Linking landscape futures with biodiversity conservation strategies in northwest Iberia –
a simulation study combining surrogates with a spatio-temporal modelling approach

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Abstract

The most prominent factors inducing landscape change in the rural regions of south west Europe are depopulation and the associated socio-ecological modifications. The aim of this work was to assess the future implications of these processes on land use/land cover and biodiversity in northwest Iberia. To achieve our goal, we developed a virtual spatially explicit dynamic model to simulate regional socio-ecological dynamics. For the period between 1960 and 2040, we tested four different environmental scenarios ranging from small decreases in conifer forest and a stabilization of agricultural areas and shrublands to more radical shifts, substantial decreases in agricultural areas and massive expansion of eucalyptus stands. The model considers also fire, whose role increased significantly in the scenarios of expanded forests. Bird assemblages, which we used as a surrogate for biodiversity, showed complex patterns although with overall decreases in richness and abundance. Species with narrow niche and from Mediterranean open habitats were particularly sensitive to the ongoing changes simulated. Our results suggest that landscape management actions and planning assessments designed for conserving biodiversity should focus on the maintenance of the traditional agricultural mosaic combined with a regulatory legislation limiting the expansion of fast growing tree forests (e.g. Eucalyptus stands). This strategy can contribute to maintaining a diversity of land use/land cover in a heterogeneous landscape and the prevention of the occurrence of large wildfires, fundamental for the implementation of national biodiversity strategies and action plans.

Keywords: Socio-ecology; Bird community; Farmland; Agent-based models; Biological Conservation; Forest

1. Introduction

Human-dominated landscapes reflect the long-term interactions between people and the environment (Moreira et al., 2001a), mediated by processes as land use/land cover (LULC) intensification or abandonment, which are mostly determined by socioeconomic and political issues (van Vliet et al., 2015). With the multifold and rapid ecological and social changes characterising the Anthropocene, one of the greatest challenges is to predict how environmental trends can influence local environments and trigger changes in ecosystems and landscapes. Divergent trends of biological communities, both in structure and in species composition, are considered good surrogates to detect and quantify those changes (Fernández-Llamazares et al., 2015). Ecosystems are resilient, to some degree, against most natural environmental changes and episodic disturbances, although major shifts can result in novel structures and functioning (Mori, 2011). The global environmental crisis, however, can reduce the resilience of ecological systems and thus increase the risk of regime shifts (e.g. Andersen et al., 2009; Cote and Nightingale, 2012). In cases where coupled ecological and social systems are involved, one refers to a social-ecological regime shift (Kull et al., in review). For example, agricultural intensification as well as the abandonment of agriculture may drive many rural systems into alternative states with reduced biodiversity or ecosystem services. Such shifts are difficult to reverse because of the long-time lags characterizing the dynamics of socio-ecological systems, amongst other factors (Figueiredo and Pereira, 2011). Also these shifts do not display nonlinear features and difficult-to-reverse nature of regime shifts interceded by tipping points identified in several works (e.g. Figueiredo and Pereira, 2011). In fact societal and ecological mechanisms usually interact, generating changes in biological communities,

degradation of ecosystem services and effects on human well-being in a much step by step linear trend but also imperceptible and misunderstood way (Bennet et al., 2009).

In this context, the implementation of the Convention of Biological Diversity (CBD) by National Biodiversity Strategies and Action Plans (NBSAPs) is a key endeavour (Meyerhoff et al., 2012). However, their effectiveness will demand urgently: (i) a better understanding of the interactions between socio-economic changes and biodiversity trends, both in theory and in practice (Maxim, 2012); and (ii) a fair trade-off between socio-economy needs and biodiversity conservation. Nonetheless, key decision-makers are often local people, whose land management choices directly affect biodiversity. Those choices can be associated to several types of economic activities, therefore diverse scenarios of socio-economic development should be considered for a consistent implementation of NBSAPs (Meyerhoff et al., 2012).

The North-western corner of the Iberian Peninsula represents a good example of a highly biodiverse and dynamic landscape (Moreira et al., 2001a). This region was, until the 1960s, a heterogeneous mosaic of traditional LULC: small villages, pastures and arable land in the valleys, surrounded by privately owned shrublands and forests used as a source of timber, wood, and litter. In the mountains, large areas of common land heaths were used as pasture (Appendix A) (Calvo-Iglesias et al., 2009). In such a multi-purpose land management system, arable and livestock farming, combined with forest harvesting were the main means of subsistence for rural communities in the region (Calvo-Iglesias et al., 2009). Socio-economic and political pressures, rural exodus, farmland abandonment, afforestation and emigration resulted from the 1970s onwards in severe reductions in the population and in agricultural areas (Pinto-Correia and Vos, 2004). The replacement of shrublands and native woodlands by plantations of fast growing species, for the production of wood or pulp, have significantly increased in the

last decades as a result of changing LULC policies in parallel with the decline of traditional agriculture (Appendix B) (Pinto-Correia and Vos, 2004; Lopes et al., 2013). As one of the largest industrial and economic sectors in the region, the pulp and paper industry has a great influence on the LULC and ultimately on ecosystems and biodiversity (Kardell et al., 1986; Pérez et al., 2006). Moreover, population loss was (and is) associated with a lack of investment in arable land, which was subsequently colonized by shrubs by processes of re-naturalization (Pérez et al, 2006). The current forest cover is therefore a ‘social’ phenomenon (Appendix C). As a result, the region now has the highest fire frequency in Europe, a trend that is still increasing (San-Miguel-Ayanz, 2013) (Appendix C).

In order to detect the effects of social-environmental change on the structure and functioning of ecological systems, surrogates of biodiversity have been widely used and proven to be suitable (e.g., bird communities - Larsen et al., 2012). Birds usually occur in relatively high densities and diversity and play various functional roles in food webs (e.g. Santos and Cabral, 2004; Bregman et al., 2014). Moreover, birds can be easily observed, and crowdsourcing data are often available via amateur bird watchers (e.g. Morelli et al., 2015). Birds are also considered as a group highly sensible to landscape changes and to trophic changes in the ecosystems (e.g. Bregman et al., 2014). For many bird species demography, behaviour, distribution and phenology are connected with spatiotemporal changes in LULC, like those induced by agricultural and/or agro-forestry practices, and they often recover in response to conservationist measures in previously disturbed ecosystems (Santos and Cabral, 2004).

