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Future land-use effects on the connectivity of protected area networks in southeastern Spain

María Piquer-Rodríguez a,*, Tobias Kuemmerle b, Domingo Alcaraz-Segura c, d, Raul Zurita-Milla e, Javier Cabello c

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Land-use change is a major driver of the global biodiversity crisis, mainly via the fragmentation and loss of natural habitat. Although land-use changes will accelerate to meet humankind’s growing demand for agricultural products, conservation planning rarely considers future land uses and how they may affect the connectivity of ecological networks. Here, we integrate land-use models with landscape fragmentation and connectivity analyses, to assess the effects of past and future land-use changes on the connectivity of protected area networks for a highly dynamic region in southeast Spain. Our results show a continued geographical polarisation of land use, with agricultural intensification and urban development in the coastal areas, and the abandonment of traditional land use in the mountains (e.g., 1,100 km² of natural vegetation are projected to be lost in coastal areas whereas 32 km² of natural vegetation would recover in interior areas from 1991 to 2015). As a result, coastal protected areas will experience increasing isolation. The connectivity analyses reveal that the two protected area networks in place in the study area, the European “Natura 2000” and the Andalusian “RENPA” networks, include many landscape connectors. However, we identify two areas that currently lack protection but contain several important patches for maintaining the region’s habitat connectivity: the northwestern and southwestern slopes of the Sierra Cabrera and Bédar protected area. Our results highlight the importance of considering future land-use trajectories in conservation planning to maintain connectivity at the regional scale, and to improve the resilience of conservation networks.

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Introduction

Land-use change is the main driver of current biodiversity decline due to the loss, fragmentation, and degradation of natural habitats (CBD 2010; MA 2005). Protected areas are a cornerstone of conservation efforts worldwide and are generally effective in preventing habitat loss inside them (Andam et al. 2008; Radolff et al. 2010). However, land-use changes in the surroundings of protected areas increasingly isolate protected areas (Newmark 2008; Seifering et al. 2011). To avoid such isolation, conservation planning needs to move from site-based protection towards more holistic approaches that consider the landscape as a whole (Naveh 2000). Hence, the challenge for decision makers is to prioritise conservation efforts that maintain the functionality and connectivity of habitat networks (Knight et al. 2006). This is particularly important in the face of climate change, since maintaining functionally connected landscapes is the single most effective conservation action to allow species to adapt to a changing environment (Heller and Zavaleta 2009; Pullin et al. 2009).

Many regions of the world are simultaneously experiencing both loss and recovery of natural habitats due to land-use change (Barbier et al. 2010). On the one hand, the expansion and intensification of human land uses to satisfy the ever-growing demand of natural resources (e.g. food, wood, fiber, bio-energy) is a major driver of habitat loss (Keys and McConnell 2005). On the other hand, urbanisation and industrialisation often result in a decline of rural populations and traditional land uses, triggering ecosystem
recovery in abandoned lands (Kuemmerle et al. 2011; MacDonald et al. 2000). The co-occurrence of land-use expansion (here: converting land that was under alternative use to agriculture) and land abandonment (here: conversion of agriculture to natural vegetation) result in complex land-use change effects on habitat connectivity. This poses challenges to conservation planning with the goal to identify effective and resilient protection networks.

Traditionally, prioritising areas for conservation has been based on the selection of a number of sites to reach an overall conservation goal (e.g., protection of 10% of all habitats or 95% of all threatened species) (Margules and Pressey 2000; Molain et al. 2009). Yet, most conservation planning efforts have been exclusively based on contemporary landscape patterns (Rouget et al. 2003). We are only aware of a handful of studies that have (at least partly) considered future land-use trajectories when selecting conservation sites. For example, current and potential land-use threats and future vulnerability values were incorporated to map conflict areas between land transformation and conservation sites at the regional scale (Joshi et al. 2011; Reyers 2004; Wessels et al. 2003) or between land transformation and biodiversity hotspots (Perez-Vega et al. 2012).

Similarly, using the potential extent of future habitat loss owing to land-use and climate changes and global scenarios showed land-use change pressures that drive the endangerment of biodiversity at the global and continental scale (Araújo et al. 2008; Giam et al. 2010; Viscotti et al. 2011). However, linking land-use projections with connectivity analyses for networks of protected areas has rarely been done. Major reasons for the scarcity of studies linking land use modelling, landscape connectivity analysis, and conservation prioritisation is the lack of future land-use maps at scales detailed enough to allow for conservation planning, as well as the lack of tools to assess the fragmentation and connectivity of landscapes in a spatially explicit way.

