminishes either when the farming intervention is intensified beyond a certain threshold in industrial monocultures, or abandoned (Fig. 1). Both may entail a process of landscape deterioration and biodiversity loss (Farina 2000; Antrop 2005; Agnoletti 2014).

Figure 1 Long-term bio-cultural heritage. Conceptual scheme of the Intermediate Disturbance Hypothesis (IDH) in a Mediterranean cultural landscape context.



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We have started to develop an intermediate disturbance-complexity (IDC) model of cultural landscapes (Marull et al. 2015a) using a multi-scalar experimental design in the island of Mallorca, at the core of the Mediterranean biodiversity hotspot (Myers et al. 2000), taking as a natural experiment the Land-Cover and Land-Use Change (LCLUC) from c.1850 to 2012. The main results of this LCLUC and their impact on landscape ecology are presented in this article. In this section we expose the aims and background of our research. Section two presents the case study and methods used. Section three discusses the results obtained and suggests a few hypotheses on the economic driving forces and socio-political agencies behind. Section four concludes.

1.1. Cultural landscapes in a globally changing world

Cultural landscapes are the historical outcome of interactions between socioeconomic and biophysical spatial patterns and metabolic flows (Wrbka et al. 2004; Liu et al. 2007; Rindfuss et al. 2008). Four decades ago pioneering work on the energy analysis of agro-ecosystems revealed a substantial decline in energy throughputs of contemporary farming, brought about by the consumption of fossil fuels and other external inputs (Odum 1984, 2007; Giampietro et al. 2011; Pelletier et al. 2011). More recently, several studies are reassessing the role traditional agrarian knowledge and practices have played to create complex-heterogeneous landscapes whose legacy is increasingly praised for its role in biological conservation (Tress et al. 2001; Kumaraswamy and Kunte 2013; Hong et al. 2014). Yet, the role of energy and material flows (Haberl 2001) as driving forces of contemporary LCLUC is still a pending research issue (Peterseil et al. 2004). We aim to contribute to the IDH research by exploring the relationships between socio metabolic impact as a proxy of human pressure, and landscape metrics that account for ecological functionality, applied to a multi-scalar analysis of LCLUC throughout socio-ecological transitions (Fischer-Kowalski and Haberl 2007; González de Molina and Toledo 2014).

LCLUC is a global factor of biodiversity loss that poses significant land-use policy questions (Schroter et al. 2005; Young et al. 2014), and challenges scientific research to develop better models and indicators (De Groot 2006; Turner et al. 2007; Haines and Young 2009). In turn, landscape ecology provides quantitative tools to characterize landscapes (Turner and Ruscher 1988; Li 2000) and land-use change (Reed et al. 1996) by linking ecological patterns and processes (Tischendorf 2001; Helming et al. 2007; Verburg et al. 2009). However a considerable disagreement still remains on whether the removal of human intervention in landscapes undergoing an abandonment process results in a positive impact on biodiversity conservation (as seen from a land sparing or a forest transition approach) or rather a negative one (as seen from a land sharing and a wildlife-friendly farming approach) (Green et al 2005; Matson and Vitousek 2006; Bengston et al. 2003; Fischer et al. 2008; Perfecto and Vandermeer 2010; Tscharntke et al. 2012). According to Robson and Berkes (2011), land-use decline may result in a loss of agro-forest mosaics and to local biodiversity decrease. A meta-analysis made by Plieninger et al. (2014) founds some patterns linking biodiversity and land abandonment in the Mediterranean, but they seem too complex to draw definite conclusions.

Exploring this bio-cultural interface is an exciting and pressing scientific challenge (Phalan et al. 2011) that calls for a better understanding on how farm systems affect the relationship between farming land-uses, biological primary productivity and landscape functionality. A useful indicator is the Human Appropriation of Net Primary Production (*HANPP*), a top-level indicator of environmental pressure (Vitousek et al. 1986; Haberl et al. 2007; Krausmann et al. 2013) that can assess the impact of farming on biodiversity (Firbank et al. 2008) according to the species-energy hypothesis (Hawkins et al. 2003). Although mathematical modelling suggests that the output of ecosystem services generally peaks at some intermediate level of LCLUC intensity (Braat and ten Brink 2008), this is rather complex interplay. Schwartz et al. (2000) found little support to establish a linear relationship between biodiversity and ecosystem functioning (i.e., biomass, nutrient cycling, etc.), while Balvanera et al. (2006) suggested the contrary from a meta-analysis on different biodiversity components that corroborate the basic scientific consensus and the remaining uncertainties on the subject (Hooper et al. 2005).

We consider that simple gradients of LCLUC are unable to explain the variations in biodiversity, unless the functional ecological complexity of landscapes is taken into account (Opdam et al. 2006; Pino and Marull 2012; Marull et al. 2014, 2015b). It is known that landscape heterogeneity arises in nature as one among many looping ways through which energy dissipation leads to the formation of self-organized structures, able to perform a historical succession ruled by adaptive selection (Morowitz 2002). When humans increase the dissipated energy up to a critical point, complexity is reduced and environmental degradation ensues (Ulanowicz 1997). In complex agro-ecosystems, instead, the storage of energy and information at some points reduces internal entropy thanks to the exploitation of other spaces of lower complexity but larger production within a joint encompassing structure (Margalef 2006). As in other living organisms, these heterogeneous space-time structures may allow keeping more mature organized spaces linked together with simpler productive ones within an interdependent set of patterns and flows able to provide resilience to the system (Ho and Ulanowicz 2005).

1.2. Disturbance ecology in cultural landscapes

The intermediate disturbance hypothesis (IDH) is a non-equilibrium explanation to understand the maintenance of biodiversity in ecosystems (Wilson 1990). Yet, there is considerable debate around which are the mechanisms that promote coexistence among species (Padisak 1994; Dial and Roughgarden 1998; Buckling et al. 2000; Sheil and Burslem 2003; Miller at al. 2012; Fox 2013; Huston 2014). There are different definitions of disturbance (van der Maarel 1993), but a common one is the destruction (or harvest) of biomass (Calow 1987) leading to the opening up of space and resources for recolonizing species—an approach that foregrounds the variation of its spatial extent in ecosystem communities (Wilson 1994). The earliest version by Hutchinson (1951) already considered disturbance intensity in a spatial context, that led to the idea of a humped-shaped trend later introduced by Horn (1975) and further amplified by Connell (1978). Coexistence would require spatially patchy disturbance that leads to a trade-off between species able to perform best at different stages of post-disturbance succession (Chesson and Huntly 1997). At intermediate disturbance frequencies both competitive and dispersal species may coexist (Roxburgh et al. 2004; Shea et al. 2004; Barnes et al. 2006). Wilson (1994) labelled it a between-patch mechanism (Collins and Glenn 1997), which has been renamed as a succession-mosaic hypothesis that views disturbances as events that alter niche opportunities (Shea and Chesson 2002).

Whereas IDH has been evaluated by mathematical modelling (Petraitis 1989), and widely supported in studies of terrestrial (Molino and Sabatier 2001), freshwater (Padisak 1993) and marine communities (Johst et al. 2006), it has been seldom used in agro-ecosystem so far (Gliessman 1990, Fahrig and Jonsen 1998; Sasaki et al. 2009). Yet, if IDH holds true in natural ecosystems, it should play a similar role in the interplay of human activity with ecological processes (Farina 2000). Agro-forest mosaics offer habitats to different species, creating a greater amount of ecotones which in turn provide opportunities to edge species (Benton et al. 2003), as well as more permeable land-matrix allowing dispersion among local populations (Shreeve et al. 2004). Thanks to the edge effect and high connectivity, a complex land-cover pattern may host greater biodiversity than more uniform landscapes (Harper et al. 2005). Understanding and managing correctly these patchy agro-forest mosaics require an interdisciplinary approach to the bio-cultural diversity (Arts et al. 2012; Parrotta and Trosper 2012; Cocks and Wiersum 2014) embedded in agro-ecological landscapes (Antrop 2006; Matthews and Selman 2006; Blondel 2006; Verdasca et al. 2012).

In order to create and maintain agro-ecosystems, farmers have to continuously invest over the land matrix certain amounts of energy and information that shape the spatial patterns of an agro-ecological landscape embodied with a bio-cultural heritage (Marull et al. 2015c). The impact of this farming ecological disturbance (Margalef 2006) on biodiversity may be either positive or negative, depending on the intensity and shape of these socio-metabolic flows and the complexity of landscape mosaics (Altieri 1999; Swift et al. 2004; Cardinale et al. 2012).

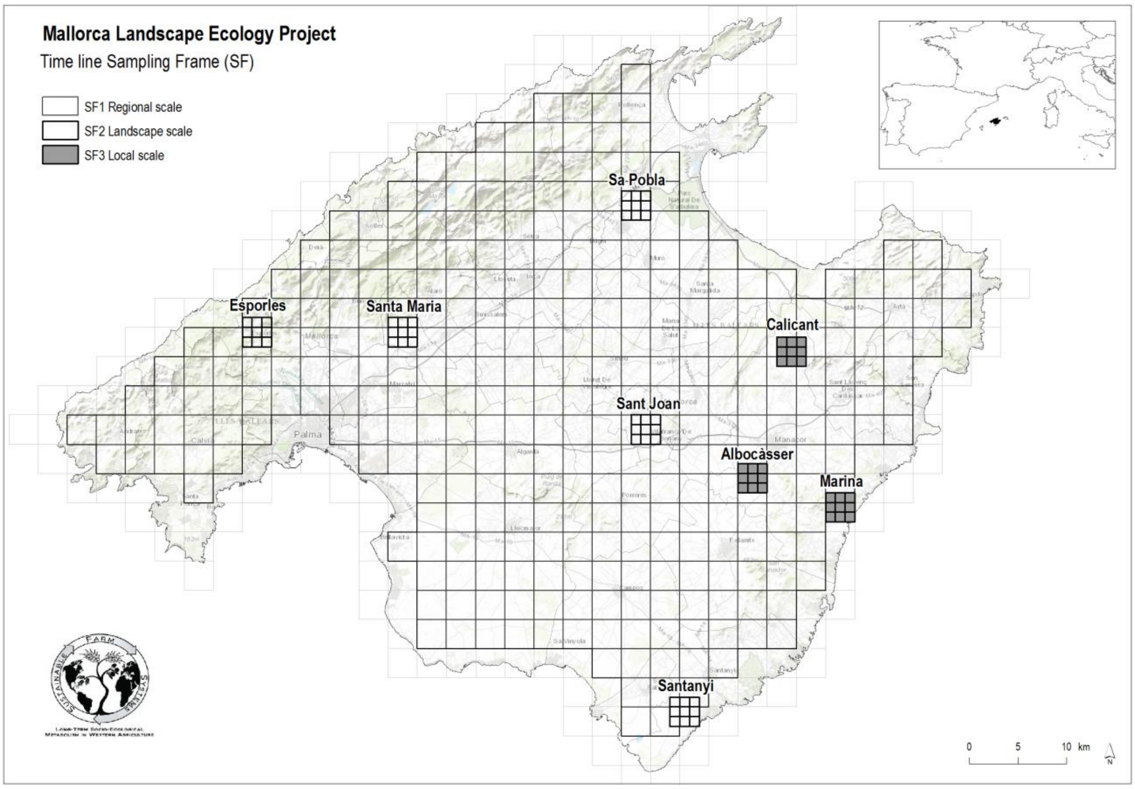
2. Materials and methods

2.1. A multi-scalar experimental design of the study area

In the Mediterranean World, wilderness was early disturbed by human action. Since Ancient times, farmers and shepherds have long shaped the land with agroforest and grazing mosaics (Grove and Rackhman 2003). The starting point of our case study is not from a pristine wilderness but a much transformed nature (Gil-Sánchez et al. 2002). The island of Mallorca, located in the Mediterranean Sea (Fig. 2), has an extension of 3,603 km^2 of calcareous origin. The coast combines sand beaches with cliffs raised by a mountain range that runs parallel to the North coast, the Serra de Tramuntana, and the eastward Serres de Llevant. Between them there is a great plain with a Mediterranean mild climate. Annual precipitation ranges from 300 mm in the South to 1,800 mm in the North, largely concentrated in winter, while the average annual temperature is around 16 ºC and peaks during the dry summers. The island vegetation, adapted to these agro-climatic features as well as to a long-lasting human intervention (Murray 2012), combines scrubland, pines and residual oak forests with a variety of annual crops (grains and vegetables) and arboriculture (olive groves, almonds, figs, carobs, vineyards).