In northwest Iberia, biological communities and especially bird communities are characterized by a combination of Mediterranean species (Specmeds), mostly associated with open habitats; habitat specialists (Specialists), occurring in specific habitats with

restrict distributions; and generalist species that have broader niches and are mostly Eurasian (Generalists) (e.g. Moreira et al., 2001b). Agriculture abandonment and afforestation are expected to have a positive effect on the occurrence of generalist species but also on some species with restricted niches (Specialists) and a negative effect on Mediterranean open habitat species (Specmeds) (e.g. Moreira et al., 2001b). These effects will be modulated by the initial state of the landscape and the occurrence of major disturbances, especially wildfires (Appendix C) (e.g. Zozaya et al., 2012).

Spatiotemporal modelling is an important tool for integrating ecological knowledge in effective landscape management (e.g. Santos et al., 2013), such as in the assessment of socio-ecological changes triggered by disturbances (Parrot and Meyer, 2012). Even simple models can improve our understanding of the mechanisms that disrupt ecological systems and support the identification of socio-ecological thresholds, the crossing of which makes recovery highly improbable (Parrot and Meyer, 2012, Suding and Hobbs, 2009). In this context, Agent Based Models (ABMs) have been proven to be particularly useful to study interdisciplinary problems and interactions between humans and the environment (e.g. Kelly et al., 2013; Ferreira et al., 2015). Additionally, the theoretic combination of reductionist methods (ABM derived) with holistic concepts was also suggested as a step forward for understanding socio-ecological process (Santos et al., 2013; Santos et al., 2015; Santos et al., in press).

In the present work a simple and flexible ABM derived approach was developed, integrating both reductionist techniques with holistic concepts within the same application (Santos et al., 2015). This approach, focusing on the interactions between biodiversity and habitat dynamics, should also be easily adaptable to other regions and case studies. We conducted a study of the likely responses of key surrogates of biodiversity to agricultural abandonment and/or afforestation for a test area

representative of the landscape of northwest Iberian Peninsula (Moreira et al., 2001a). Because biodiversity is dependent on particular habitat and landscape characteristics, we tested the following: (1) the modelled spatio-temporal trends for the landscape depict realistic LULC changes, (2) the bird community metrics selected are representative of the overall biodiversity, both varying in accordance with landscape changes and (3) the observed trends can be extrapolated to the real world and the model can be used as an supporting tool improving national biodiversity strategies and action plans (NBSAPs) implementation (Prip et al., 2010).

2. Material and methods

2.1. Selection of background research

In order to parameterize the model, we selected published studies providing quantitative information on bird communities for different habitats in northwest Iberia affected by land abandonment in relatively similar socio-economical contexts. The selected studies were required to meet the following criteria: data on bird species occurrence or richness per group (Specmeds, Specialists, and Generalists) and abundances for regions matching strong socioeconomic changes that led to widespread abandonment of traditional agricultural activities (e.g. Beilin et al., 2014). Bird community densities had to be assessed either by point-count censuses or by line transects. We found eleven studies matching these selection criteria (Appendix D).

2.2. Study area description

The model was developed for a virtual 10,000 ha landscape with the initial state representing the characteristics of the north-western Iberian landscape of the 1960s (Fig. 1) (Moreira et al., 2001a). In the 1960s land ownership was very fragmented, and the agricultural/urban areas (farmland) occupied approximately 40% of the study area, although as a fine-grained mosaic of small parcels of diverse crops including corn,

vineyards, potatoes and grasslands (Fig. 1). Shrublands cover was also nearly 40% of the area, and the most widespread shrub formations were dominated by species of the genera *Cytisus* and *Ulex* (Fabaceae) on granitic soils, or *Erica* (Ericaceae) and *Cistus* (Cistaceae) on shallow soils and in areas submitted to frequent fires. Forests and woodlands covered the remaining 20% of the area, dominated by conifer stands (mostly Maritime Pine, *Pinus pinaster*), deciduous forests (dominated by Oaks, *Quercus* sp.) and mixed forest (diverse combinations of *Pinus* sp., *Quercus* sp. and other species). The density of conifer stands, deciduous forests and mixed forests is typically lower than 600 individuals per ha, while the artificial stand plantations of eucalyptus (Blue gum, *Eucalyptus globulus*, planted since the 1970s) tend to be denser, but are typically less than 900 individuals per ha (Appendix A, Appendix B, Appendix C). We hypothesized that landscape changes in this region could be simulated from historical trends in the landscape: the major predicted changes were related to population decline, agricultural abandonment and afforestation – with Maritime pine and Blue gum–, contributing to an increase in fire prevalence and a decrease in landscape heterogeneity. Based on previous research (e.g. Moreira et al., 2001a; Moreira et al., 2001b), we further established that these landscape changes fostered biotic homogenization, decreasing the biodiversity overall (represented here by the bird community metrics) and compromising the NBSAPs (Appendix C).

2.3. Simulation model

The conceptual description of the model follows the specifications of the standard protocol ODD (Overview, Design Concepts and Details) to describe ABMs (Grimm *et al.*, 2010). The software Netlogo 5.3 (Wilensky, 1999) was chosen to perform the simulations of LULC and to calculate the biodiversity outcomes.

2.3.1. Overview

2.3.1.1. Purpose

The model investigates how the interplay between several processes related with the dynamics of various LULC, namely the increase of forest areas and the abandonment of farmland, has determined changes in the landscape and thereby also affected bird communities. Using this background as a demonstration, we intend to reproduce landscape pathways in order to understand and anticipate biodiversity changes associated with socio-ecological pressures related to agricultural abandonment and afforestation.