Relatively simple and robust land-use simulation models are increasingly becoming available for conservationists and land-use planners to explore future pathways of land-use systems (e.g. Clarke and Gaydos 1998; Pontius et al. 2001; Verburg and Overmars 2009; Verburg et al. 2006). Similarly, novel spatially explicit approaches that analyse landscape connectivity have been developed that allow identifying those landscape elements that are of particular importance for habitat connectivity. For instance, morphological image segmentation can stratify habitat patches into core, edge, and corridor habitat. Integrating such spatially explicit fragmentation approaches with graph theory tools (Kindlmann and Burel 2008), which assess the structural connectivity of habitat networks, provides powerful tools for identifying priority sites for conservation (Saura and Torné 2009; Ziółkowska et al. 2012). Such approaches refer to the structural connectivity of landscapes (i.e., the topological relations among natural habitat patches). Landscape structural connectivity is a good proxy for functional connectivity for many species, since highly connected landscapes allow for better dispersal of intermediate-mobility species and, hence, for lower risk of isolated populations (Saura and Rubio 2010). Our goal here was to combine a spatially explicit land-use model and fragmentation analysis in order to assess the effects of future land use on the connectivity of current protected area networks. As a study area, we selected a relatively large region in the southeast of Spain which undergoes rapid land-use changes, with co-occurrence of both land-use expansion and abandonment. Our specific research questions were:

1. How will future land-use patterns develop during the next years in southeast Spain and how will they affect the fragmentation and connectivity of natural habitats?
2. Do current protected areas comprise those landscape elements that are important for maintaining the regional landscape connectivity? Which important gaps should be considered for conservation?

Methods

Study area

Our study focuses on the province of Almería in the southeast of Spain, Andalusia (Fig. 1). This area is located in the Mediterranean basin, which is highly dynamic in terms of land-use changes with close co-occurrence of both loss and recovery of natural vegetation (SOER 2010). On the one hand, the province of Almería has undergone land use expansion over the last decades, especially near the coast, leading to habitat loss due to the expansion of greenhouses and urbanisation (the latter mainly for tourism) (Mota et al. 1996; Peñas et al. 2011). Indeed, the current economy of Almería mainly depends on semi-industrial intensive agriculture (Sánchez-Picón 2005) and tourism (Junta de Andalucía 2009), which are concentrated along the coast. On the other hand, rural exodus from inner regions towards the coast, has led to the abandonment of traditional land-use practices (del Barrio et al. 2010), followed either by the subsequent vegetation recovery (Peñas et al. 2011) or increased soil erosion (Vogiatzakis et al. 2006).

The province is a representative site of the Mediterranean biodiversity hotspot (Médail and Quézel 1999), including 2800 plant species (Blanca et al. 2009), 20% of the Iberian endemic plant genera (Cabello 2002), and 3300 km² of unique Habitats of European Community Interest (Fig. 1, Directive 92/47/CEE). Thanks to its great climatic and relief variability, the region harbours diverse vegetation communities ranging from semi-arid scrublands and steppes in the valleys and coastal hills, over to Mediterranean sclerophyllous and coniferous forests in the mid-mountains, and to alpine grasslands in the higher mountains (up to 2800 m a.s.l) (Valle 2004).

There are currently two protected area networks in force in the study area: (1) the Andalusian network of protected areas, Red de Espacios Naturales Protegidos de Andalucía (RENPA, Decreto 95/2003), consists of 19 sites in Almería and protects 33% of the province surface; and, (2) the European Natura 2000 network contains six additional sites, increasing the protection to 52% of the province. Natura 2000 sites were classified as sites of community importance (SCIs) in 2006 because they contribute to maintain (1) priority habitats or species of community interest, and (2) the coherence of the Natura 2000 network (Directive 92/43/EC, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CONSELEG:1992L0043:20070101:EN:PDF). The SCIs are fully protected in Spain since their approval by the European commission in 2006 and have the same legal status as the RENPA protected areas. Development is only fully prohibited in Spanish National Parks, where there are core zones. In Almería there is only part of a trans-boundary National Park (National Park of Sierra Nevada) with no core areas (zone R). All other protected areas in Almería have a lower protection status (i.e., natural parks, natural reserves, natural monuments and natural places) and land-use changes can thus, to some extent, take place inside them. For the sake of simplicity, we further refer to the Andalusian network of protected areas as the RENPA network, to the six SCIs enlarging the RENPA as Natura 2000 protected areas and to the whole European Natura 2000 network of protected areas (RENPA and SCIs) as the Natura 2000 network (Fig. 2).

Land use data

We used two regional land-use maps from 1991 and 1999 in this work. These maps were produced via photointerpretation of aerial, Landsat Thematic Mapper, and Indian Remote Sensing Satellite
Fig. 1. Study area in the Mediterranean Basin. Eco-regions (numbered polygons) and Priority Habitats of European Community Importance (in grey) are shown for the Almeria province (SE Spain) in Andalusia (south Spain).