There are six regions in Mallorca (Rullan 2002) with different traits (Fig. 2): i) *Tramuntana* comprises all the northern mountains, with an abrupt morphology and a rainfall of 1,400-1,800 mm a year (the 3x3 km^2 studied area is '*Esporles*' scene); ii) *Raiguer* is the piedmont between *Tramuntana* and the inland plane, whose soil, precipitation and edge character provide the best conditions for an intensive and diversified agriculture (the 3x3 km^2 study area is '*Santa Maria*' scene; next to *Raiguer* we find '*Sa Pobla*' scene characterized by its drying works of wetlands and watering intensification); iii) The *Pla* is a central plane where cereal crops have been most cultivated (we take the 3x3 km^2 '*Sant Joan*' scene); iv) *Llevant* is located eastward and combines relative small elevations with valleys that contribute to its rich landscape diversity, representative of all Mallorca landscapes, including flat grain-growing zones, agro-forest mosaics in the hills and areas of shallow soil and arid vegetation (we set three 3x3 km^2 scenes: '*Albocàsser*', quite similar to '*Sant Joan*'; '*Calicant*', similar to '*Esporles*'; and '*Marina*' similar to the *Migjorn* region); v) *Migjorn*, in the Southeast, is the driest region with barren land with shrubs that hinders agriculture (the 3x3 km^2 scene is '*Santanyí*').

Figure 2 Location of the Mallorca case study performed at three scales: SF-1 (1:50,000), SF-2 (1:5,000), SF-3 (1:500).



Source: our own.

This set of scenes allows us to gain in-depth insights that might be lost in the broader view of the whole island. In order to test the relationship between *HANPP* and ecological patterns and processes taking place in these cultural landscapes, we used the following multi-scalar experimental design: 1) regional scale (SF-1; 1:50,000) takes into account the entire island divided into 3x3 km^2 cells (Fig. 2), and to avoid the sea edge effect the analysis area is limited to 331 inland cells studied in three time points

(1956, 1973, 2000) using land-cover digital cartography (GIST, 2009); 2) landscape scale (SF-2; 1:5,000) takes into account eight 3x3 km^2 analysis scenes distributed in five agro-ecological regions of Mallorca divided into nine 1x1 km^2 cells (Fig. 2), so as to have a better approximation to the landscape transitions along three time points (1956, 1989 and 2010); and 3) local scale (SF-3; 1:500) takes into account three 3x 3km^2 analysis scenes (Fig. 2) in the *Llevant* region, as a representative sample of Mallorca landscapes, dividing each scene into 36 cells of 0.5 x 0.5 km^2 and extending backwards the time frame from the 1850s to 1956 and 2012 using land-cover cartography digitized from historical land-use maps.

This multi-scalar dataset will be used to test in Mallorca the hypothesis that landscape heterogeneity in a well-connected land matrix could potentially host greater biodiversity than in the more uniform land-covers we tend to have at present. This hypothesis has already been tried out for different species and ecosystems (Bengtsson et al. 2003; Tscharnkte et al. 2012; Gabriel et al. 2013). The novelty is to apply this to cultural landscapes, by adopting a bio-cultural approach that relates the farming disturbance exerted through *HANPP* to the landscape ecology assessment of land-use patterns.

2.2. Assessing HANPP and land-cover change at three different scales

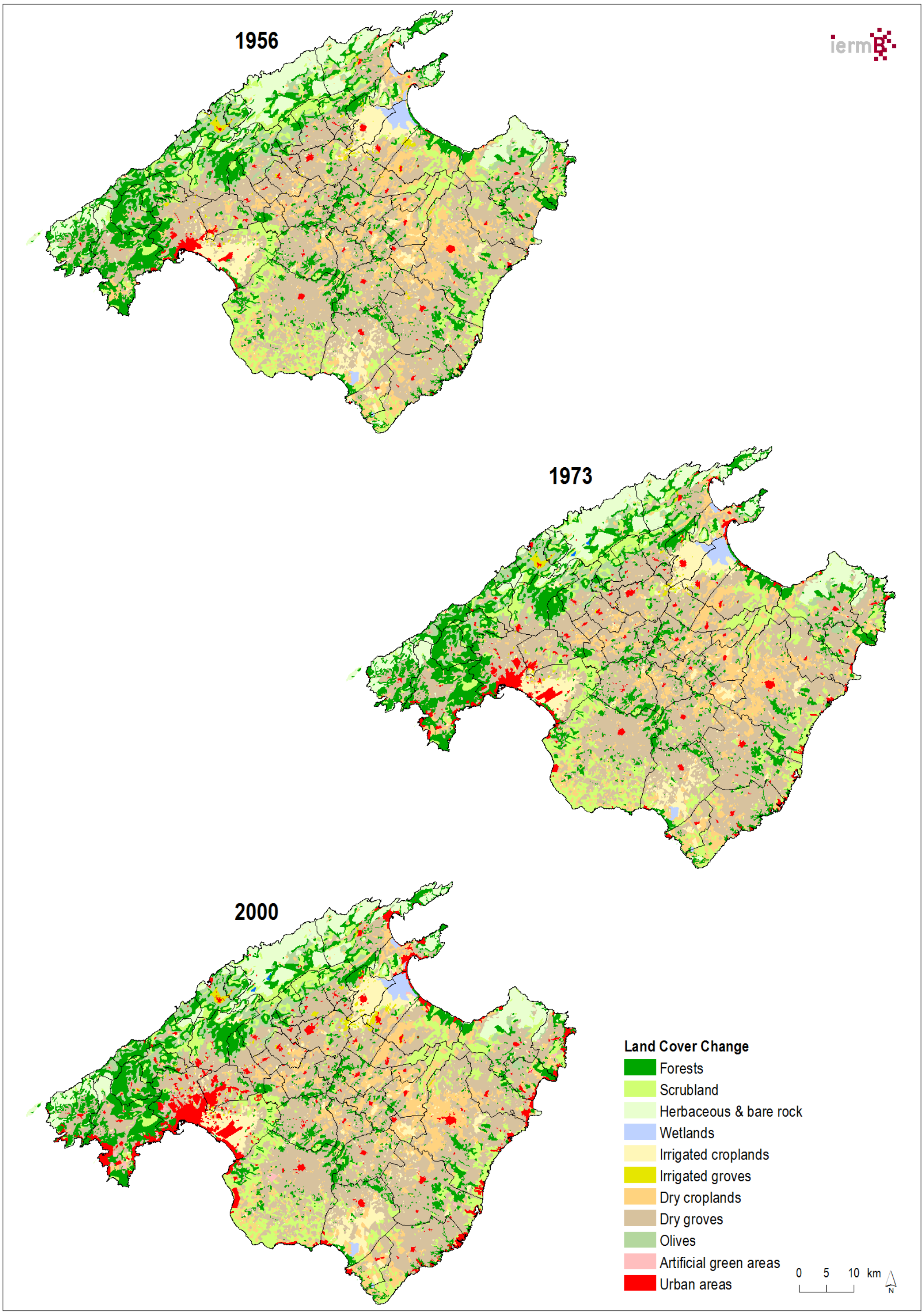
Based on the digital maps available for the whole island in 1956, 1973, 1989 and 2000 provided by GIST (2009), we have analysed the historical shifts in land-cover patterns of the study area (SF-1; Fig. 3) by using the metrics listed and explained in Table 1. Also relying on photointerpretation of the landscape scenes (SF-2; Fig. 6), we analysed in 1956, 1989 and 2011 the ecological landscape patterns listed and explained in Table 2. After digitising some of the cadastral land-use maps available at local scale (SF-3; Fig. 8) from historical archives (Rosselló-Verger 1982), we analysed the corresponding shifts in land-use patterns calculated per parcel and/or within 0.5 x 0.5 km^2 sample cells for three study areas located in the Manacor municipality ('*Albocàsser*', '*Callicant*' and '*Marina*') c.1850, in 1956 and 2012 by using the metrics listed and explained in Table 3.

Table 1 Quantitative Agro-ecological Landscape Analysis. Metrics useful at regional scale (SF-1)[*].

Typology	Indicator	Description	Calculation
Land-cover Change[1]	*Main Land Cover (MLC)*	Measures the most representative land cover category in a sample cell.	Land cover category with more proportion of land matrix surface per each sample cell. Unit: category
	Land Cover Richness (LCR)	Measures the number of different land covers in a sample cell.	Number of land cover categories per each sample cell. Unit: number {1… 10}
Land-cover Structure[2,3]	*Shannon-Wiener Index (H')*[4]	Measures the land cover equi-diversity. H' increases as more land-cover categories with similar proportions build up the land-cover mosaic.	$H' = \sum_{i=1}^{c} (P_i * \ln P_i)$ Where P_i is the proportion of land matrix occupied by each type of land cover category *i* and *c* the number of categories within each sample cell. Unit: number {0… 1}
	Effective Mesh Size (MESH)[5]	Measures the inverse of the extent of fragmentation.	$MESH = \sum_{i=1}^{p} (A_i^2) * 1000 / \sum_{i=1}^{p} (A_i)$ Where A_i is the area of each land cover polygon *i* and *p* the number of polygons within each sample cell. Unit: km^2
Land-cover Functionality[6,7]	*Landscape Metric Index (LMI)*[8]	Based on the landscape's structure capacity -as affected by human activities- to support organisms and ecological processes.	$LMI = 1 + 9 (\gamma_i - \gamma_{min}) / (\gamma_{max} - \gamma_{min})$; $\gamma = I_1 + I_2 + I_3 + I_4$ Were γ_i is the sum of the indicators for each point in the region, while γ_{min} and γ_{max} are the minimum and maximum values, respectively, in the study area under consideration. I_1 = potential relation; I_2 = ecotonic contrast; I_3 = human impact; I_4 = vertical complexity. Unit: number: {1… 10}
	Ecological Connectivity Index (ECI)[9]	Assesses the functionality of the land matrix according to its ability to host and connect the horizontal flows of energy, matter and information which sustain biodiversity.	$ECI_a = \sum_{i=1}^{m} ECI_b / m$ Were ECI_a is the absolute ecological connectivity index, ECI_b is the basic ecological connectivity index for each ecological functional area (*EFA*) *i* and *m* is the number of *EFA* considered. $ECI_b = 10 - 9 \ln (1 + (x_i - x_{min})) / \ln (1 + (x_{max} - x_{min}))^3$ Were x_i is the adapted cost-distance value in a pixel, x_{max} are the maximum and x_{min} are the minimum adapted cost-distance values on a given area. Unit: number {0… 10}

Source: our own. Notes: [*]All variables are calculated on 3 x 3 km^2 inland sample cells (N = 331) for three time points (1956, 1973 and 2000); [1]Bender et al. (1998); [2]Forman (1995); [3]Fischer and Lindenmayer (2007); [4]Shannon (1948); [5]Jaeger, J. (2000); [6]Opdam et al. (2006); [7]Gilbert-Norton et al. (2010); [8]Marull et al. (2007); [9]Marull and Mallarach (2005).

Figure 3 Land-cover changes at regional scale (SF-1; 1:50,000) in 1956, 1973, 2000.



Source: our own, from GIST (2009).

Table 2 Quantitative Agro-ecological Landscape Analysis. Metrics useful at landscape scale (SF-2).