2.3.1.2. Entities, State Variables and Scales

The model includes two entities: patches (unit cells) that make up the landscape extent of the study area (10,000 ha) (Fig. 1) and small-scale farmers (land managers, our mobile agents, see Table 1 for details). Each patch was characterized by the dominant LULC, with each LULC type being associated to specific values of the following bird community metrics (surrogates for diversity): species richness, richness of Specialists, richness of Specmeds, and total abundance. The values used for model parameterization of each metric were based on averages of reference works selected according to the criteria described in section 2.1. (Appendices D and E, a). The time unit is the month, adjusted to comprehend realistic managing options and natural and disruptive changes (such as fire events) occurring in the landscape. Each simulation represented a possible evolution of the landscape for a period of 80 years encompassing both past tendencies and present and future possible LULC trends (1960-2040) (<http://web.letras.up.pt/asaraujo/Trabalhos/26%20Eixo.pdf>).

2.3.1.3. Process Overview and Scheduling

Each simulation followed a sequence of steps in order to replicate the evolution of the virtual landscape from 1960 to 2040. The simulation of spatial interactions is the

representation of spatial information and the neighbourhood structure: spatial interactions among patches were defined by nearest neighbours allowing more flexibility in spatial interactions than vector polygons and the transition rules were based on the LULC trends for all LULC classes evolved on each simulated dynamic scenario (Mas et al., 2014). Therefore, and for demonstration purposes, the simulation algorithm increases, for instance, the area of a given LULC through the identification of the susceptible to change nearest neighbours, inducing a consequent decrease in the respective area of former dominant LULC. In more realistic terms, since an initial type of LULC may result in several different LULC changes, this study analyses the various neighbourhood of a particular LULC change pattern type (Mas et al., 2014). During each simulation, the following sub-models are carried out in the following sequence: Abandonment (natural succession); Farmer management; Plantation (logging, replantation); Wildfire; Measurement of the Bird diversity metrics. Details concerning initial conditions and processes are explained below and depicted in Appendix F.

2.3.2. Design Concepts

2.3.2.1. Emergence

The pathway of the diverse LULC and associated biodiversity emerges from the complex interaction between abandonment trends, forest investments and initial landscape structure. Fire intensity and extension also emerges from the type of forests and landscape patchwork.

2.3.2.2. Sensing

Every week the virtual small-scale farmers are programmed to have absolute perception of the LULC in neighbouring patches, moving to a tall shrubland in order to reproduce the traditional practices.

2.3.2.3. Interaction and Collectives

Tall shrublands are transformed to low shrublands by small-scale farmers. No collectives were formed.

2.3.2.4. Stochasticity

All variables related to the patches were assigned stochastically, based on realistic landscape metrics. In addition the model includes a number of stochastic processes, particularly regarding the parameters that regulate the LULC dynamics and fire episodes that depend on the scenario chosen and on the prevailing landscape characteristics.

2.3.2.5. Observation

Most results concerning LULC, fire, richness, richness of Specmeds, richness of Specialists and abundance per patch and for the all landscape were obtained at the end of each simulation. At any time anyhow, was possible to know the exact location of each small-scale farmers and the characteristic of each patch, if needed.

2.3.3. Details

2.3.3.1. Generic overview

The key actors in the region (mobile agents of the model) are small-scale farmers that manage the landscape by an intensive usage of farmland supplemented by resources of the native forest and shrublands such as timber, organic matter and food for communal herds in mountain heathlands. Before the current trends of depopulation and agricultural abandonment, namely until the 1960s, the natural succession of most patches was averted by the intensive land management. The landscape structure was pictured by a complex patchwork of shrublands, forests and agriculture, and fire events were controlled due to the dominance of fire suppressive LULCs (Appendix A). After the 1960's the population started the migration to the biggest cities in Portugal, and even to central European countries, promoting the abandonment of the less productive areas and

the loss of the communal herds (Appendix B). This reflected on the increasing of tall shrubland areas and the natural forests by re-naturalization (succession). Nonetheless, after the 1970s, the shrubland / natural forest areas suffered a competition for space with the industry demand for wood, increasing the areas of forests plantations, namely fast-growing exotic species such as eucalyptus (Appendix C). The landscape patchwork became more homogenous, with large areas of uniform LULC, usually shrublands and planted forest stands (Appendix C).

2.3.3.2. Initialization

Landscape and small-scale farmers

The initial values of the variables *Agriurban* and *Shrublands* were set to reproduce a landscape dominated by continuous areas of farmland (circa 40 % of the patches) surrounded by shrublands (circa 40% of the patches) and diverse types of natural and semi-natural forests (20% of the patches, dominated by conifer and mixed forests but with a small percentage of deciduous) (Moreira et al., 2001a; Moreira et al., 2001b). Depending on the scenario chosen the variable *Abandonment* was defined as 0.0000 (scenario 1), 0.0008 (scenarios 2 and 3) or 0.0016 (scenario 4) for reproducing no abandonment of farmland or actual rates of farmland reduction for different locations (Pinto-Correia and Vos, 2004; Lopes et al., 2013). The variable *Plantation-effort* was inserted with two different values to reproduce different patterns of plantation effort associated to each scenario. The variables *Cutting-year-eucalyptus* and *Cutting-year-conifer* were held constant with values 12 and 50 to reproduce the average years until logging of Blue gum and Maritime Pine, respectively (<http://www.icnf.pt/portal/florestas>). The *Resprouter-rate* variable, reproducing the recovery of vegetation was parametrized for the model in accordance with reference information (Calvo-Iglesias et al., 2009). The number of farmers was defined for

reproducing the average number of monthly active small-scale farmers that associate forestry and livestock to agriculture (<http://ec.europa.eu/eurostat/web/agriculture/farm-structure>). These active small-scale farmers move to a specific tall shrubland and transform the tall shrubland in a low shrubland.

2.3.3.4. Processes overview

2.3.3.4.1. Abandonment (natural succession)

Each patch dominated by low-statured shrublands increases its biomass and becomes a tall shrubland over time. The tall shrubland can evolve to become a natural forest (considering the neighbouring patches) or, in the case of low shrublands emerging from a wildfire, recovering to the previous forest (Fig. 2, Abandonment). The farmland patches might enter this process along the simulation when abandoned, by conversion into low shrublands (Fig. 2, Abandonment). Details concerning Abandonment are depicted in appendix F.