Fig. 2. Vegetation changes for the period 1991–2015 with the RENPA and Natura 2000 protected areas (PAs) overlaid.
images at a scale of 1:50,000. Both land use maps have a pixel size of 30 m × 30 m and an overall accuracy of 94.4% (Junta de Andalucía. Consejería de Medio Ambiente 1999). A more recent map (2007) was not suitable for our analysis due to differences in the thematic and spatial resolution. We reclassified the 1991 and 1999 land-use maps into five thematic classes: (1) urban areas; (2) agriculture; (3) natural vegetation; and, (5) freshwater (Table 1). Next, we followed the methodology suggested by Jongman et al. (2006) to stratify land–use patterns by using an eco-region zonation (Fig. 1) based on climate, terrain, potential vegetation, and regional hydrology (Borja Barrera and Montes del Olmo 2008).

### Table 1

<table>
<thead>
<tr>
<th>LU class</th>
<th>LU description</th>
<th>Area change (%) 1991–2005</th>
<th>Area change (%) 1999–2007</th>
<th>Area-adjusted producer accuracy (%) 2005</th>
<th>Area-adjusted user accuracy (%) 2005</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban</td>
<td>Urban nucleus, rural settlements, roads and quarries</td>
<td>1.64</td>
<td>−0.577</td>
<td>18.25</td>
<td>75.00</td>
</tr>
<tr>
<td>Agriculture</td>
<td>Non-irrigated and irrigated arable lands</td>
<td>6.16</td>
<td>−2.166</td>
<td>84.72</td>
<td>63.16</td>
</tr>
<tr>
<td>Vegetation</td>
<td>Natural vegetation</td>
<td>−12.16</td>
<td>4.336</td>
<td>86.07</td>
<td>84.72</td>
</tr>
<tr>
<td>Freshwater</td>
<td>Rivers, water reservoirs and marshes</td>
<td>0.18</td>
<td>−0.095</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Greenhouses</td>
<td>Intensive agricultural</td>
<td>4.18</td>
<td>−1.498</td>
<td>3.74</td>
<td>71.43</td>
</tr>
<tr>
<td>Overall</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>77.22</td>
</tr>
<tr>
<td>Allocation</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>12.00</td>
</tr>
<tr>
<td>Disagreement</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>8.50</td>
</tr>
</tbody>
</table>

**Land use change modelling**

To estimate potential land-use conversions and to produce change maps, we projected future land uses from the transitions occurred between 1991 and 1999 based on Markovian chains and on a cellular automata model (Eastman 2003). Markovian chains were used to derive the conditional probabilities of change by calculating the land-use transitions from one land use (1991) to another (1999) and treating them as a stochastic process with discrete time-steps (Bell 1974). Thus, using the 1991 and the 1999 land-use maps, we calculated transition probabilities matrices and transition area matrices (i.e., amount of pixels changing from one category to another) for the years 2005, 2007 and 2015. The latter two years were chosen according to the 8-year time period between the input maps (1991-1999). The year 2005 was used for an in situ field validation. Thus, our projection can be interpreted as a baseline scenario depicting the ‘Business as Usual’ pathway.

Markov transition probabilities depend on the time interval (T) used to simulate land-use change (NRCS 2000). If T equals the time lag between the input datasets (i.e., for the years 2007 and 2015), transition probabilities are calculated using Eq. (1):

\[ P_{i,j,T} = \frac{n_{i,j}}{\sum n_{i,j}} \]

where \( P_{i,j,T} \) is the transition probability from land use \( i \) to \( j \) in time interval \( T \), and \( n_{i,j} \) is the number of pixels that change from one land use to another.

If \( T \) does not match the interval between the input datasets (i.e., for the year 2005), then transition probabilities are calculated using Eq. (2):

\[ P_{i,j} = \frac{1 - \exp\left[-(1-P_{i,j,T})\right]}{T} \]

All Markovian probabilities matrices were corrected for potential errors (\( P_{i,j}^{\varepsilon} \)) derived from spatial variability in the input land-use maps between the land-use values mapped and the real in situ land-use values, following Eq. (3) (NRCS 2000):

\[ P_{i,j}^{\varepsilon} = \begin{cases} P_{i,j}(1 - \varepsilon) & \text{for } i = j \\ P_{i,j} - P_{i,j}(1 - \varepsilon) & \sum \forall i \neq j P_{i,j} \text{for } i \neq j \text{and } \sum \forall i \neq j P_{i,j} \geq 0 \\ 1 - P_{i,j}(1 - \varepsilon)/(N - 1) & \text{for } i \neq j \text{and } \sum \forall i \neq j P_{i,j} = 0 \end{cases} \]

where \( \varepsilon \) is the overall error, i.e., input land-use maps overall production error, and \( N \) is the number of land–use classes. In our case, \( \varepsilon \) was assumed to be 0.15 taking into account that land-use maps are recommended to have as an average 85% overall accuracy (Foody 2002). Higher production accuracies were reported by the input maps producers but we were skeptical on the validation measures undertaken (which were spatially distributed only along main roads) and preferred to consider lower accuracies as a precaution.