Typology	Indicator	Description	Calculation
Landscape Transitions	*Landscape Dynamics* (*LD*)	Measures the sample cell average of the landscape change of each pixel: 0 (no change); 1 (change).	$LD = \sum_{i=1}^{n} (C_i) / n$ Where C_i are pixels = 1 and n the total number of pixels (0, 1) in a given sample cell. Three stability regimes could be obtained: stable (*LD* = 0-0.2); semi-stable (*LD* = 0.2-0.4); non-stable (*LD* = 0.4-1). Unit: number {0… 1}
	Landscape Pressure (*LP*)	Measures the percentage of pixels that change from more ‘natural’ to more human modified landscape for each sample cell: 0 (no change); 1 (total change).	$LP = \sum_{i=1}^{n} (V_i) / n$ Where V_i is the value of ‘human pressure’ per pixel and n the total number of pixels in a given sample cell. Human pressure: low (*LP* = 0-0.25); medium (*LP* = 0.0.25-0.5); high (*LP* = 0.5-0.75); very high (*LP* = 0.75-1). [Human pressure values: 0 = forest, 0.1 = scrubland; 0.2 = grove land mixed with scrub; 0.3 = shelterbelts; 0.4 = homogeneous dry groves; 0.5 = heterogeneous dry groves; 0.6 = grassland; 0.7 = dry crops; 0.8 = irrigated groves; 0.9 = irrigated crops; 1 = urban areas]. Unit: number {0… 1}
Landscape Patterns[1]	*Landscape Core Area* (*LCA*)[*2]	Measures the sample cell average of the landscape unit core areas, which is an important quality of the appearance of inner species.	Maximum radius of the circle which can be drawn within the boundaries of similar landscape units per each sample cell. [Landscape units: ‘semi-natural’ (forest, scrubland, grove land mixed with scrubs); ‘dry groves’ (homogeneous and heterogeneous); dry crops; irrigated crops; grassland]. Unit: km
	Landscape Shape Complexity (*LSC*)[*]	Measures the sample cell average of the landscape shape complexity, which is an important quality of border species.	Relation between the area of the element and the area of the bounding rectangle per each sample cell. Unit: number
Landscape Naturalness	*Landscape Naturalness* (*LN*)	Measures the degree of preservation of the ‘pristine state’.	Sample cell average of the landscape naturalness: $LN = \sum_{i=1}^{n} (N_i) / n$ Where N_i is the value of ‘naturalness’ per pixel and n the total number of pixels in a given sample cell. [Naturalness levels: 1 = forest, 0.9 = scrubland; 0.8 = grove land mixed with scrub; 0.7 = shelterbelts; 0.6 = homogeneous dry groves; 0.5 = heterogeneous dry groves; 0.4 = grassland; 0.3 = dry crops; 0.2 = irrigated groves; 0.1 = irrigated crops; 0 = urban areas]. Unit: number {0… 1}
	Landscape Anthropogeneity (*LA*)[3]	Measures the extent to which landscapes are dominated by strongly human-altered systems.	$LA = \log 10 (U + A) / N$ Were U denotes urban area, A agricultural area, and N ‘natural’ or ‘semi-natural’ areas. Unit: number

Source: our own. Notes: [*]Analysis not presented in depth in this article; [1]Wrbka et al. (2004); [2]Forman and Godron (1986); [3]O’Neill et al. (1988).

Table 3 Quantitative Agro-ecological Landscape Analysis. Metrics useful at local scale (SF-3)*.

Typology	Indicator	Description	Calculation
Land-use Change[1]	*Land Use Change (LUC)***	Measures the cell average of the 'land use typology' change of each pixel: 0 (no change); 1 (change).	$LUC = \sum_{i=1}^{n} (\alpha_i) / n$ Where α_i are pixels = 1 and n the total number of pixels (0, 1) in a given sample cell. Three stability regimes could be obtained: stable (LUC = 0-0.2); semi-stable (LUC = 0.2-0.4); non-stable (LUC = 0.4-1). The land use change regressive LUC_r measures the change to urban land uses. The land use progressive LUC_p measures the change to 'natural' land uses. Unit: number {0… 1}
	Land Use Richness (LUR)	Measures the cell average of the number of 'land use categories' per parcel.	$LUR = \sum_{i=1}^{r} (\alpha_i) / p$ Where α_i is the number of land use categories per parcel and r the number of parcels in a given sample cell. Number of land use categories per parcel. Unit: number
	*Land Use Diversity (LUD)***[2]	Measures the probability of 'land use category' in a sample cell.	$LUD = 1 - \sum_{i=1}^{c} P_i^2$ Where P_i is the probability of the occurrence of the land use category i and c the number of categories within the sample cell. Calculated as Simpson Diversity Index. Unit: number
Land-use Structure[3]	*Largest Patch Index (LPI)*	Measures the parcel's grain thickness of the land matrix.	Surface of the largest parcel in each sample cell. Unit: km^2
	Edge Density (ED)	Measures the potential exchanges between 'land use typologies' (ecotony).	Total length of perimeters of the parcels with the same land use typology (dissolved) in relation to the surface area of the cell. Unit: km
	Polygon Density (PD)	Measures the parcel's (or 'land use typology') fragmentation.	Number of parcels of all the land uses taken together (or number of land use typology polygons). Unit: number
Parcel's Distribution	*Parcel Typology (PT)*	Measures the parcel's size for each land use typology	Parcel's size by land use typology. Unit: m^2
	Parcel Ownership (PO)	Measures the possessions distribution according parcel's size and land use	Number of owners by parcel's size and land use. Unit: number

Source: our own. Notes: *All variables were calculated per parcel and/or within 0.5 x 0.5 km^2 sample cells (N = 27) for three Manacor 'case study areas' in three time points (1850, 1956, 2012); **Analysis not presented in this paper; [1]Bender et al. (1998); [2]McGarigal and Marks (1994); [3]Forman (1995).

Our intermediate disturbance hypothesis *(*IDH*)* is based on variables that describe both spatial land pattern (Shannon-Wiener index, *H'*) and human disturbance (Human Appropriation of Net Primary Production, *HANPP*). We work with squared cells from land-unit (LU) maps, so that:

$$\sum_{i=1}^{k} p_i = 1$$

where p_i is the proportion of LU *i* in a specific cell, and *k* is the number of LU. We will refer to *p* as vector $p = (p_1, \ldots, p_k)$.

In order to check the IDH with the historical LU maps available, we analysed the corresponding change in the spatial pattern of the study area by using *H*' (Shannon 1948) that measures equi-diversity of LU in a cell:

$$H' = -\sum_{i=1}^{k} p_i \log_k p_i$$

where *k* is the total number of LU in the study area, and p_i is the proportion of LU *i* in a specific cell.

HANPP is used as a measure of disturbance, where *NPP* is the net amount of biomass produced by autotrophic organisms (green plants) that constitutes the main nutritional basis for all food chains over a year. *HANPP* measures the extent to which humans modify the amount of *NPP* available for other species, either by changing the land-covers or removing a share of *NPP* (Haberl et al. 2007; Krausmann et al. 2013). Hence, *HANPP* is calculated using the following identities:

$HANPP = \Delta NPP_{LU} + NPP_h$

$\Delta NPP_{LU} = NPP_0 - NPP_{act}$

where NPP_h is the *NPP* appropriation through harvest, and ΔNPP_{LU} is the change of *NPP* through human-induced land conversions. ΔNPP_{LU} is defined as the difference between the *NPP* of the potential (NPP_0), and actual (NPP_{act}) vegetation. *HANPP* is associated to each LU of the study area, so that *HANPP* is calculated multiplying a fixed coefficient (w_i) for some LU *i* by the surface occupied by this LU:

$$HANPP = \sum_{i=1}^{k} w_i p_i$$

where w_i denote the weight of LU *i*. Variations in *HANPP* not only depend on the variations of *p*, but on the variations of *w* as well. As a result we have spatially-explicit values of *H'* and *HANPP* for each cell measured on the same LU database. Taking as reference the work done by Schwarzlmüller (2009) on

Spain, these *HANPP* values have been estimated after assessing different *NPP* and harvested amounts (in tonnes of dry matter per LU and year).

In the work presented here bio-cultural diversity is represented in the land matrix and not in the species richness. Recent studies in Mediterranean cultural landscapes reveal that the conservation of heterogeneous and well-connected land matrix with a positive interplay between human disturbances and land-cover / land-use complexity are able to hold high species richness at regional scale (i.e. birds; Marull et al 2015b), landscape scale (i.e. orchids; Marull et al 2014) and local scale (i.e. butterflies; Marull et al 2015a). In order to test our hypothesis at the regional scale, we analyse a set of landscape ecology metrics as a function of *HANPP*. To do this, we obtain a new variable *L* ('Landscape Metrics' as a proxy of biodiversity) using Principal Components Analysis (PCA). Once we have *L*, we will perform a regression analysis with *HANPP* as the independent variable and *L* as the dependent.

3. Results and discussion

3.1. Land-cover dynamics at regional scale (SF-1)

Despite the seemingly low land-cover change seen from a regional view (Fig. 3), landscape metrics show a decrease from 1956 to 2000 as the joint result of urban sprawl, agricultural intensification and rural abandonment (Fig. 4). Urban areas (277%) and golf courses (1,796%) increased the most. Agricultural covers decreased, mainly in dry crops (-8.8%), dry groves (-4.3%) and olive trees (-9.6%). Shrubs (-3.6%), woodland (-4.5%) and wetlands (-5.2%) experienced a lesser decrease, while irrigated cropland grew 14.6% (Table 4).

Accordingly, the number of patch types per cell (*LCR*) tended to diminish. Land-cover richness (*H'*) measured by the number of different patch types and their proportional area distribution (richness and evenness), presented lower values as well—strongly correlated with *MESH* values as the inverse of fragmentation. *LMI* values confirm the progressive loss of landscape functional structure, thus lessening its capacity to support ecological processes and likely biodiversity. *ECI* values of landscape ecological connectivity also decreased (Fig. 4 and Fig. 5) due to the impact of new transport facilities and low-density urban developments. Urban sprawl has isolated woodland, cropland and natural protected areas one another, while the retreat of farming decreased landscape diversity and ecotones. Taken together these metrics indicate a loss in landscape heterogeneity that would ultimately lead to lesser biodiversity. Some critical areas for the potential ecological connectivity between protected natural areas and the remaining agricultural mosaics can be detected in Fig. 5, which should be preserved from the barrier effect of linear infrastructures and urban developments in future.

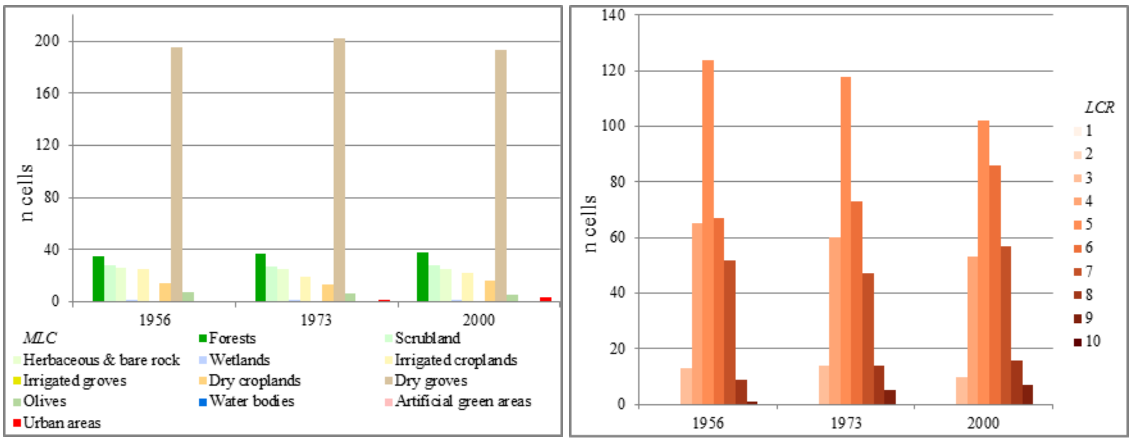
Table 4 Long-term Cultural Landscapes Analysis. Land-cover change (km^2) in Mallorca (1956, 1973, 1995, 2000).