2.3.3.4.2. Farmer management

Small-scale farmers obtain their resources through an active management of the landscape by an intensive use of the farmland patches, and by extensive management of the native forest and shrublands by gathering timber, litter and food for communal herds in mountain heathlands. Each smaller scale farmer thus chooses randomly a tall shrubland in a specific radius around its initial location (fixed in a farmland patch) and converts it into a low shrubland patch (Fig. 2, Management). This simulates the traditional organic matter collection from the outer common lands into the farmland areas around settlements. All the other LULC, including low shrublands used for several purposes, remain in that same LULC type throughout simulations, provided that the associated management is maintained (Fig. 2, Management). Details concerning Farmer management are depicted in appendix F.

2.3.3.4.3. Plantation (logging, replantation)

Forest plantations comprise conifer forests and eucalyptus stands. Most of the LUUC, with the exception of deciduous and mixed forest, can be converted into forest plantations (Fig. 2, Plantation). The rate trends for plantations were based on historical information for northwest Iberia (Mendes et al., 2004; Calabuig et al., 2000). Planted forest patches are logged considering the average time period that each forest type is usually harvested (Thomas et al., 2000) (Fig. 2, Plantation). Typically, after the harvesting, the same type of forest is replanted, although transitions among planted forests might occur, depending on the scenario (Thomas et al., 2000). Details concerning Plantation are depicted in appendix F.

2.3.3.4.4. Wildfire

Single or multiple fire ignitions can occur randomly or by specifying the spatial coordinates of the ignition cells. Some of the processes that affect fire spread across the landscape are considered, including diffusive spread between patches and the type of LULC (Hargrove et al., 2000). Fire spreading from each ignited patch to any of eight unburned neighbours is considered as an independent stochastic event with a probability associated to the LULC type of the neighbouring patches. The numerical value of this critical threshold is a function of the number of neighbours and the considered rule to determine the spread towards the neighbours within a given time step. The model explicitly simulates different fire proneness by changing the probability of fire spread as a function of the patch LULC: conifer forests, low and tall shrublands were considered highly flammable, other forests and eucalyptus stands intermediate, while farmland and burned patches were considered non-flammable (Silva et al., 2009; Carmo et al., 2011) (Fig. 2, Wildfire). Details concerning Wildfire are depicted in appendix F.

2.3.3.4.5. Bird diversity metrics

Bird diversity associated to each patch is described considering four metrics: total richness, richness of Mediterranean species (Specmeds), richness of specialists (Specialists) and total abundance. For each LULC type, values of bird diversity for the four metrics were attributed based on selected previous studies (Appendix D and Appendix E), although the final calculation for each metric was derived from the interaction between the patch and the adjacent neighbours. (Appendix E, b) (Devictor and Jiguet, 2007). For exemplification we illustrate in Appendix G the conceptualization of the metric richness calculation for a specific patch.

2.4. Sensitivity analysis

According to Lee et al. (2015) the purpose of a sensitivity analysis is provide a measure of the robustness of the model, measuring the sensitivity of the obtained results to changes in parameters, forcing functions and /or sub-models. Local sensitivity analysis (SA) was done by one-parameter-at-a-time technique (OAT), changing the population parameters of the model with +/- 10 % and +/- 50% variation of the respective values and observing changes in the response of the selected state variables (Ligmann-Zielinska, 2013). Global SA, measuring the effect of combined parameters of a model in terms of sensitivity, was performed by estimating standardized regression coefficients (SRC) associated to a matrix of combinations of the parameters with +/- 10% variation of the respective values (Lee et al., 2015). SRC expresses the magnitude and significance of the effect of combined parameters measured using different units, as well as the explained variance, gauging the main effects of the input parameters. A stepwise multiple regression analysis was used to test for relationships between state variables and parameters. A step down procedure was applied so that the effect of each parameter in the presence of all others could be examined first with the least significant parameter being removed at every step. The analysis stopped when all the surviving

parameters had a significance level <0.05 (Zar, 1996). The lack of substantial intercorrelation among parameters was confirmed by the inspection of the respective tolerance values. The statistical analysis were carried out using the software SYSTAT 8.0[®].

2.5. LULC scenarios

Scenario analysis is an important methodology in environmental sciences for analysing the effects of several driving forces and for assessing the associated uncertainties (Bryan et al., 2011). We based LULC trends on historical records for the region, namely tendencies from 1958 to 1995 (Moreira et al., 2001a; Calabuig et al., 2000). The LULC scenarios were selected to illustrate the impacts of different socioeconomic changes on landscape dynamics and especially on biodiversity, using bird diversity metrics as surrogates. We expected to predict with realism the landscape changes that have been occurring in the different areas of the region, project “future landscapes”, and evaluate how these changes impact the bird diversity metrics, especially for some specific groups and overall abundance (Moreira et al., 2001b).

We considered four scenarios, trying to project realistic future landscapes from similar initial conditions:

scenario 1, maintaining the traditional management and assuming the historical dynamics at the landscape level; although improbable, this scenario is useful for contrasting with the other scenarios (baseline scenario);

scenario 2, population decline, rural abandonment and investment in conifer forests, observed in sub-mountainous areas;

scenario 3, population decline, abandonment with investments in conifer forests and eucalyptus stands, common in many areas of the region; and

scenario 4, massive population decline and abandonment of crop and animal farming, with massive investments in eucalyptus stands, expected for depressed areas nearby paper-pulp industries.

2.6. Comparing the different scenarios

Considering that most of the data did not show a normal distribution and other parametric assumptions (Shapiro-Wilk W test), we performed the comparative statistical analysis using non-parametric tests (Royston, 1993). The Mann-Kendall non parametric regression (MK) test (Zar, 1996) was applied to detect significant trends for each scenario and simulated period, considering each LULC present in the study area. The MK test can be used instead of a parametric linear regression analysis to test if the slope is different from zero (H_0 : no trend). We applied the same test for detecting possible trends in fire occurrences along the simulation period. We applied the nonparametric Wilcoxon test (Zar, 1996) to determine differences in the bird metrics between the initial condition and two selected time frames per scenario (intermediate t, which coincides with the year 2000, and the final t of the simulation period, representing the year 2040). Additionally, Cohen's effect size was quantified (Sullivan and Feinn 2012) to reveal the magnitude of the differences in the diversity metrics for the selected time frames. All statistical analyses were performed using Mypstat 12[®] and GenStat[®] 12th Edition.