To spatially allocate the amount of land-use change predicted by the Markov chain probabilities, we used a cellular automata (CA) model. The CA model uses the Markov transition areas matrix and the latest input land-use map (1999) to allocate future land uses based on the distribution of local land-use neighbours. In other words, the area of change calculated in the transition matrix is the same as the total area of change spatially allocated by the cellular automata. A 5 × 5 cell moving window was selected to spatially allocate the projected land-use quantities in the surroundings of previous land–use cells. The size of the moving window was defined by the median patch size of the 1999 land-use map (5.928 m²) since the patch size distribution was skewed (Levin and Fox 2004). This resulted in three projected land-use maps for 2005, 2007 and 2015, as well as a vegetation change map from 1991 to 2015 that contained the classes (1) natural vegetation loss, and (2) natural vegetation gain. For the vegetation change map other land-use transitions were not considered. Both, the Markovian chains and the cellular automata model were implemented using the software IDRISI Kilimanjaro (Eastman 2003).

Because our land-use model derived current and future land-use maps, we could validate the current land-use map. To do so, we collected field data in September 2005 where we evaluated 240 random plots of 30 m × 30 m. These plots were stratified based on the eco-regions map (Stehman and Czaplewski 1998). Each plot was labeled according to the five land-use classes (Table 1) and was evaluated as correct when the assigned label matched that of the 2005 projected land-use map. We estimated two measures that inform about different aspects of the model accuracy (Foody 2002): (1) we calculated the confusion matrix, the area-adjusted producer’s and user’s accuracies; and (2) the quantity and allocation disagreement (Pontius et al. 2008; Pontius and Milonos 2011). The producer’s and user’s accuracies were adjusted to the area actually covered by the land cover classes to correct for the bias inherent to stratified sampling procedures (Card 1982; Cochran 1977).
Inter-annual changes of habitat connectivity

To analyse the fragmentation and structural connectivity of natural habitat, we used an approach that combines image segmentation with graph theory (Saura and Rubio 2010). Fragmentation assesses the configuration of individual landscape elements, e.g. the shape of individual patches, and characterises how land-use changes affect habitat topology and configuration (Fahrig 2003). We used morphological spatial pattern analysis (Vogt et al. 2007) to quantify the fragmentation of natural habitats. This approach segments an input image (e.g. the natural vegetation map in our case) into a set of fragmentation components that reveal information about patch size, shape, geometry and connections (Soille and Vogt 2009); core; bridge (i.e., connections among core areas); islet (i.e., small habitat patches without core habitat); edge; and, perforation (i.e., edge habitat inside larger core patches). We used an 8-neighbour rule and a 1-pixel edge width (without distinguishing between internal and external edges, Vogt et al. 2009)). To characterise inter-annual changes in habitat fragmentation, we carried out the same analysis for 1991, 1999, 2007, and 2015.

Structural connectivity refers to the topological connections among natural habitat patches across the entire landscape. Graph theory (Kindlmann and Burel 2008) provides a comprehensive framework to analyse landscape connectivity and to characterise the relative importance of particular patches (i.e., nodes) and corridors (i.e., links) for the regional connectivity of an entire habitat network. Using the fragmentation components maps derived from the morphological spatial pattern analysis, graph models were obtained for 1991, 1999, 2007, and 2015. We used the index of patch probability of connectivity (dPC, Saura and Rubio, 2010), as an estimator of the contribution of each patch to overall landscape connectivity (Baranyi et al. 2011). Patches were grouped into nodes and links depending on their area and relative position in the landscape (i.e., nodes are core areas that may connect to other nodes, while links are usually smaller and connect nodes). For each of our land-use maps, we identified the most important habitat patches for the maintenance of overall structural landscape connectivity. Finally, to evaluate the inter-annual changes of habitat connectivity, we calculated changes in node importance, area of the most important node, number of connectors, and mean distance between connectors.