Land-cover	1956	1973	1995	2000	1956-2000
Forest	574.01	569.77	549.88	547.94	-26.07
Scrubland	445.48	434.30	431.54	429.62	-15.86
Herbaceous & bare rock	275.07	276.60	279.43	280.07	5.00
Wetlands	25.34	24.61	24.02	24.02	-1.32
Irrigated cropland	173.70	161.37	174.04	173.36	-0.34
Irrigated groves	21.61	14.73	24.94	24.76	3.15
Dry cropland	436.35	412.85	401.28	398.12	-38.22
Dry groves	1,486.76	1,499.99	1,426.93	1,422.62	-64.15
Olives	136.03	131.30	123.01	123.01	-13.02
Water bodies	0.00	1.02	1.02	1,02	1.02
Artificial green areas	0.96	4.64	15.31	18,20	17.23
Urban areas	47.81	91.96	171.74	180.38	132.57
Total	3,623.13	3,623.13	3,623.13	3,623.13	-

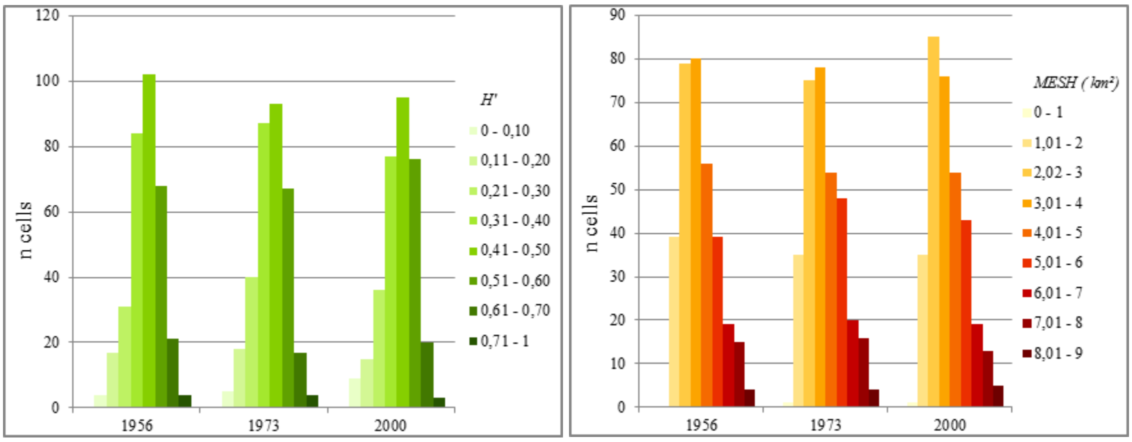
Source: our own, calculated from GIST (2009).

Figure 4 Metrics applied at regional scale (SF-1): Main Land Cover (*MLC*), Land Cover Richness (*LCR*), Shannon-Wiener Index (*H'*), Effective Mesh Size (*MESH*), Landscape Metric Index (*LMI*) and Ecological Connectivity Index (*ECI*) in 1956, 1973, 2000.

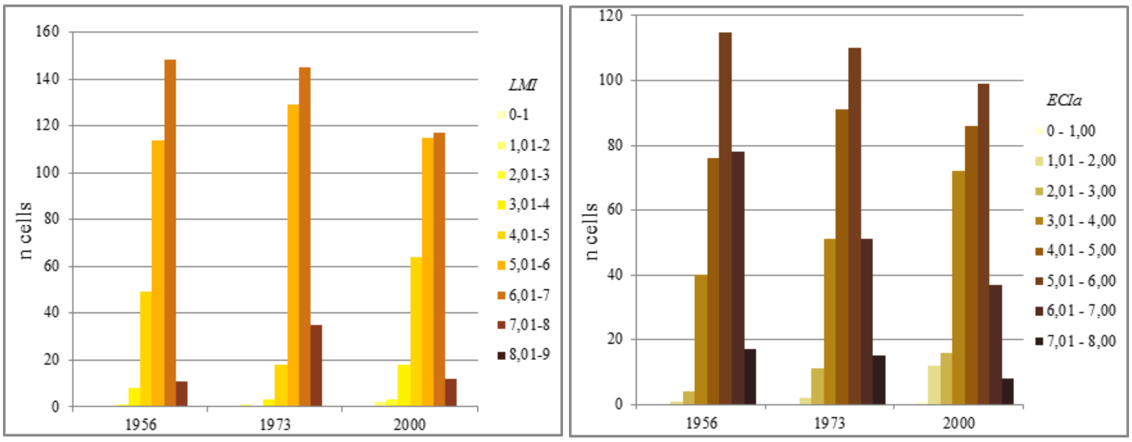
a) Land-cover change



b) Land-cover structure

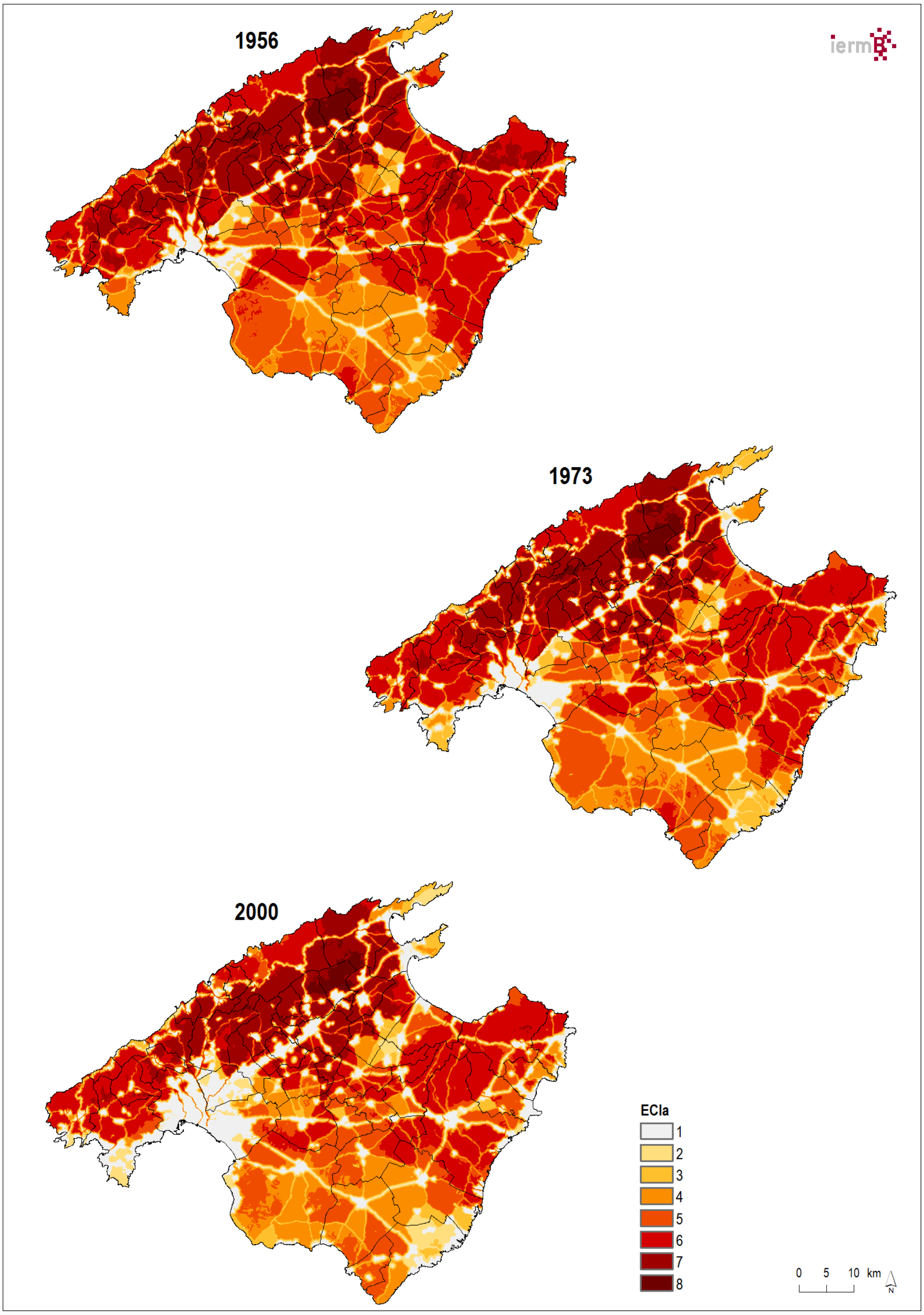


c) Land-cover functionality



Source: our own.

Figure 5 Ecological Connectivity Index (*ECI*) at regional scale (SF-1) in 1956, 1973, 2000.



Source: our own.

Transitions seen at landscape scale (SF-2)

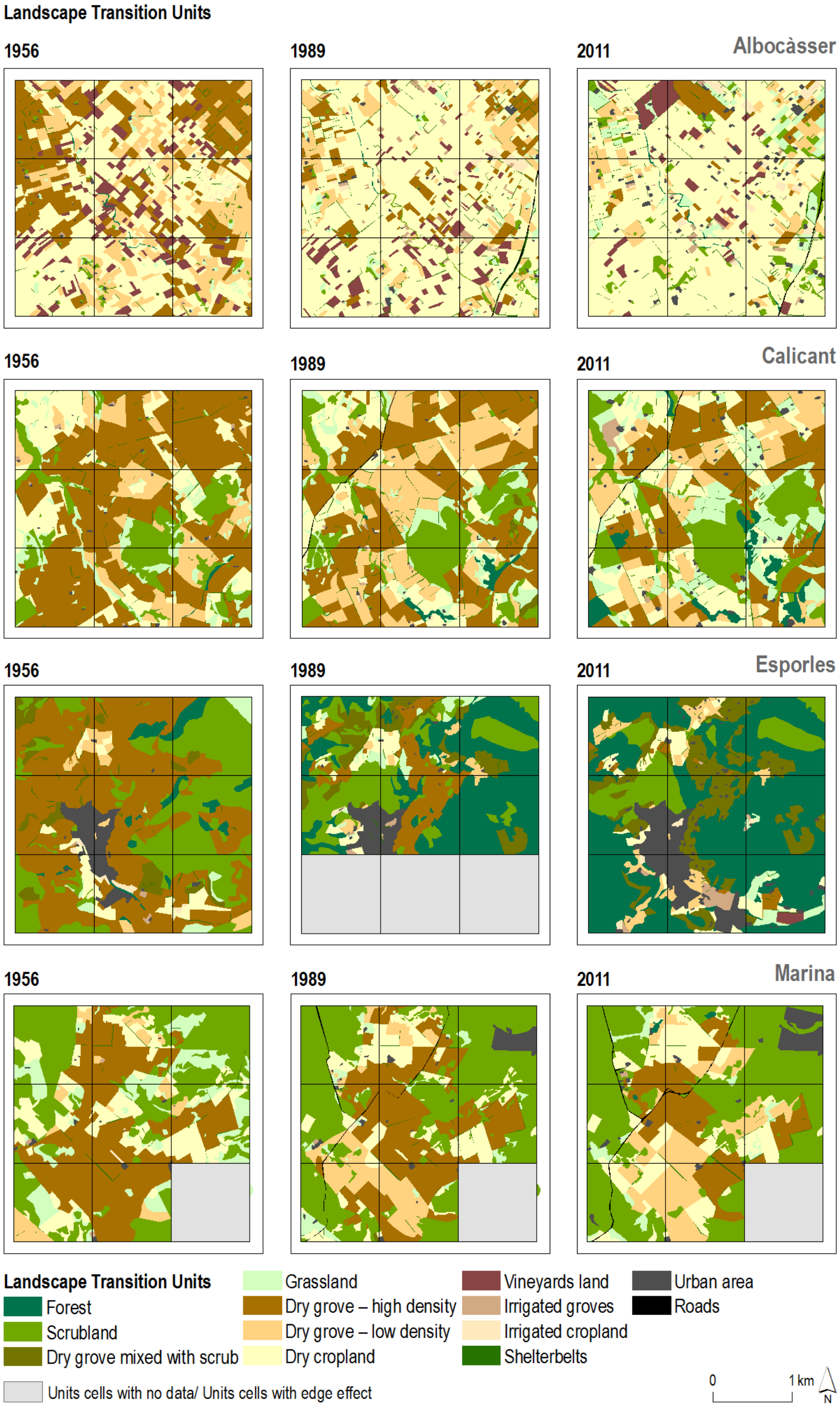
The aerial photointerpretation highlights three main landscape changes from 1956 to 1989 and 2011 in the eight scenes (Fig. 6): abandonment of rain-fed arboriculture (almond groves change to cereals; olive groves change to woodland); spontaneous reforestation following the abandonment of forestry uses (charcoal making, wood pasture, etc.); and urban sprawl (mainly tourism in coastal areas and new inland urban developments in former farm dwellings). The traditional integrated polycultures tended to be replaced by disjoint patch units of grassland, woodland, cropland and urban covers, that in most cases have led to a higher number of possible land-uses in a cell—e.g. in the '*Sant Joan*' scene. In others, the predominant trend has been towards more uniform land-covers—as the loss of land-use diversity driven by tourist urbanization in the '*Marina'* scene. In all cases this polarization has tended to the vanishing of the former landscape mosaics.