3. Results

3.1. Sensitivity analysis

The results from the OAT sensitivity analysis show that the parameters associated with plantation of fast-growing species (Plant), the abandonment of farmland (Abandonment) and the initial area occupied by farmland (Farmland) cause the main individual changes on the modelling outcomes, specifically our surrogates of diversity

(Appendix H, Table H1). From the analysis of the SRC is possible to detect the global sensitivity of all surrogates of diversity to farmland (Farmland), corroborating the results obtained in the local sensitivity analysis (Appendix H, Table H2). Abandonment of farmland (Abandonment), the year eucalyptus are logged (Logging-eucalyptus), the initial number of farmers (Number-of-farmers) and the probability of a planted forest being logged (Cut) are also parameters with key influence on the outputs of most surrogates (Appendix H, Table H2). Nevertheless, all surviving parameters to the stepwise procedure can be considered with influence on the model outcomes, even in the presence of variance in the other parameters.

3.2. Landscape dynamics and biodiversity outcomes

Model simulations using the four LULC scenarios revealed important differences in landscape futures. In the first scenario a decrease in the conifer forest area and an increase in the deciduous forest, mixed forest and shrubland areas were predicted both for 2000 and 2040 (Fig. 3) illustrating the LULC dynamics in this landscape for the baseline scenario. The results of the MK test confirm the visual inspection, together with the inexistence of significant trends for both the farmland patches, and fire occurrences (significant test for fire but with no trend detected, probably associated to the fluctuating patterns of fire occurrence along the simulation period; Table 2). Fire prone LULC, such as conifer forests and shrublands, were combined with barriers of reduced flammability, specifically patches dominated by farmland, deciduous forests and mixed forests, controlling fire occurrence and magnitude. No significant changes were predicted for the bird diversity metrics in response to the detected LULC trends. This is illustrated in figure 3 for richness and confirmed by the calculated median differences and effect sizes between 1960, 2000 and 2040 for richness, richness of Specialists, richness of Specmeds and abundance (Table 3).

LULC changes predicted and projected using scenario 2 indicate positive trends in the forest area, particularly conifer forests planted and logged for commercial uses, although also for “natural” mixed and deciduous forests and tall shrublands. A negative trend is predicted for farmland, and low shrubland LULC (Fig. 4 and Table 2). These landscape changes, associated with a decrease in the rural population have obvious effects in fire occurrences and extension, confirmed by the MK test results (Table 2). The LULC change is expected to create a less diverse community, as shown in the decrease for most biodiversity metrics, while with a large increase in the overall bird abundance (Fig. 4 and Table 3). Nevertheless Specialists and Specmeds exhibit a dramatically richness decline, especially for 2040, confirmed by the Cohen’s effect size test results (Table 3).

In scenario 3 the MK test shows positive trends for eucalyptus plantations but also mixed, deciduous forests and tall shrublands LULC (Fig. 5 and Table 2). Fire occurrences are expected to increase in a similar way as in scenario 2, and conifer forests are expected to have a moderate decrease (Table 2). The decrease of conifer forest area is probably due to both fire susceptibility associated and long cycles of planting and logging (circa 50 years). The decline in farmland as well as low shrublands is also similar to the one observed in scenario 2 (Fig. 5 and Table 2). While with similar trends for most LULC, the expected changes in the diversity metrics considering this scenario are much more severe than in the scenario 2. Actually, richness is predicted to have a large decrease, and a very large decline is expected for Specialists and Specmeds richness (Cohen’s effect size) (Fig. 5 and Table 3), especially for 2040. Nonetheless the expected trend for birds’ abundance suggests maintenance values for 2000 or even a small increase for 2040 (Table 3).

Finally, for scenario 4, the major expected trends are associated to large declines in the population (twice the scenarios 2 and 3), represented by decline in the farmland patches combined with intense afforestation of fast growing species. Massive landscape changes occur along the simulation period as well as the increase in eucalyptus stands plantations (Fig. 6). The results of the MK test confirm the visual inspection and also demonstrate the expansion of tall shrubland areas, associated with abandonment tendencies and fire occurrences, tending to increase faster when comparing with the previous scenarios, specifically scenarios 2 and 3 (Table 2). All other LULC are expected to have a significantly decrease, namely low shrublands, conifer forests and farmland areas (Table 2). All biodiversity metrics used as surrogates for diversity assessments, richness, richness of Specialists, richness of Specmeds and abundance, shown large declines, exhibiting an overall change in the community to accommodate to this “novel” landscape (Fig.6 and Table 3).

4. Discussion

4.1. Spatio-temporal changes in the landscape

Our study aimed to inform decision makers about potential future outcomes of biodiversity under different landscapes built on diverse land use /land cover (LULC) trends, helping the implementation of local conservation actions associated with National Biodiversity Strategies and Action Plans (NBSAP). Moreover the modelling outcomes could be used to support strategic thinking about local landscapes planning and the guidance of environmental/agriculture policies and investments. Specifically for the Northwest of Iberia, our study considered real policy processes associated with depopulation, assumed to be the most important driver modifying regional economy and consequently LULC through time. As a consequence of an aging population and migration the region is faced with agricultural abandonment and a considerable private

afforestation (Santos et al., 2006). The four modelled scenarios illustrate the impacts of different contrasting, but realistic, socio-ecological changes on biodiversity (expressed by bird diversity metrics) and might support promising actions required to achieve the national targets for biodiversity determined for the North of Portugal (<https://www.cbd.int/doc/world/pt/pt-nr-04-en.pdf>) and Galicia regions (<https://www.cbd.int/doc/world/es/es-nr-05-es.pdf>). We have focused this study on realistic scenarios already observed in diverse locations across the region, recognising that processes of planning and management involve multiple social and ecological objectives that differ between location and context. Although considering that the obtained landscape evolution and projections underestimated some realistic LULC tendencies for the region, the general pattern highlights the relationships between agricultural abandonment, fire risk and biodiversity. Moreover, the LULC types that are more resistant to fire (farmland) have been progressively replaced, in most of the scenarios, by fire-prone LULC types (shrublands and forests). Previous studies conducted at local and regional levels have found similar patterns of agriculture abandonment, shrub encroachment and increased fire frequency (e.g. Moreira et al. 2001a; Calvo-Iglesias et al., 2009), which supports the robustness and seems to confirm the interest and relevance / potential application of our modelling approach.