The role of protected area networks in preserving landscape connectivity

Targeting priority conservation settings for a broad range of users (landscape planners, conservationists, decision makers, park managers, etc.) should consider an ensemble of connectivity areas which would do the most to create a network of natural landscapes rather than a ranked list of areas (Beier et al. 2011). To evaluate the role of protected areas in preserving the most important areas for landscape connectivity in our study area, we integrated the results from the land-use projections and from the landscape connectivity analysis. First, we identified the most important patches for the maintenance of the structural connectivity (patches with highest dPC values) for the period under study. Second, we assessed whether protected area networks (RENPA and Natura 2000) efficiently preserved important areas for connectivity. Finally, we highlighted important areas for landscape connectivity by averaging the contribution to connectivity (dPC value) of each patch over the whole study period and by overlaying the RENPA and Natura 2000 protected areas networks. Landscape connectors with highest dPC values were considered good connectors and important areas for conservation. If these important areas for conservation did not fall inside the protected area networks, they were identified as potential conservation gaps.

Results

Land use projections

The Markov land use transition probability matrix for 2007 showed that around 10% of the area of each land-use class was expected to change from 1999 to 2007 (Table 2). The greatest probabilities of change were observed for the transitions from natural vegetation to agriculture, from freshwater to natural vegetation, from agriculture to natural vegetation, from greenhouses to agriculture, and from urban areas to agriculture. Markov transition probabilities of change for 2005 and 2015 were similar to those of 2007, especially for natural vegetation changes. In terms of area, the rate of expansion in agricultural land increased (100 km² in 1991–1999 and 234 km² in 2007–2015). The same increasing trend with a remarkable acceleration was found for greenhouse developments (55 km² increase in 1991–1999 and 177 km² in 2007–2015) and urban activities (23 km² increase during 1991–1999 and 102 km² during 2007–2015).

According to our projections, between 1991 and 2015, land use expansion throughout the study region will be mainly at the expense of natural vegetation, as agriculture (6.2% increase), greenhouses (4.2% increase), and urban areas (1.6% increase) all expand (Fig. 3). Land abandonment (and subsequent natural vegetation recovery, 32 km²) was not very widespread in our projections and was restricted to a few spots in rural interior mountains or inside protected areas (Fig. 2b). The major areas of agricultural expansion were observed in the interior eco-regions, while the construction of new greenhouses was clustered in coastal eco-regions (Table 3). Urbanisation was also concentrated along the coast (Fig. 3). Accordingly, most of the expected vegetation loss from 1991 to 2015 (1100 km²) occurs in coastal eco-regions (Table 3).

The validation of our land-use projections for 2005 showed good reliability of the land-use maps (overall area-adjusted accuracy ~77.2%; percentage of landscape with quantity disagreement ~8.5%; percentage of landscape with allocation disagreement ~12%). Classification accuracies differed among land-use classes (Table 1). Land-use classes with a large spatial extent (natural vegetation and agriculture) showed higher producer’s accuracies than land uses with lower surface (freshwater, urban areas, and greenhouses, Table 1). The 2005 projection agreed with field observations in 189 points and disagreed in 51 points.

Changes in habitat fragmentation and connectivity

Our results suggest that the landscape of Almería province will become more fragmented from 1991 to 2015. According to the projected land-use changes, the area of core natural vegetation decreases by 6.5% (1077 km²) covering 64% of the province in 2015, edge area increases by 6.1% (92 km²), and bridges (0.18%
change) and islets (0.07% change) remain relatively stable (Fig. 4). The total number and the area of perforation elements decreased (from 6321 in 1991 to 2558 in 2015, and 4.19% in 1991 to 3.67% in 2015, respectively), as perforation elements tend to be converted into edge elements (Table 3). The greatest increase in edge area occurred in mountainous eco-regions (Surco-Bético Oriental, Sierra de Baza-Filabres and Sierra de Gádor) and in the coastal eco-region of Cuencas Litorales Almerienses (Table 3). The decline of islets (Fig. 4) mainly occurred in the coastal areas (Fig. 5b), for example in the Cuencas Litorales Almerienses eco-region where 1% of the region experiences loss of islets which means a 42% disappearance of islets and a loss of 22.6% of total islets. In general, bigger islets

![Land use categories](image)

Fig. 3. Land use maps for 1991, 1999 (both inputs) and 2007 and 2015.

<table>
<thead>
<tr>
<th>Increase in the provincial area (%)</th>
<th>2015 Interior eco-regions</th>
<th>2015 Coastal eco-regions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture expansion</td>
<td>715</td>
<td>12</td>
</tr>
<tr>
<td>Urban constructions</td>
<td>153</td>
<td>10</td>
</tr>
<tr>
<td>Greenhouse developments</td>
<td>385</td>
<td>10</td>
</tr>
<tr>
<td>Vegetation recovery</td>
<td>32</td>
<td>10</td>
</tr>
<tr>
<td>Vegetation loss</td>
<td>1100</td>
<td>23</td>
</tr>
<tr>
<td>Perforations</td>
<td>18.2</td>
<td>23.3</td>
</tr>
<tr>
<td>Perforations to edge</td>
<td>19.6</td>
<td>17.6</td>
</tr>
<tr>
<td>Core to edge</td>
<td>20.7</td>
<td>14.2</td>
</tr>
</tbody>
</table>

Table 3 Relative changes (%) from 1991 to 2015 in the area occupied by each land use and fragmentation element in nine different eco-regions as simulated by a Markovian cellular automaton model.
tend to persist in the landscape (432 islet patches covering 1.25% area in 1991 vs. 288 patches covering 1.65% area in 2015).