These contrasting trends of land-use intensification and abandonment have taken place along different scales and periods, as landscape metrics help to reveal (Fig. 7). Less than a quarter of the sample cells have experienced low degrees of land-cover change along the period 1956-2011. Yet during the first phase from 1956 to 1989, there were more land-use changes mainly driven by the green revolution in farm management and mass tourism in the coast. After 1989, the main drivers were rural abandonment ensuing Spanish entry to the EU (1986) and a new inward-oriented urban sprawl. These differences are shown in the rising values of land pressure (*LP*) and human-altered landscapes (*LA*) during the first phase, and the polarization trend towards either low and high levels of pressure (*LP*) or naturalness (*LN*) together with increasingly homogenised levels of human-altered landscapes (*LA*) in the second phase.

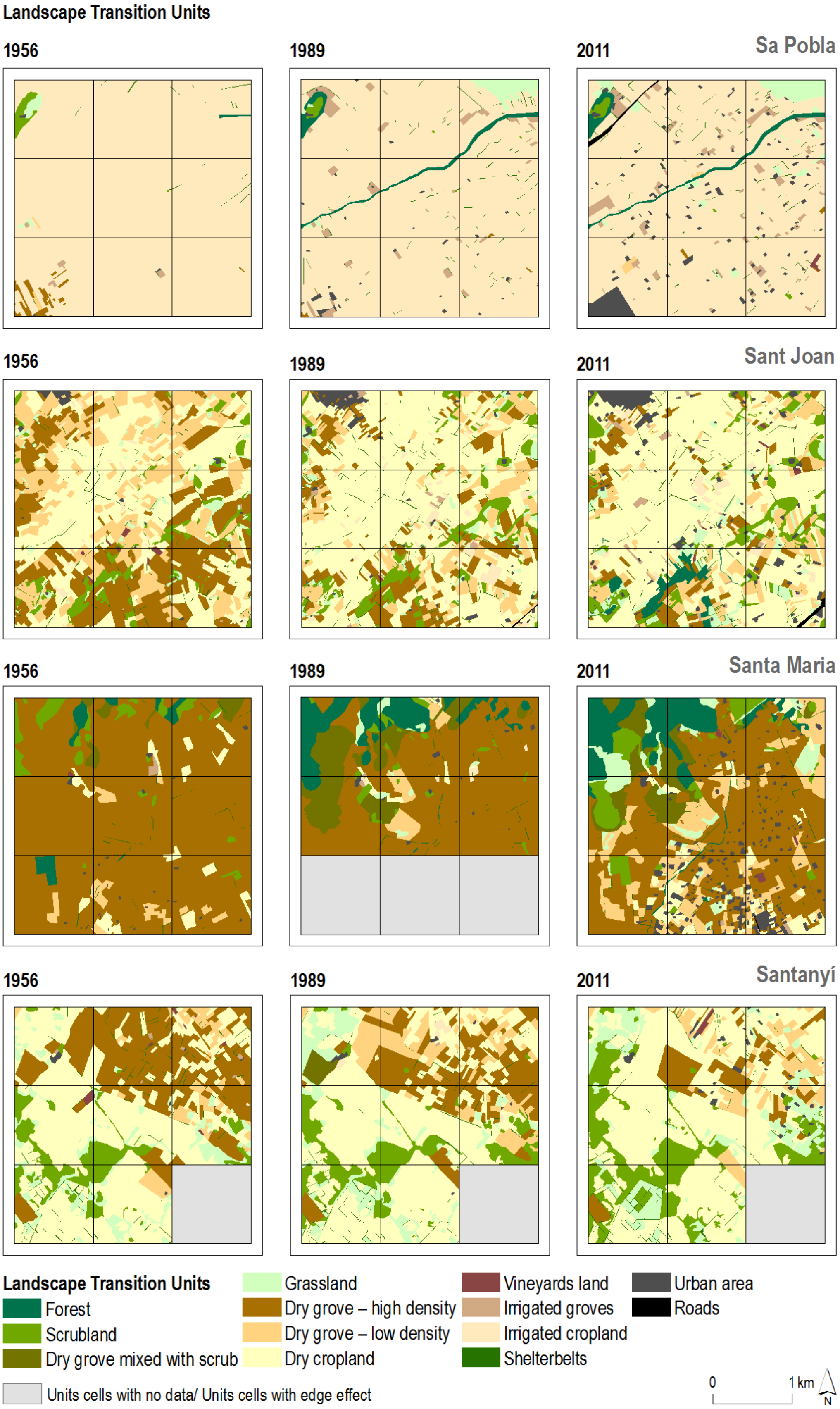
In '*Santanyí*' and '*Marina*' the loss of cultivated groves at the expense of urban developments was lower, and former rangelands were substituted by scrubland (in the southwest angle of '*Santanyí*' an unchanged area appears which corresponds to a single big estate). In '*Esporles*', in the *Tramuntana* mountains, the land-cover changed from olive groves to pine forest. In '*Santa Maria*', in the *Raiguer*, dry groves predominated and are still found despite the proliferation of isolated houses and reforestation. Due to the lack of replacement of dead almond, carob and fig trees, arboriculture has been lost in '*Calicant*', although an interesting landscape mosaic remains there except in the reforested hills. The plain areas of '*Albocàsser*' and '*Sant Joan*' have evolved from a polyculture of dry groves combined with rain-fed crops to a cereal monoculture devoid of tree cover, while some abandoned cropland and grazing areas have been conquered by woods. In '*Sa Pobla*' irrigated land remained unchanged except by the growing number of dwellings and small wetlands. The maintenance of shelterbelts is also noticeable (Fig. 6).

Figure 6 Transitions at landscape scale (SF-2; 1:5,000) from 1956 to 1989 and 2011.

a) *Albocàsser*, *Calicant*, *Esporles* and *Marina* landscape scenes



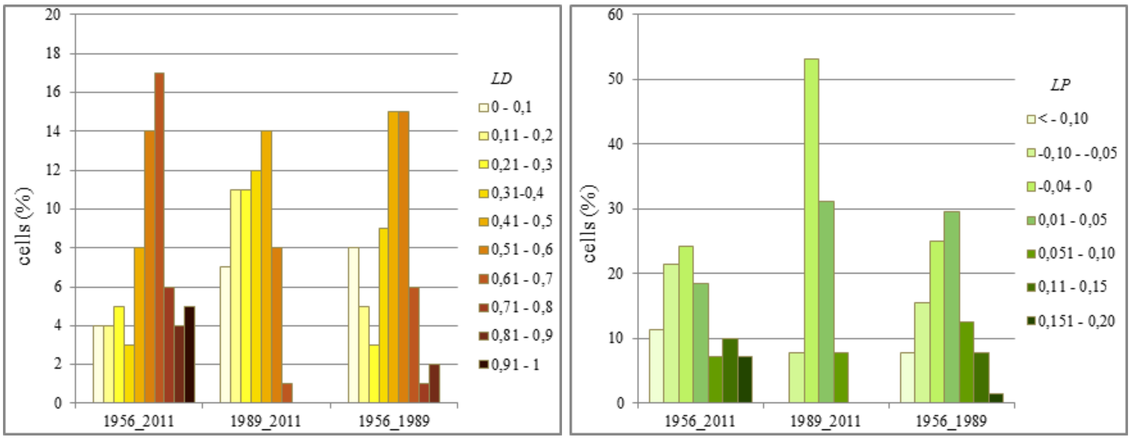
b) *Sa Pobla*, *Sant Joan*, *Santa Maria* and *Santanyí* landscape scenes



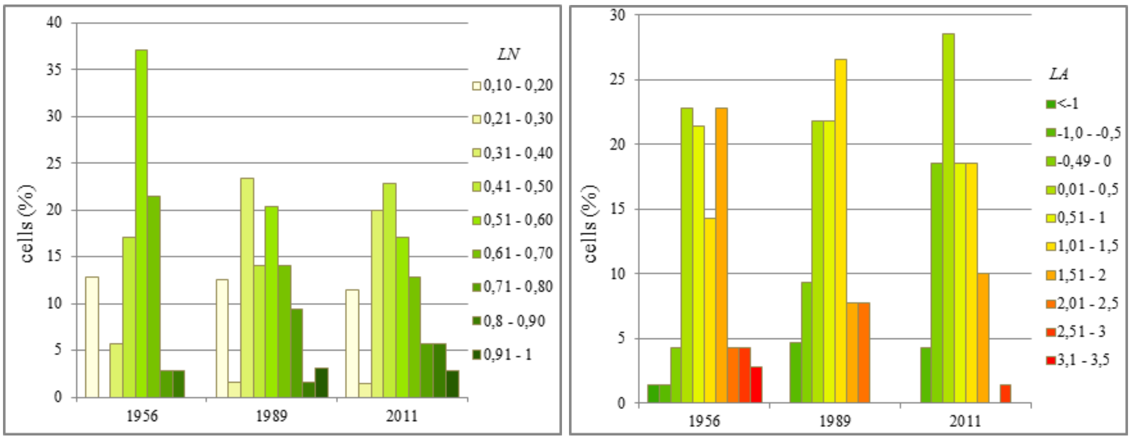
Source: our own.

Figure 7 Landscape Dynamics (*LD*), Landscape Pressure (*LP*), Landscape Naturalness (*LN*) and Landscape Anthropogeneity (*LA*) assessed at landscape scale (SF-2) from 1956, to 1989 and 2011.

a) Landscape transitions*



b) Landscape naturalness



Source: our own.

Table 5 Long-term Cultural Landscapes Analysis. Metrics of Parcel's Distribution in *Albocàsser*, *Calicant* and *Marina* scenes of the Manacor municipality: Parcel Typology (*PT;* average and maximum size, in m^2) and Parcel Ownership (*PO*; in number of parcels).