4.2. Effects of LULC scenarios on biodiversity surrogates and implications for the implementation of NBSAPs

The selected LULC scenarios show that the abandonment of traditional agriculture together with afforestation generates a less diverse, homogeneous landscape, with negative impacts on the objectives of NBSAPs: our biodiversity surrogates show decreasing trends, especially the most emblematic specialist species and Mediterranean restricted species. When applying scenarios 2 and 3, the decline of traditional

agriculture co-occurs with forestry associated with novel management paradigms, showing similar trends towards a landscape dominated by planted forests. Nevertheless the differences in the outcomes of the biodiversity surrogates show that different policy interventions may substantially influence the socio-ecological systems. This result is particularly dramatic for scenario 4, which illustrates a currently common landscape in several locations of the study area, and probably dominant in the near future across Galicia and North of Portugal, threatening the implementation of NBSAPs in this region (Santos et al., 2006; Calvo-Iglesias et al., 2009; Carvalho-Ribeiro et al., 2010).

The results obtained confirm the hypothesis that recent landscape changes promote significant impacts on biodiversity patterns in north-western Iberia as already shown at a local scale (e.g. Calvo-Iglesias et al., 2009). The increase in species abundances associated with LULC changes masks intricate patterns of temporal changes in responses (diversity and / or species richness levels). More generalist species are affected by the LULC changes, but specialists and Mediterranean species are the least favoured, in agreement with broad patterns observed in Europe (Midterm review of the EU Biodiversity strategy to 2020, http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/mid_term_review.pdf). However, the impact on biodiversity was not detected in scenario 1, confirming the importance for biodiversity conservation of maintaining traditional agricultural and landscape management. Indeed, traditional management practices can guide the regional implementation of NBSAPs actions in human-dominated landscapes (Fischer et al., 2012). The spatially explicit simulation of the model, plus the analysis of the importance of future landscape projections in explaining the diversity metrics changes, suggests the existence of diverse drivers shaping the regional patterns. Thus, while succession changes previously farmed and grazed habitats into shrublands, human

actions convert the previous LULC into planted forests that are less diverse in specialists and Mediterranean species and more prone to fire (Oliveira et al., 2014). Abandoned shrublands are converted naturally into forests, increasing abundance and the richness of generalists and some specialists' bird species, but also increasing fire probability (Moreira et al, 2001b; Oliveira et al., 2014). Fire, the ultimate LULC shaper (Silva et al., 2011), creates new shrubland areas, specifically low shrublands encompassing an interesting proportion of specialists and Mediterranean species (Moreira et al, 2001b).

4.3. Methodology suitability for supporting NBSAPs

NBSAPs are considered to be an effective and widely used tool for taking a strong sustainability approach to environmental planning but, in most cases an evaluation a priori is still lacking (Margules and Pressey, 2000). NBSAPs objectives are ambitious: (1) to develop national strategies, plans or programmes for the conservation and sustainable use of biological diversity or adapt for this purpose existing strategies, plans or programmes, and (2) to integrate, as far as possible and as appropriate the conservation and sustainable use of biological diversity into relevant sectorial or cross-sectorial plans and policies (<https://www.cbd.int/nbsap/introduction.shtml>). NBSAPs models that predict future landscapes enable the integrated assessment of the impacts of different socio-ecological changes on biodiversity (Parrot and Meyer, 2012; Santos et al., 2013; Bastos et al., 2015). Modelling also allows the evaluation of the influence of internal processes and their performance under scenarios of change induced by external drivers (Nagendra et al., 2013), essential to inform strategic spatial prioritisation in order to efficiently implement NBSAPs. The methodology introduced here offers unique features that complement the outcomes of other spatio-temporal approaches such as Cellular automata, Species Distribution Models (SDM), Stochastic-dynamic-

Methodology (StDM), ABM among others when studying the impacts of landscape changes on biodiversity (Santos et al., 2015). In fact StDM, SDM, ABM and Cellular automata have all proved successful in a number of works associated with predicting the response of diversity to LULC dynamics but none represents a "best" approach or provides a universal procedure (Santos et al., 2013; Santos et al., 2015). In holistic approaches, such as SDM, many relevant details are not considered, such as individual variability, adaptation, local idiosyncrasies, and highly dynamic ecological phenomena. Although this simplification enables the understanding of whole-system processes it is difficult to follow at other scale approaches, namely when local properties are crucial to global system behaviour. This is one of the major advantages of methods such as ABM (Grimm, 1999; Santos et al., 2013): to enable information crossing at a local scale, potentiating a better understanding of ecosystem dynamics, and leading to emergence and similar system characteristics. The presented model concept has emerged from the idea of so-called hybrid models (Santos et al., 2015): modelling ecosystem changes on the basis of large-scale holistic relationships while at the same time considering the most important processes. Its advantage as a general spatio-temporal approach is due to the possible integration within the same software and application of predetermined dynamic LULC changes trends coupled with external and internal driving forces from where biodiversity patterns emerge.