As a consequence of the projected increase in landscape fragmentation, structural connectivity decreased in terms of smaller area of the most important connector with increasing connectivity importance through time (Table 4). Accordingly, the number of habitat connectors (nodes and links) lowered through time (Table 4) and distances among connectors increased (Table 4). In addition, our results indicate that the location of the most important nodes to preserve connectivity changes (Fig. 5a). In 1991, the most important node was found in the southern part of the study area (Fig. 5a) while in 1999 the most important node was located in the upper part of the Almanzora basin (Fig. 1) and the relative importance of the southern node decreased. Results for 2007 highlight an important area for connectivity in the lower part of the Almanzora basin. Finally, in 2015 the most important area for connectivity was located in and northwest of Sierra de Bédar (Fig. 5a).

The role of protected area networks in preserving landscape connectivity

Overlaying the RENPA network to the map of landscape connectors (Fig. 5a) showed that this network largely fails to protect the natural vegetation elements (nodes and links) that are most important for enhancing or maintaining landscape connectivity in 1999 and 2007. However, although habitat prioritisation principles of the EU Habitats Directive do not consider the protection of connectivity elements (nodes and links), RENPA protected areas include more than half (68.5%) of the landscape connectors that were found to be important in 1999 (Fig. 6). Although the Natura 2000 network

<table>
<thead>
<tr>
<th>Year</th>
<th>Highest node importance</th>
<th>Area most imp. node (sq km)</th>
<th>No. elements (nodes and links)</th>
<th>Observed mean distance among elements (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1991</td>
<td>0.8</td>
<td>3019.86</td>
<td>1056</td>
<td>871.5</td>
</tr>
<tr>
<td>1999</td>
<td>4.075</td>
<td>182.73</td>
<td>1093</td>
<td>914.11</td>
</tr>
<tr>
<td>2007</td>
<td>6.00</td>
<td>60.85</td>
<td>815</td>
<td>1008.75</td>
</tr>
<tr>
<td>2015</td>
<td>13.04</td>
<td>23.66</td>
<td>480</td>
<td>1506.86</td>
</tr>
</tbody>
</table>

Fig. 5. Increase in habitat fragmentation and spatial shifts in the landscape elements important for connectivity between 1991 and 2015. (a) Change in the structural connectivity elements and (b) change in the location of the most important landscape connector.
would preserve a larger area of the important landscape connectors (1493.2 km² in 2015, Fig. 6) than the RENPA network (677.1 km², Fig. 6), a smaller share of the connector elements will be protected in 2015 by the Natura 2000 Network (36.2% by Natura 2000 vs. 46.7% by RENPA protected areas, Fig. 6). In short, our results show that the protection of landscape connectors (total area and proportion) tend to decline for both the RENPA and the Natura 2000 networks, as the location and extent of natural vegetation elements crucial for overall landscape connectivity change (Fig. 6).

Despite the restrictions regarding land use within protected areas, our model suggests that an additional area representing 19.31% of the natural vegetation is expected to be lost between 1991 and 2015 in the province of Almería, of which 100.54 km² would be located in RENPA protected areas and 104.54 km² in Natura 2000 protected areas. Among the previous 19.31% vegetation lost, a net loss of 480 km² of EU-priority habitats (mainly located near coastal Natura 2000 protected areas) is projected, out of which 108 km² would be located inside the Natura 2000 network. The main vegetation losses in the vicinities of protected areas were projected to be distributed around the first 2 km of the boundaries of Natura 2000 protected areas (29.2% of vegetation lost, Fig. 2a). Yet, the general tendency in our results was towards an increase in vegetation loss with increasing distance to the protected areas boundary (Fig. 2a).

Averaging the importance for connectivity over the whole study period (Fig. 7) revealed that the Natura 2000 network would still preserve more landscape connector area (2739.9 km²) than the RENPA network (1354.9 km²). We detected two regions that could enhance the protected areas’ network connectivity substantially were they protected from intensive land conversion in the future: (1) the region northwest of the Sierra de Cabrera and Bédar protected area; and (2) the region between the Sierra de Cabrera and Bédar and the Sierra Alhamilla protected areas (Fig. 7). The area connecting these two regions contains EU-priority habitats (Fig. 1) and their protection would not necessarily need to restrict future development of the region.

Fig. 6. Comparison of the area of important connectors within protected areas (PAs). In 1991 and 1999 there are no data for the Natura 2000 protected areas because this network was established in 2006.