Scene	Land-use		c. 1850			1956			2012		
			PT	PT_{max}	*PO*	*PT*	PT_{max}	*PO*	*PT*	PT_{max}	*PO*
Albocàsser	LU_1	Rain-fed arable land	37.505,87	975.160,42	181	7.288,11	97.162,81	474	6.594,70	93.973,93	963
	LU_2	Almond groves	2.686,84	3.560,84	2	9.361,18	168.081,99	94	7.526,84	49.990,35	90
	LU_3	Carob groves	0,00	0,00	0	3.712,08	7.507,45	13	8.499,38	61.958,79	41
	LU_4	Fig groves	8.229,48	109.904,66	159	8.061,13	74.839,14	308	6.617,38	34.471,69	69
	LU_5	Olives groves	0,00	0,00	0	27.040,11	114.200,91	5	4.181,27	10.813,68	9
	LU_6	Almond with carob trees	0,00	0,00	0	9.652,88	11.659,57	3	13.419,10	33.522,45	7
	LU_7	Carob with fig trees	0,00	0,00	0	12.413,07	83.370,47	91	0,00	0,00	0
	LU_8	Almond with fig trees	17.804,42	78.189,31	12	36.447,00	71.497,23	9	11.650,82	23.684,19	10
	LU_9	Almond, carob and fig trees	172.763,16	1.038.785,81	8	7.739,44	56.138,51	128	7.208,55	7.208,55	1
	LU_{10}	Vineyards land	5.434,43	55.228,07	95	11.228,09	18.747,29	2	4.620,24	20.887,19	61
	LU_{11}	Irrigated groves	0,00	0,00	0	3.070,77	26.869,97	64	2.399,63	20.076,26	87
	LU_{12}	Irrigated arable land	0,00	0,00	0	24.740,40	46.324,88	2	5.175,74	51.875,36	79
	LU_{13}	Forest	19.489,23	40.896,64	11	18.714,92	48.740,29	6	6.621,33	68.951,49	12
	LU_{14}	Scrubland	32.675,78	213.749,15	17	0,00	0,00	0	3.022,67	35.111,77	101
	LU_{15}	Meadow and pasture	0,00	0,00	0	0,00	0,00	0	4.385,87	8.966,79	24
	LU_{16}	Hydrography	13.000,88	14.667,69	2	5.687,53	9.724,54	5	3.390,17	9.724,54	11
	LU_{17}	Unproductive	6.231,41	142.036,55	28	9.833,99	268.598,34	29	640,51	20.424,75	749
	ND	No data	7.259,95	64.151,66	128	-	-	-	-	-	-
Calicant	LU_1	Rain-fed arable land	157.716,76	1.443.900,49	52	46.921,51	200.473,80	17	16.267,42	326.649,61	135
	LU_2	Almond groves	13.226,23	15.018,08	2	28.349,30	127.720,11	26	31.582,55	224.430,75	137
	LU_3	Carob groves	0,00	0,00	0	25.328,73	25.328,73	1	22.774,22	107.892,70	17
	LU_4	Fig groves	12.402,68	28.137,29	8	21.331,18	39.051,01	4	20.235,58	64.775,99	24
	LU_5	Olives groves	0,00	0,00	0	0,00	0,00	0	0,00	0,00	0
	LU_6	Almond with carob trees	0,00	0,00	0	67.219,82	220.571,07	10	37.029,57	117.847,53	13
	LU_7	Carob with fig trees	0,00	0,00	0	38.166,77	50.754,29	3	22.726,44	93.078,92	13
	LU_8	Almond with fig trees	75.097,90	340.158,45	5	67.295,38	402.536,77	61	0,00	0,00	0
	LU_9	Almond, carob and fig trees	9.744,89	19.521,85	5	117.598,69	672.867,31	32	0,00	0,00	0
	LU_{10}	Vineyards land	0,00	0,00	0	0,00	0,00	0	817,48	1.121,28	2
	LU_{11}	Irrigated groves	0,00	0,00	0	0,00	0,00	0	1.043,65	2.171,55	4
	LU_{12}	Irrigated arable land	0,00	0,00	0	0,00	0,00	0	19.171,27	25.332,79	2
	LU_{13}	Forest	0,00	0,00	0	0,00	0,00	0	6.485,23	13.844,48	3
	LU_{14}	Scrubland	217.359,37	2.087.345,61	36	98.557,74	751.453,04	21	22.560,67	494.944,05	132
	LU_{15}	Meadow and pasture	0,00	0,00	0	0,00	0,00	0	9.132,42	25.201,22	6
	LU_{16}	Hydrography	16.329,60	28.401,43	3	12.229,66	16.680,95	4	6.974,40	15.614,09	7
	LU_{17}	Unproductive	4.917,83	18.424,61	15	5.412,08	117.639,93	24	981,25	17.699,68	225
	ND	No data	-	-	-	-	-	-	-	-	-
Marina	LU_1	Rain-fed arable land	1.522.358,89	2.992.231,31	5	42.011,44	108.184,00	6	24.398,92	285.480,81	62
	LU_2	Almond groves	0,00	0,00	0	122.139,54	414.107,47	8	59.052,65	177.507,46	38
	LU_3	Carob groves	0,00	0,00	0	8.268,15	16.001,39	3	10.862,61	37.188,01	12
	LU_4	Fig groves	3.512,48	3.512,48	1	26.631,59	41.442,41	6	20.356,78	69.046,19	8
	LU_5	Olives groves	0,00	0,00	0	0,00	0,00	0	0,00	0,00	0
	LU_6	Almond with carob trees	0,00	0,00	0	8.766,49	12.822,50	3	55.314,06	73.746,41	2
	LU_7	Carob with fig trees	10.436,24	16.529,72	2	31.559,93	31.559,93	1	44.620,28	44.620,28	1
	LU_8	Almond with fig trees	11.798,01	16.680,34	4	68.681,55	229.328,39	11	9.721,01	14.206,42	2
	LU_9	Almond, carob and fig trees	0,00	0,00	0	66.320,42	280.886,03	33	0,00	0,00	0
	LU_{10}	Vineyards land	0,00	0,00	0	0,00	0,00	0	0,00	0,00	0
	LU_{11}	Irrigated groves	0,00	0,00	0	0,00	0,00	0	1.949,60	3.851,72	10
	LU_{12}	Irrigated arable land	0,00	0,00	0	0,00	0,00	0	8.637,14	12.700,66	2
	LU_{13}	Forest	0,00	0,00	0	0,00	0,00	0	70.308,90	812.845,28	24
	LU_{14}	Scrubland	574.498,44	2.959.575,03	22	180.658,01	2.548.448,79	59	54.768,37	764.616,64	111
	LU_{15}	Meadow and pasture	0,00	0,00	0	0,00	0,00	0	0,00	0,00	0
	LU_{16}	Hydrography	0,00	0,00	0	0,00	0,00	0	19.632,32	23.983,06	2
	LU_{17}	Unproductive	9.567,51	33.287,31	13	5.979,11	159.289,45	30	9.107,98	867.187,07	127
	ND	No data	-	-	-	-	-	-	-	-	-

Source: our own.

Land-use patterns at local scale (SF3)

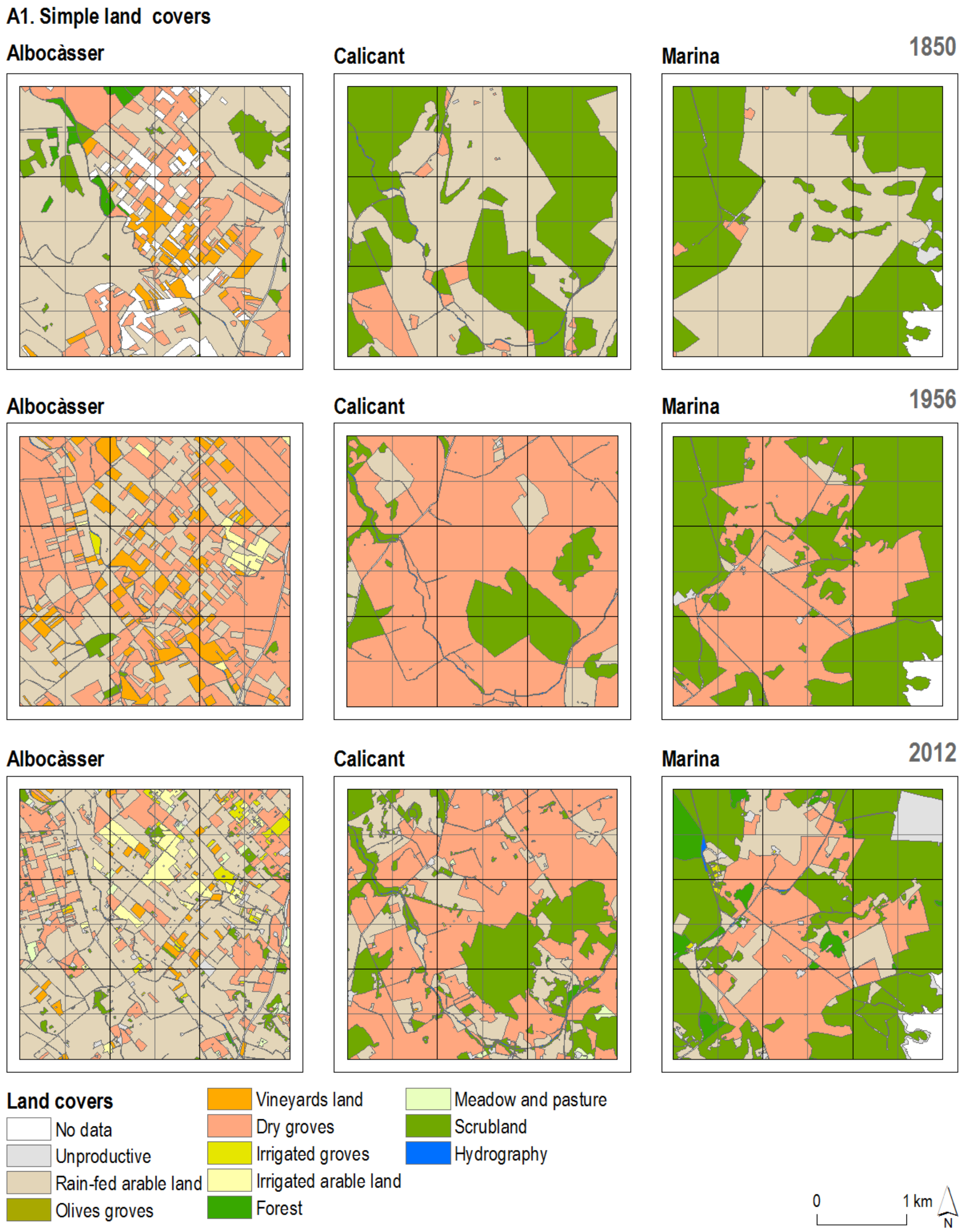
The closest approach allows us to capture finer relationships between land-use changes, ownership regimes and socioeconomic drivers of landscape change. We can observe in the three local scenes of Manacor municipality the expansion of dry polycultural groves from c.1850 to 1956, at the expense of rain-fed arable land, woodland and scrubs (Fig. 8 and Table 5). This happened as a result of the financial and political crisis of the old large estates (the so-called *possessions*) during the second half of the nineteenth century and the first two decades of the twentieth, which opened up a process of land parcelling allotted to small peasants offering them an option to make a living with a labour-intensive farming (Suau 1991; Manera 2001). The allotment process is more clearly shown in *Albocàsser* than in the mountainous area of *Calicant*, and even more than in *Marina* due to poor soils and aridity (Table 5), but everywhere crop diversity increased with the extent of landownership (Table 6). Not only leguminous carobs, but also almond and fig trees were grown in association with cereals and legumes, and even caper plants were grown in summer at the foot of the trees in the whole island (Bisson 1977). These multi-cropping groves of almonds and carobs grew from 6,048 and 7,789 ha in 1860 to 47,560 and 21,875 ha in 1930 respectively (Urech and Cifre 1869; Cela-Conde 1979).

Table 6 Long-term Cultural Landscapes Analysis. Relative areas of land-uses according to property size in the Manacor municipality scenes (c. 1850, 1956, 2012).