5. Conclusions

Climate, LULC and resource management constrain disturbance regimes shaping agricultural landscapes that are highly dependent on population density, economic value of each crop, plant diseases, and other socio-ecological factors (e.g. Fernandes et al., 2013). LULC shifts and effects associated with these highly dynamic landscapes are only partially understood (e.g. Moreira et al., 2001a; Fernandes et al., 2013). On the

other hand, environmental policies and strategies must contribute to, and follow, LULC planning decisions and projections of possible outcome scenarios. Thus, it is fundamental to characterise and understand the main cause–effect relationships between changes in the landscape and ecosystem responses (Santos et al., 2013; Santos et al., in press). Enhancing and / or maintaining biodiversity and the ecosystem services can be the basis for adapting the landscapes to global and regional changes, as well as to the design of risk-decreasing strategies as they can deliver benefits that will increase the resilience of socio-ecological systems (Larigauderie and Mooney, 2010). Current biodiversity patterns in northwest Iberia are affected by large-scale changes such as land abandonment and afforestation. In terms of bird biodiversity, which we used as a surrogate for biodiversity assessments, our model predicts that the most affected species are those typical from Mediterranean landscapes as well as habitat specialists, often composed by species with a high conservationist status and considered surrogates of overall biodiversity (Flohre et al., 2011; Herrando et al., 2014). Enhanced predictions of LULC changes can help the efficient design of improved strategies, emphasizing the need to better understand the combined effects of constraints imposed by policies. In order to support our findings, more studies are needed, including improved quantifications of the relation between surrogates and overall diversity, which is currently not fully understood or recognized (e.g. Flohre et al., 2011; Larsen et al., 2012; Gao et al., 2015). While modelling can offer a basis for decision-making considering policy options, accurate parameterization and robust validation using field data is essential for the application to real landscapes. Despite the inherent limitations of an academic demonstration, our approach can be easily adapted for other areas, modifying the context-specific models and simulation rules. Despite all uncertainties of a global world where ecosystems are being transformed at extraordinary scales, rates

and speed, we consider that modelling regional biodiversity trends in future projected landscapes will become a fundamental practice to inform decision makers in choosing the best viable conservation actions for implementing improved NBSAPs.

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Figure 1

Figure 2

Conceptual model diagram

Figure 3

a

b

1960

2000

2040

Figure 4

a

b

1960

2000

2040

Figure 5

a

b

1960

2000

2040

Figure 6

a

b

1960

2000

2040

Table 1 – Description of the variables of each conceptual entity used in the model.

Entity	Variable	Brief description
Patches	<i>Number</i>	10,000 patches arranged in a square grid which corresponds to a total of 10,000 ha.
	<i>Size</i>	Individual patch – 1 ha square (100 x 100 meters).
	<i>Soil-Use</i>	Each patch contains a single class of land cover: we considered the dominant land use by patch: <i>mixed forests</i> (MF); <i>conifer forests</i> (CF); <i>deciduous forests</i> (DF); <i>farmland</i> (A); <i>tall shrublands</i> (TS); <i>low shrublands</i> (LS); <i>eucalyptus plantations</i> (EP). An additional land covers is represented by burned areas (F).
	<i>Agriabandonment</i>	Probability, defined by the user, of each farmland patch becoming abandoned, converting into low shrublands (LS)
	<i>Fire-start</i>	Probability of burning (F), considering the type of soil-use. Defined from published works.
	<i>Fire-spread</i>	Probability of contagion burning considering the type of soil-use and the neighbours' condition (burning or not burning).
	<i>Resprout</i>	Probability of vegetation recover after burning
	<i>Mature</i>	Probability of forest recover considering the previous dominant soil-use and the dominant patches neighbours' soil-use
	<i>Plant</i>	Probability of being planted with conifer forests and / or eucalyptus plantations.
	<i>Cut</i>	Probability of the forest being logged, considering the average years that planted forests are exploited.
	<i>Estimate-Richness</i>	Average bird richness for the patches by soil-use, considering the influence neighbouring patches.
Small-scale farmers (Managers)	<i>Number</i>	Defined by the user, 1/25 of the number of farmland patches (A).
	<i>Search-shrublands</i>	The manager searches a patch of tall shrublands (TS), in a radius defined by the user.
	<i>Manage-shrublands</i>	The manager reaches a patch of tall shrublands (TS) and controls vegetation, transforming in a low shrubland (LS) patch.

Table 2 – Results of the Mann-Kendall test (MK) for detecting trends in the modelled landscape (H0, no trend versus H1, two-sided trend) for the four scenarios considered for a period of 80 years. Scenario number (Sc), Number of distinct time points (N) (tied observations in parentheses), Slope (95% confidence limits in parentheses), Mann-Kendall statistic (MK), the asymptotic standard error (ASE) and associated z value with their significance level (non significant, n.s.; ** - $p < 0.01$; *** - $p < 0.001$) and trends (slope = 0, = (no trend); $0 < \text{slope} < 0.250$, ↑ (small increase); $0.250 < \text{slope} < 0.500$, ↑↑ (medium increase); slope > 0.500 , ↑↑↑ (large increase); $0 > \text{slope} > -0.250$ ↓ (small decrease); $-0.250 > \text{slope} > -0.500$ ↓↓ (medium decrease); slope < -0.500 , ↓↓↓ (large decrease)). Details of all variables are given in Table 1 and Appendix F.