Fig. 7. Average importance of connectors across the study period 1991–2015. Landscape connectors in dark grey highlight their persistence in time as good landscape connectors. Landscape connectors in black highlight their importance in maintaining structural connectivity. (1) Sierra Alhamilla protected area (PA) and (2) Sierra de Cabrera and Bédar protected area.
Discussion

Our land-use model projects continued vegetation loss in south-east Spain, mainly due to the assumed expansion of agricultural activities, which is in strong agreement with other studies. For example, Muñoz-Rojas et al. (2011) showed that the main land-use changes for the south of Spain were an increase in intensive agriculture between 1956 and 2007. Our results also suggest that the threat of natural vegetation losses is highly clustered throughout the study area and especially widespread in the coastal areas (Cuencas Litorales Almerienses, due to greenhouse and urban developments) and in the interior eco-regions (Surco-Betico Oriental, due to agriculture, Table 3). Overall, natural habitats in Almería continue to be fragmented heavily, resulting in an increasing isolation of the remaining patches (Piquer et al. 2004). Our land-use model was a ‘Business as Usual’ type of scenario and further research should assess the range of alternative land-use scenarios that may occur in Almería province, considering socio-economic drivers (e.g., continued rural exodus, climate change, and policy reforms such as the upcoming CAP reform in 2013. In this context, European-wide future scenarios and land-use maps have a strong potential for conservation which is far from fully exploited; however, they are still too coarse in scale for regional planning.

Fragmentation did not occur uniformly throughout our study region and understanding the spatial variation in fragmentation processes is therefore a major step towards choosing appropriate conservation management actions (Ritters and Coulston 2005). Here, habitat fragmentation was mainly characterised by an increase in edge elements in interior mountainous areas (e.g., due to the creation of holiday houses and sparse greenhouse developments) and a loss of islets in coastal eco-regions (due to massive recreational complexes for tourism and a dense fabric of greenhouses, especially in the surroundings of protected areas, Fig. 5). Both fragmentation and isolation processes have important ecological consequences. Increasing edge length translates into more edge effects (e.g., species invasions, pollution, higher predation) (Murcia 1995), which especially affects species habitat specialists. Losing islets means that stepping stones acting as refugia or facilitating dispersal or genetic flows are lost (Cox and Underwood 2011). This process especially threatens those ecosystems distributed along coastal eco-regions (Fig. 5), such as the endemic arboreal matorral of Ziziphus lotus, a priority habitat under the Habitat Directive (code 5220) close to extinction in Europe due to ongoing greenhouse expansion in the study area (Mota et al. 1996). The relationship between fragmentation, structural connectivity and species extinction risk is not linear but characterised by thresholds (Fahrig 2001 2002 2003; Metzger and Décamp 1997). Determining these thresholds for threatened species, and assessing whether such thresholds would be breached based on future land-use protections, such as those we generated here, are important steps towards sustainable land-use systems that align production and conservation goals.

While the study showed that the Natura 2000 network was relatively effective in safeguarding natural vegetation patches until 2015 inside Natura 2000 protected areas (where some land-use practices are allowed), a main result from the projections is also that overall landscape connectivity of natural habitats could strongly decline. A major reason for this likely to be that conservation planning has so far not adequately considered landscape connectivity and potential changes therein. This is surprising given that preventing habitat fragmentation and the isolation of protected areas in regions with rapid land-use changes is one of the top conservation issues in Europe (Heller and Zavaleta 2009; Pullin et al. 2009) and important in light of accelerating climate change (Heller and Zavaleta 2009). Indeed, the European Habitats Directive takes into account the importance of connecting elements in the landscape (article 10 of the Habitat Directive). However, the definition of conservation status of natural habitats lack guidelines (Mehtalá and Vuorisalo 2007; Velázquez et al. 2010) and priority conservation areas are defined solely based on expert knowledge, without using a systematic approach to evaluate the future risk of habitat loss, often resulting in a patchwork of protected areas (Gurrutxaga et al. 2010; Maiorano et al. 2007).

Increasing protected area isolation in coastal regions is further highlighted by the finding that although most land-use changes in the study area occurred outside protected areas (Fig. 2b), highest vegetation loss rates were projected for the land closest to Natura 2000 protected areas (i.e., within the first 2 km, Fig. 2a). This is in agreement with Piquer et al. (2004) that detected intensive land uses and vegetation losses in the surroundings of coastal protected areas in Almería. One explanation for this is the expansion of the protected area network itself. As more land is designated to reserves, the remaining natural vegetation areas increasingly are under much higher risk of conversion. Ultimately, this calls into question the current protection paradigm on (relatively strictly) protected areas if such a paradigm leads to an increasing isolation of natural habitats. In mountainous areas the situation was slightly different, with vegetation cover staying relatively stable through time. However, land use in the mountains is not economically profitable and therefore rare.