Year	Land-use		Property size (%)			
			<0,1ha	0,1-0,5ha	0,5-1ha	>1ha
c. 1850	LU_{1}	Rain-fed arable land	11,3	26,5	25,9	37,5
	LU_{2}	Almond groves	0,0	0,6	0,0	0,7
	LU_{3}	Carob groves	0,0	0,0	0,0	0,0
	LU_{4}	Fig groves	7,5	23,7	29,6	14,3
	LU_{5}	Olives groves	0,0	0,0	0,0	0,0
	LU_{6}	Almond with carob trees	0,0	0,0	0,0	0,0
	LU_{7}	Carob with fig trees	0,0	0,3	0,0	0,4
	LU_{8}	Almond with fig trees	0,0	0,6	4,3	4,3
	LU_{9}	Almond, carob and fig trees	0,0	0,9	0,6	3,2
	LU_{10}	Vineyards land	11,3	18,7	13,0	2,9
	LU_{11}	Irrigated groves	0,0	0,0	0,0	0,0
	LU_{12}	Irrigated arable land	0,0	0,0	0,0	0,0
	LU_{13}	Forest	0,0	0,9	0,6	2,5
	LU_{14}	Scrubland	1,9	3,7	3,1	20,4
	LU_{15}	Meadow and pasture	0,0	0,0	0,0	0,0
	LU_{16}	Hydrography	0,0	0,3	0,0	1,4
	LU_{17}	Unproductive	60,4	2,8	3,1	3,6
1956	LU_{1}	Rain-fed arable land	8,0	40,9	35,5	18,2
	LU_{2}	Almond groves	1,1	6,1	10,9	9,9
	LU_{3}	Carob groves	1,1	1,5	1,1	0,4
	LU_{4}	Fig groves	0,0	25,7	20,4	15,0
	LU_{5}	Olives groves	0,0	0,0	0,0	0,0
	LU_{6}	Almond with carob trees	0,0	0,4	0,6	2,7
	LU_{7}	Carob with fig trees	0,0	0,0	0,3	1,3
	LU_{8}	Almond with fig trees	0,0	4,3	12,8	18,6
	LU_{9}	Almond, carob and fig trees	0,0	1,0	0,6	13,7
	LU_{10}	Vineyards land	0,0	8,6	12,8	5,1
	LU_{11}	Irrigated groves	0,0	0,1	0,0	0,2
	LU_{12}	Irrigated arable land	10,2	7,1	1,1	0,6
	LU_{13}	Forest	0,0	0,1	0,0	0,2
	LU_{14}	Scrubland	1,1	2,2	2,8	12,7
	LU_{15}	Meadow and pasture	0,0	0,0	0,0	0,0
	LU_{16}	Hydrography	0,0	0,6	0,6	0,6
	LU_{17}	Unproductive	78,4	1,3	0,6	0,6
2012	LU_{1}	Rain-fed arable land	5,1	51,3	52,2	36,6
	LU_{2}	Almond groves	0,7	5,0	11,1	22,5
	LU_{3}	Carob groves	0,4	2,5	2,8	3,5
	LU_{4}	Fig groves	0,4	3,5	5,1	4,8
	LU_{5}	Olives groves	0,0	0,6	0,2	0,2
	LU_{6}	Almond with carob trees	0,0	0,3	0,8	2,4
	LU_{7}	Carob with fig trees	0,1	0,1	0,6	1,4
	LU_{8}	Almond with fig trees	0,0	0,2	0,8	1,0
	LU_{9}	Almond, carob and fig trees	0,0	0,0	0,2	0,0
	LU_{10}	Vineyards land	0,5	3,4	2,4	1,0
	LU_{11}	Irrigated groves	2,9	5,1	1,4	0,3
	LU_{12}	Irrigated arable land	1,1	3,4	4,5	1,3
	LU_{13}	Forest	0,6	0,9	1,2	2,4
	LU_{14}	Scrubland	6,0	9,8	8,3	18,9
	LU_{15}	Meadow and pasture	0,3	1,3	2,0	0,3
	LU_{16}	Hydrography	0,2	0,8	0,8	0,6
	LU_{17}	Unproductive	81,8	12,0	5,7	3,0

Source: our own.

Figure 8 Land-use changes at local scale (SF-3; 1:500) in *Albocàsser*, *Calicant* and *Marina* scenes of the Manacor municipality in c. 1850, 1956 and 2012.



Source: our own.

Thanks to smallholders' work and inventiveness, that took advantage of the growing international demand for almonds, capers, potatoes, dried fruits (figs, apricots) and vegetables (Manera 2001), there was a shift towards complex agro-forest mosaics of higher diversity—as shown in the landscape metrics of these three scenes (Fig. 10). Values of land-use richness (*LUR*), edge density (*ED*) and polygon density (*PD*) increased while large patch index (*LPI*) decreased from c.1850 to 1956, reflecting the greater land-cover diversity and ecotones of those multi-cropping mosaics interwoven with woods and pastures. Conversely, from 1956 to 2012 these scenes confirm the trend towards the disappearance of polycultural landscapes (Fig. 10 and Table 5) already observed at larger scales.

This local scale also reveals that up to the present the withdrawal of farmer's labour and knowledge has been only partial in Mallorca. The average or high values of land-use richness (*LUR*), land-cover diversity and ecotones (*ED*, *PD*) attained in 1956 are still found at present. This feature highlights the need to delve deeper into the socioeconomic drivers and ruling agencies behind this socio-ecological transition—a task which requires another forthcoming article whose main interpretive lines are outlined in the following subsection 3.5.

3.2. Human disturbance and landscape complexity in cultural landscapes

To conclude our intermediate disturbance analysis, we studied the statistical relationships between *HANPP* and all the landscape ecology metrics used as proxy for biodiversity, in the set of cells of our experimental design at regional scale. The high correlations (Table 7 a) among land-cover metrics (*H'*, *MESH*, *ECI*, *LMI* and *LCR*) aim us to carry out a Principal Component Analysis (PCA). Hence, we performed a PCA of the variables involved (Table 7 b) that shows that the major contributors for the first component (C1) are *H'* and *MESH*; and for the second component (C2) are *ECI* and *LMI*. *LCR* goes alone in all dimensions. These results have led us to consider a PCA taking only two variables, *H'* and *ECI*, so that the two first dimensions are represented—which include patterns as landscape heterogeneity, and processes by means of ecological connectivity. Once we have reduced the dimensions of the land-cover metrics, we obtain a component resulting of the linear combination of *H'* and *ECI* (component coefficient = 0.707; explained variance = 65%). We call this new *H'—ECI* component 'Landscape Metrics' (*L*).

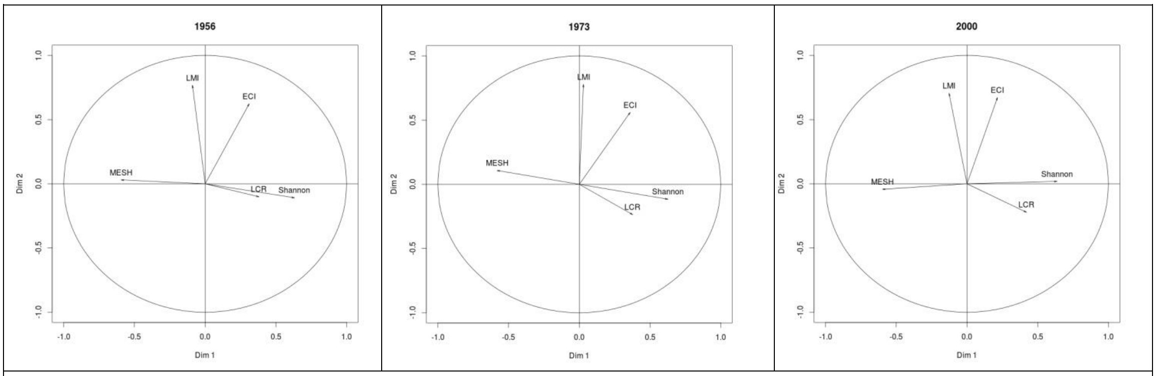
Table 7 Relationships among land-cover metrics using Principal Component Analysis (PCA) at regional scale (SF-1): Land Cover Richness (*LCR*), Shannon-Wiener Index (*H'*), Effective Mesh Size (*MESH*), Ecological Connectivity Index (*ECI*) and Landscape Metric Index (*LMI*).

a) Correlation Analysis between variables

	H'			*MESH*			*ECI*			*LMI*			*LCR*		
	1956	1973	2000	1956	1973	2000	1956	1973	2000	1956	1973	2000	1956	1973	2000
H'	1	1	1	-0.888	-0.921	-0.92	0.273	0.355	0.267	-0.164	-0.033	-0.132	0.472	0.479	0.524
MESH	-0.888	-0.921	-0.92	1	1	1	-0.33	-0.312	-0.26	0.124	0.058	0.146	-0.226	-0.251	-0.316
ECI	0.273	0.355	0.267	-0.33	-0.312	-0.26	1	1	1	0.302	0.392	0.43	0.149	0.182	0.036
LMI	-0.164	-0.033	-0.132	0.124	0.058	0.146	0.302	0.392	0.43	1	1	1	-0.083	-0.126	-0.213
LCR	0.472	0.479	0.524	-0.226	-0.251	-0.316	0.149	0.182	0.036	-0.083	-0.126	-0.213	1	1	1

Note: Correlations are shown considering each time period and all data together.

b) Principal Component Analysis



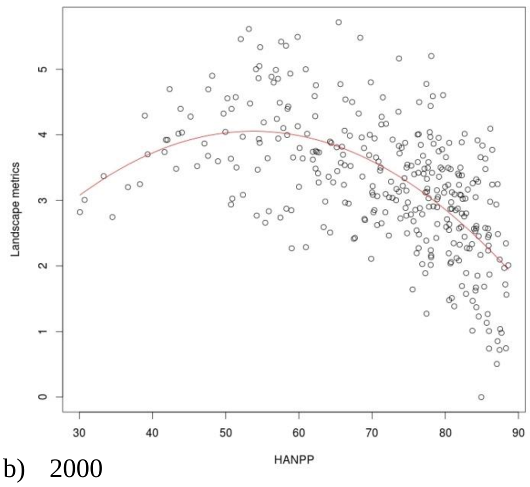
1956	C1	C2	C3	1973	C1	C2	C3	2000	C1	C2	C3
Variance (%)	45.573	25.690	16.391	Variance (%)	46.908	26.667	15.817	Variance (%)	46.522	29.271	14.041
H'	0.631	-0.108	-0.102	Shannon	0.626	-0.116	-0.179	Shannon	0.636	-0.053	-0.145
MESH	-0.594	0.031	0.411	MESH	-0.58	0.108	0.464	MESH	-0.596	0.016	0.428
ECI	0.311	0.623	-0.061	ECI	0.359	0.56	0.166	ECI	0.287	0.633	0.074
LMI	-0.09	0.767	0.137	LMI	0.029	0.778	0.048	LMI	-0.072	0.741	0.173
LCR	0.38	-0.101	0.894	LCR	0.377	-0.236	0.85	LCR	0.389	-0.218	0.872

Source: our own.

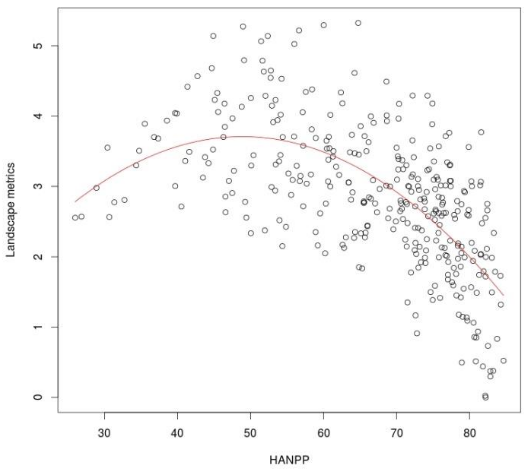
Fig. 9 shows the results of a quadratic regression analysis, where *HANPP* is the independent variable that influences *L* as a proxy of landscape's ecological patterns and processes (Table 7). In all time periods we obtain a second degree polynomial regression linking the two sets of data (socio-metabolic disturbance and landscape ecological functioning), that confirms our intermediate disturbance-complexity hypothesis (IDC) by showing a hump-shaped relationship where the highest level of landscape complexity (heterogeneity-connectivity as biodiversity proxy) is attained when *HANPP* peaks at 50-60%. The time factor should not affect the relationship between variables, given that the *IDC* hypothesis represented in the non-lineal regression does not depend on time. By changing the perspective from regional to local scale, the results found in the three Manacor scenes (Fig. 10 and Fig. 11) confirm that the historical trend that attained the highest land-cover diversity (*H'*) in 1956 was also linked to shifts in *HANPP* values. Yet the relationship seems to be more differentiated locally, which calls for a further geo-historical study of this complex interplay between biological and cultural factors.

Figure 9 Relationship between Landscape Metrics (*L*) and Human Appropriation of Net Primary Production (*HANPP*) at regional scale (SF-1) in 1956, 1973 and 2000.