Variable	Sc	N	Slope	MK	ASE	Z-value
Conifer forests	1	961 (374)	-0.636 (-0.659; -0.616)	-448,772	9,935.507	-45.168 (***) ↓↓↓
	2	961 (753)	2.422 (2.401; 2.442)	454,947	9,938.051	45.778 (***) ↑↑↑
	3	961 (276)	-0.373 (-0.404; -0.349)	-362,385	9,936.642	-36.469 (***) ↓↓
	4	961 (411)	-0.750 (-0.775; -0.722)	-444,340	9,933.613	-44.731 (***) ↓↓↓
Mixed forests	1	961 (58)	0.041 (0.038; 0.044)	182,021	9,934.383	18.322 (***) ↑
	2	961 (324)	0.496 (0.482; 0.508)	366,235	9,937.704	36.853 (***) ↑↑
	3	961 (410)	0.917 (0.888; 0.948)	398,166	9,937.533	40.067 (***) ↑↑↑
	4	961 (368)	-0.539 (-0.543; -0.534)	-458,737	9,937.847	-46.161 (***) ↓↓
Deciduous forests	1	961 (88)	0.105 (0.102; 0.108)	389,107	9,933.044	39.173 (***) ↑
	2	961 (264)	0.362 (0.356; 0.368)	393,644	9,936.694	39.615 (***) ↑↑
	3	961 (381)	0.679 (0.660; 0.699)	436,082	9,937.285	43.883 (***) ↑↑↑
	4	961 (135)	-0.160 (-0.161; -0.159)	-452,257	9,936.338	-45.515 (***) ↓
Fire occurrences	1	961 (16)	0.000 (0.000; 0.000)	-25,675	9,874.277	-2.600 (**) =
	2	961 (48)	0.019 (0.018; 0.021)	228,785	9,928.608	23.043 (***) ↑
	3	961 (45)	0.016 (0.015; 0.017)	205,283	9,927.974	20.677 (***) ↑
	4	961 (111)	0.063 (0.060; 0.066)	291,153	9,936.791	29.300 (***) ↑
Farmland	1	961(-)	0.000 (0.000; 0.000)	0,000	-	0 (n.s) =
	2	961 (875)	-2.526 (-2.549; -2.502)	-461,182	9,938.061	-46.406 (***) ↓↓↓
	3	961 (875)	-2.526 (-2.549; -2.502)	-461,182	9,938.061	-46.406 (***) ↓↓↓
	4	961 (912)	-3.595 (-3.659; -3.532)	-461,228	9,938.064	-46.410 (***) ↓↓↓
Tall shrublands	1	961 (322)	0.429 (0.405; 0.450)	304,538	9,937.646	30.645 (***) ↑↑
	2	961 (443)	0.799 (0.742; 0.857)	355,329	9,937.939	35.755 (***) ↑↑↑
	3	961 (391)	0.893 (0.834; 0.970)	283,950	9,937.801	28.573 (***) ↑↑↑
	4	961 (621)	1.937 (1.903; 1.970)	380,448	9,938.027	38.282 (***) ↑↑↑
Low shrublands	1	961 (152)	0.054 (0.048; 0.061)	174,873	9,936.630	17.599 (***) ↑
	2	961 (661)	-1.570 (-1.582; -1.558)	-441,769	9,938.041	-44.452 (***) ↓↓↓
	3	961 (623)	-1.545 (-1.555; -1.534)	-443,893	9,938.034	-44.666 (***) ↓↓↓
	4	961 (588)	-0.852 (-0.903; -0.798)	-262,121	9,938.025	-26.375 (***) ↓↓
Eucalyptus plantations	1	n.a.	n.a.	n.a.	n.a.	n.a. =
	2	n.a.	n.a.	n.a.	n.a.	n.a. =
	3	961 (663)	2.086(2.050; 2.117)	449,178	9,938.028	45.198(***) ↑↑↑
	4	961 (787)	3.404(3.308; 3.500)	440,968	9,938.053	44.372(***) ↑↑↑

Table 3 – Changes in the number of bird species (Final richness), number of habitat specialists (Final Specialists richness), number of open habitat mediterranean species under conservation concern (Final Specmeds richness) and total number of birds (Final abundance) for the four scenarios considered. Scenario number (Sc), difference between medians (Median difference) of the initial situation versus 40 years and 80 years after the implementation of the respective scenario (Wilcoxon test and their significance level in parentheses; n.s. - non significant;*** - $p < 0.001$), Effect sizes (95% confidence limits in parentheses) and Effect sizes conventions (not detected, =; ↑, small increase; ↑↑, medium increase; ↑↑↑, large increase; ↑↑↑↑, very large increase; ↓, small decrease; ↓↓, medium decrease; ↓↓↓, large decrease; ↓↓↓↓, very large decrease).

Period		0 – 40 years		0 – 80 years	
Variable	Sc	Median difference	Effect size (d)	Median difference	Effect size (d)
Final richness	1	0.00 (18534208***)	-0.07 (-0.11; -0.05) =	0.00 (20480058***)	-0.06(-0.09;-0.03) =
	2	-0.30 (32283014***)	-0.28 (-0.31; -0.26) ↓	-0.30(32024880***)	-0.28(-0.31;-0.25) ↓
	3	-0.95 (47295310***)	-0.86 (-0.89; -0.83) ↓↓↓	-1.14(32024880***)	-0.96 (-0.99; -0.93) ↓↓↓
	4	-2.73 (51943464***)	-2.33 (-2.36; -2.29) ↓↓↓↓	-3.68(52014803***)	-3.43 (-3.47; -3.39) ↓↓↓↓
Final Specialists richness	1	0.00 (15126746 n.s.)	0.00 (-0.03; 0.03) =	0.00 (2018374 ***)	0.01 (-0.02; 0.04) =
	2	-1.00 (46932357***)	-0.78 (-0.81; -0.75) ↓↓	-1.63 (50445400 ***)	-1.39 (-1.42; -1.36) ↓↓↓↓
	3	-1.25 (40625308***)	-0.91 (-0.94; -0.89) ↓↓↓	-2.00 (46831838 ***)	-1.53 (-1.56; -1.49) ↓↓↓↓
	4	-2.13 (51375702***)	-2.30 (-2.33; -2.26) ↓↓↓↓	-3.13 (51548395 ***)	-3.27 (-3.31; -3.23) ↓↓↓↓
Final Specmeds richness	1	0.00 (2018374***)	0.09 (0.06; 0.12) =	0.00 (3928577***)	0.10 (0.07;0.12) =
	2	-0.38 (41885469***)	-0.73 (-0.75; -0.70) ↓↓	-0.75 (48976333***)	-1.44 (-1.47;-1.44) ↓↓↓↓
	3	-0.63 (45375563***)	-1.07 (-1.10; -1.04) ↓↓↓	-1.13 (50777418***)	-1.83 (-1.86;-1.80) ↓↓↓↓
	4	-1.80 (51789722***)	-2.54 (-2.57; -2.50) ↓↓↓↓	-2.38 (51840413***)	-3.49 (-3.54;-3.45) ↓↓↓↓
Final abundance	1	0.03 (24387409***)	0.07 (0.04; 0.09) =	0.06 (23489813***)	0.12 (0.09;0.15) =
	2	1.39 (7407153***)	0.69 (0.66; 0.72) ↑↑	2.98 (2200125***)	1.27 (1.24;1.30) ↑↑↑
	3	0.00 (25299508 n.s.)	-0.01 (-0.04; 0.02) =	1.13 (17297407***)	0.39 (0.36;0.42) ↑
	4	-5.03 (51818704***)	-2.22 (-2.25; -2.19) ↓↓↓↓	-5.90 (51389224***)	-2.26 (-2.30;-2.22) ↓↓↓↓