The connectivity analysis also revealed that regional conservation planning has so far not considered the most important landscape connectors for natural habitat networks in southeast Spain (Fig. 5 a), although substantial areas of landscape elements providing a connector function are within the RENPA network of protected areas in Almería (Fig. 6). Furthermore, the Natura 2000 network of protected areas, although covering a much greater area, would include proportionately less important connector elements given the land-use projections (Fig. 7). These results highlight the challenges that conservation planners face, because both current and potential future land use need to be considered when deciding upon conservation measures. This is not easy in regions where land use is dynamic, and where opposite trends, such as land-use expansion (natural vegetation lost, Fig. 2b) and abandonment (natural vegetation gain, Fig. 2b) of land use take place side by side.

The results from the connectivity analyses show interesting dynamics in the central upper part of the province (Almanzora river basin, Fig. 1), where the projections indicate a marked shift in patch importance for connectivity due to land-use change (Fig. 5a). This river basin has experienced one of the most aggressive development trends in South Spain (H. Castro, pers. comm.), because of intensive mining for marble in the west, urban constructions in the centre, and irrigation of extensive cropland expansion in the lower parts of the basin (mining is part of the urban class, see Table 1). The drastic changes in overall landscape connectivity due to these intensive land uses further illustrates the lack of integrated management of the basin where all pressures and outcomes (including overall habitat connectivity) are evaluated when assessing the environmental impact of development. Including feedback mechanisms (Claessens et al. 2009) or implementing multi-objective management measures such as the Ecosystem Based Management approach (Curtin and Prellezo 2010) or cumulative environmental impact assessment, especially in the context of the Habitats Directive’s Appropriate Assessments (Söderman 2009), could be an important step towards integrated landscape management in such cases (Crain et al. 2009).

What are the conservation implications of this study? There exists a vast array of land-use models that predict land-use changes into the future without further application into the planning context. Often, these models do not generate land-use maps at scales fine enough and across areas large enough for conservation planning. Where available, land-use and conservation planning also
does not make full use of these projections. Cost-effective conservation strategies must prioritise places that are of high conservation value but that at the same time are threatened (Brooks 2001). Our study further highlights the value of combining land-use models and conservation planning tools to identify such areas at risk (Reyers 2004; Rouget et al. 2003; Wessels et al. 2003), and to identify dynamics in important hotspots of landscape connectors that would need to be protected to ensure the ecological integrity of the landscape as a whole. Our study showed three main regions that lack protection and that could be important to integrate in the protected area network:

1. The Cuencas Litorales Almerienses (coastal basins of Almeria) eco-region that faces high vegetation loss due to greenhouse and urban development, very high risk of patch (islets) vegetation being lost, and a strong trend towards conversion from perforation to edge elements.

2. The northwestern surroundings of the Sierra de Cabrera and Bédar protected area (Fig. 7) one of the most important connecting elements in our analysis, and where edge habitat increases dramatically by 2015 (Fig 5b); and

3. The region connecting the Sierra de Cabrera and Bédar protected area and Sierra Alhamilla protected area (Fig. 7), a region where land used resulted in much natural vegetation loss and an increase in edge habitat.

For these three areas we urge regional conservation managers and landscape planners to closely investigate the conservation status, fragility and future development options in light of the outstanding importance for the habitat network of Almeria province as a whole. Landscape connecting patches sometimes fall outside protected areas (such as in 1999 and 2007 in Almeria), and elements important for connectivity may change through time. Generally, we therefore recommend revising conservation planning in the Almeria province to (1) reinforce protected area networks (specifical coastal Natura 2000 protected areas for southeast Spain) by cautiously extending the protected area system (see above), (2) to reevaluate the current focus on protected areas alone to expand the scope of conservation planning beyond the boundaries of protected areas (Araújo et al. 2007; Santos et al. 2008), with a special emphasis on overall landscape connectivity, and (3) to consider future landscape connectivity in a holistic manner (Naveh 2000) to avoid social conflicts (Roca et al. 2011) and safeguard the integrity of ecosystems.

In combining a land-use simulation model and connectivity analyses we have shown that even relatively simple tools (cellular automata of Markov chains and morphological spatial pattern analyses), if used jointly, can substantially advance understanding of hotspots of landscape connectivity and how they may change in the future. Such approaches are urgently needed to allow conservation planners to evaluate the effectiveness of protected area networks and future threats to conservation. Preserving landscape connectivity is an increasingly important conservation goal in an ever more dynamic world, and approaches such as the one implemented here for southeast Spain highlight a way forward for conservation planning to deal with the uncertainties of the future.

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