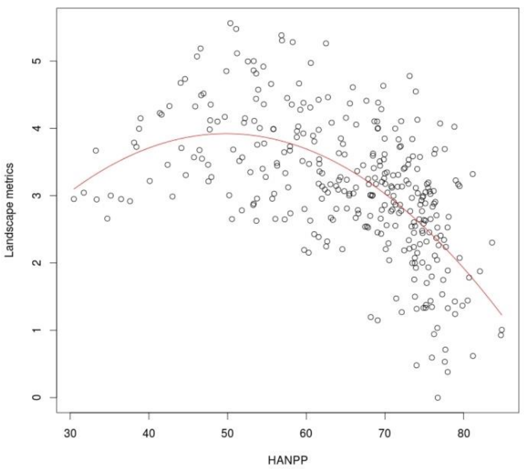
a) 1956



b) 1973



b) 2000

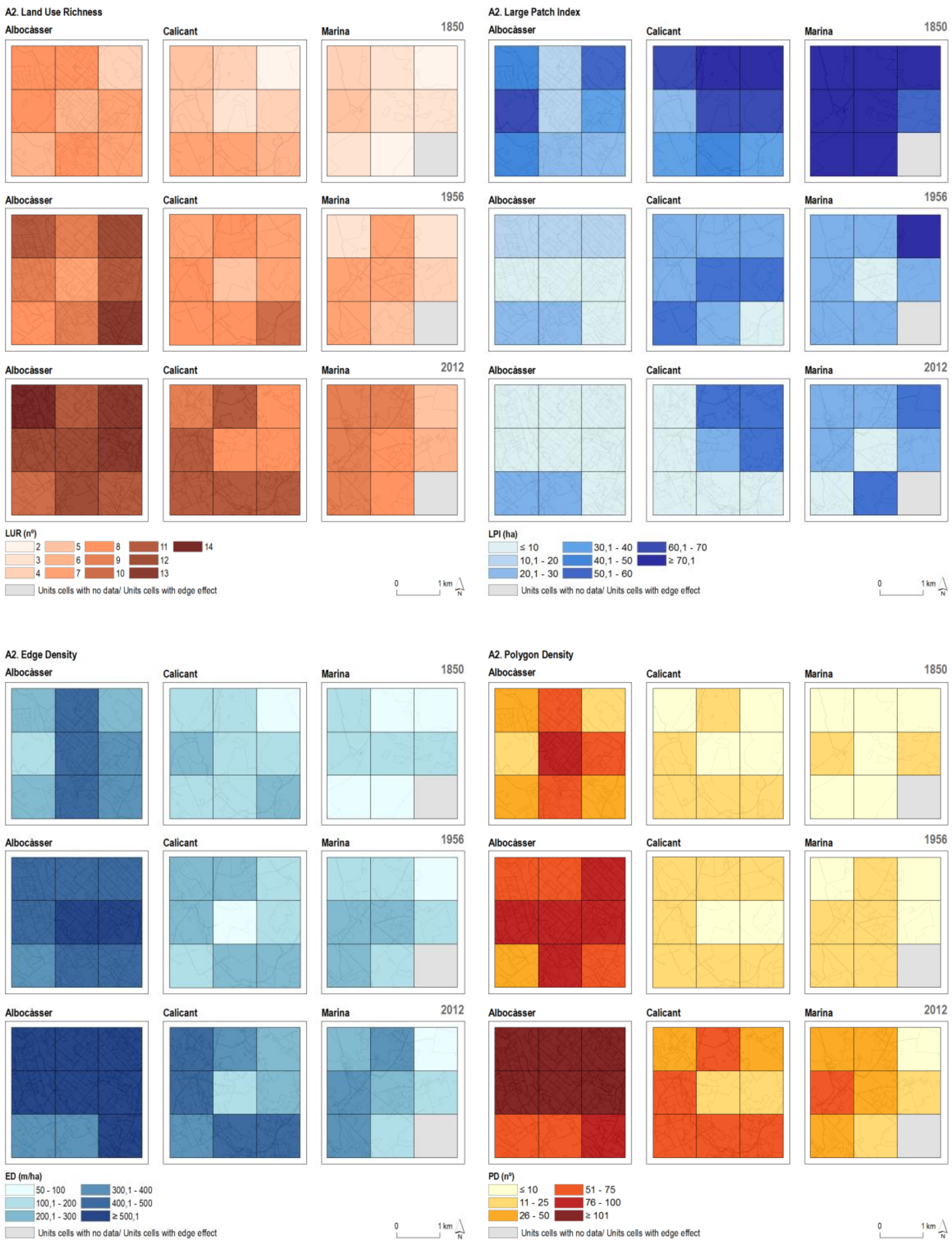


Coefficients	1956		1973		2000	
(Intercept)	-0.961		-0.502		-1.527	
HANPP	0.186	***	0.172	***	0.218	***
HANPP²	-1.73e-3	***	-1.76e-3	***	-0.002	***
Multiple R^2	0.383		0.424		0.379	

Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Source: our own. Note: Results of the quadratic regression analysis, where *HANPP* is the independent variable that influences *L* as proxy of ecological patterns and processes (Table 5).

Figure 10 Landscape metrics applied at local scale (SF-3) in *Albocàsser*, *Calicant* and *Marina* scenes of the Manacor municipality in c. 1850, 1956 and 2012: Land Use Richness (*LUR*), Largest Patch Index (*LPI*), Edge Density (*ED*) and Polygon Density (*PD*).



Source: our own.

Figure 11 Shannon-Wiener Index of land-cover diversity (*H'*) and Human Appropriation of Net Primary Production (*HANPP*) applied at local scale (SF-3) in *Albocàsser*, *Calicant* and *Marina* scenes of the Manacor municipality in c. 1850, 1956 and 2012.

Source: our own.

3.3. Driving forces and ruling agencies of socio-ecological change

From Middle Ages onwards (Jover and Soto 2002; Soto 2015) the agrarian change in the island was driven by the conflicting relationship between large estates (*possessions*) that hoarded most of the land, and peasant smallholders of tiny plots confined in the outskirts of the inner villages—who, in turn, supplied the wage labour hired to farm big estates. While the landowners practised extensive land usages and an export-oriented farm management (with olive oil trade as the main commercial driver), small peasants' farming was highly intensive, diversified, and household or locally oriented (Bisson 1997, Manera 2001). In order to prevent a rise of agricultural wages as a result of a reduction of farmhands' supply, big landowners tried to restrain the advance of those peasant land belts of intensive poly-culture, until they went bankrupt in the nineteenth century (Jover and Manera 2009). The parcelling of many large estates from the 1860s to the 1920s entailed a significant change in the cultural landscapes kept by this dual agrarian class structure (Cela-Conde 1979; Rosselló-Verger 1982). Thus, and foremost, the wonderful 'traditional' landscapes which attracted elite visitors to Mallorca, from George Sand and Frederic Chopin (1838-39) to the Archduke Ludwig Salvator von Habsurbg-Lorena (1847-1915) who wrote a famous nine-volume treatise on the Balearic Islands, were to a large extent a relatively recent creation of small peasants who made advances in the age-old fight to have access to the land.

Tourism development of Mallorca from the elites of the Belle Époque up to the mass invasion of sun-and-sea holidaymakers has cast a Midas curse. Urban sprawl extended from coastal hotels to inland houses built in former rural dwellings, together with the highways linking them, which jointly entailed a growing environmental impact that tended to destroy the same landscape beauty that led Mallorca to become a tourist destination known worldwide (Pons et al. 2014). Developed land multiplied by 3.8 from 1956 to 2000, and doubled after 1973, as seen in Fig. 3 and Table 4 (Murray 2012). Yet the impact of tourism on the island's agriculture has been twofold. On the one hand it has entailed a strong socioeconomic marginalisation of farming, leading to rural abandonment—with the usual ecological impacts such as wildfires (Gil-Sánchez et al. 2002) and disruption of complex dry stone hydraulic systems (Estrany et al. 2010). On the other hand, this effect started so early that, after the halt of Franco's autarky (Naredo 2004), the intensification of farm and livestock management following the green revolution lines was tempered to some extent—with the usual outcomes of monocultures, soil degradation and water pollution (Roca 1992). Our SF-2 assessment shows that industrialization of agriculture left a clear imprint in the evolution of cultural landscapes mainly during the 1956 to 1989 period. But it was comparatively soft in regard to what happened in other parts of the Mediterranean basin, such as the province of Barcelona in Catalonia (Marull et al. 2010).

Three factors may explain the relatively high resilience (Marull et al. 2015b) of the cultural landscapes that peasants created in Mallorca before traditional organic farming ended. First, the commitment of local population that kept buying foodstuffs grown on the island (many years before the zero-km and slow food movements began) helped to maintain a precarious part-time agriculture that sought a compromise between traditional-organic and industrial farm managements. Second, following the Spanish EU membership in 1986 the main socioeconomic driver was rural abandonment that pushed towards relying on the increasing amount of imported food (Murray 2012). Small farms have been maintained mostly thanks to the hard work of non-professional peasants who have remained attached to the land for cultural and emotional reasons. The ageing of this group is one of the most important threats for bio-cultural preservation currently (Binimelis and Ordines 2008). In spite of this, the esteem of the local population for their food, tastes and landscapes was reinforced from then on by the growing environmental movement (Rayó 2004) led by the Grup d'Ornitologia Balear (GOB). Together with the EU environmental directives, this social pressure became a third factor that helped to preserve some natural sites and restrain urban sprawl to some extent—despite the ambiguous and shifting policies adopted by the autonomous and Spanish governments (Rullan 2010).

Not only the agricultural landscape and traditional peasant knowledge are currently threatened by low incomes and lack of farmers' replacement, but also the rich diversity of local species varieties as well (Socies 2013). The entire bio-cultural heritage of the Mallorca Island is at stake. Last but not least, a local turning towards organic farming is on the way. Its promoters are younger and with a higher education than old peasants, and the shift towards high-quality foodstuffs can help to increase farming incomes—provided that consumers are willing to pay for them, and public policies are reoriented to foster local organic food instead of promoting tourism and urban developments at the expense of farming as it currently does. Despite the lack of political support, organic food is growing thanks to the efforts of small peasants and social movements. If there is a sustainable future for a cultural landscape able to hold a high biodiversity in Mallorca, this clearly belongs to the role of organic farming as heir of the rich bio-cultural heritage of this beautiful Mediterranean island (Alcover et al. 2003).

4. Conclusion

An intermediate-disturbance conceptual approach has been applied to the land-use changes of cultural landscapes underwent in the island of Mallorca from c.1850 to the present. It accounts for the joint multi-scalar behaviour of human appropriation of photosynthetic capacity (*HANPP*) and landscape heterogeneity. We obtained a second-degree polynomial regression linking *HANPP* with landscape ecological functioning, jointly assessed by Shannon Index (*H'*) of land-cover patterns and ecological connectivity (*ECI*) of landscape processes, which confirms our intermediate disturbance-complexity hypothesis. As far as we know, few authors have studied the relationship between these variables, or other similar ones (Wbrka et al. 2004: Haberl et al. 2005; Vackar et al. 2012).

The results found show the usefulness of transferring the concept of intermediate disturbance to agro-ecological landscapes (Gliessman 1990; González de Molina and Toledo 2014), and suggest that rural development and land-use planning policies should consider the territory as a whole instead of applying a string of ad hoc decisions on minor parts of cultural landscapes as usual (Rullan, 2010; Agnoletti 2014). The historical landscape analysis performed and the driving forces described show that traditional farming played a crucial role in shaping and maintaining a complex set of land-use mosaics. Our results suggest that a great deal of the biodiversity currently existing in Mallorca may actually be associated to the remaining agricultural and forest mosaics still worked by the local peasantry. We deem that the keeping of this bio-cultural heritage may underlie the hump-shaped relationship we have found between *HANPP* and landscape ecological functionality jointly assessed with land-cover diversity and ecological connectivity -a result that fits with the intermediate disturbance hypothesis. Protecting natural spaces but at the same time allowing their isolation by the spread of anthropogenic barriers that decrease ecological connectivity will eventually lead to a biodiversity loss in the whole land matrix (Pino and Marull 2012). Conversely, the conservation of heterogeneous and well-connected landscapes with a positive interplay between intermediate level of farming disturbances and land-use complexity would preserve a wildlife-friendly agro-ecological matrix that is likely to hold a great biodiversity—perhaps with the exception of rare specialist species that require some specific habitats and other conservation policies (Loreau et al. 2010; Tscharnkte et al. 2012).

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