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Land use and biodiversity patterns of the herpetofauna: The role of olive groves

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ABSTRACT

The intensification of agriculture has significant environmental consequences. This intensification entails the simplification and homogenisation of the landscape, which leads to strong negative impacts at ecosystem level, including declines in animal biodiversity. The purpose of this study was to assess the effect of different land uses on reptilian and amphibian biodiversity patterns at a regional scale by analysing a large database on the presence of amphibians and reptiles in Andalusia (southern Spain). GIS techniques and the Ecological-Niche Factor Analysis (ENFA) were applied in order to assess whether the habitat was suitable for each reptilian and amphibian species, when the land use variables were excluded. The incongruence between the potential and the observed species richness was then correlated with the main types of land use in Andalusia. Our results showed that irrigated and unirrigated olive groves were associated with a biodiversity deficit of amphibians and reptiles respectively, whereas natural forests and pastures, along with more heterogeneous crops areas, were more suitable. A clustering analysis showed that generalist species were related to olive groves whereas rare and specialist species were related to land uses related to natural vegetation. In summary, our results indicate that large areas covered by olives groves harbour less amphibian and reptilian diversity, thus suggesting that agro-environmental schemes should be carried to promote the species richness in these crops.

1. Introduction

Agriculture intensification is characterised by an increase in management intervention and external inputs with the intention of enhancing agricultural yield, which includes increases in mechanisation, the removal of natural vegetation, fertilisation, and the wide use of pesticides (Kizos and Koulouri, 2006; Plieninger et al., 2013). As a general rule, agricultural intensification entails the simplification and homogenisation of landscapes, which lead to an overall decline in farmland biodiversity (e.g. McLaughlin and Mineau, 1995; Benton et al., 2003; Medan et al., 2011), thus being considered the factor to have had the largest effect on the loss of biodiversity in agroecosystems (Sala et al., 2000).

In the Mediterranean region, which is considered to be one of the 25 “biodiversity hot spots” in the world (Myers et al., 2000), agricultural intensification has been common since the 1950s (Matson et al., 1997). The olive tree (*Olea europaea* L.) is one of the main crops in this area (Sokos et al., 2013). Half the world's olive production is located in Spain, and the amount of land given over to olive orchards in Spain increased by 300,000 ha between 1996 and 2008 (COI, 2013). This intensive olive-producing agriculture threatens traditional agro-ecosystems such as winter cereals, extensively grazed pastures and low-input olive farming, since the traditional mosaic landscape has been replaced with intensive olive monocultures (Beaufoy, 2001; Stoate et al., 2009). This intensive olive system is characterised by the establishment of younger and more productive varieties with a higher tree density and drip irrigation, along with an increased use of agrochemical products (Palomares et al., 2015), which have lead to a reduction in habitat heterogeneity, higher pollution and soil erosion, which have in turn decreased animal and plant diversity (Santos and Cabral, 2003;

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Siebert, 2004; Metzidakis et al., 2008).

Solutions to the negative impact of farm intensification are complex. Green et al. (2005) proposed two alternatives: wildlife-friendly farming (which boosts densities of wild populations on farmland but may decrease agricultural yields) and land sparing (which minimises the increasing demand for farmland by yield). The authors concluded that high-yield farming may allow more species to persist. However, the conclusions reached by Green et al. (2005) were based on birds since no other taxa permit such a detailed and comprehensive analysis. The only taxa to have been reasonably well studied are birds and mammals, and many management decisions are made on the basis of their ecological needs (Stoate et al., 2009; Robledano et al., 2010).

Reptiles and amphibians are recognised as being extremely sensitive to local habitat changes (Anadón et al., 2006; Castellano and Valone, 2006) owing to their ecological and physiological constraints (such as temperature or water), low dispersal capacity and small home ranges (Huey, 1982), and it is therefore supposed that both groups will be more prone to the risks associated with agricultural intensification than other vertebrate taxa (White et al., 1997). Indeed, Fryday and Thompson, 2012 identified 155 papers related to amphibians in agricultural habitats, but none of them were focused on olive groves (but see García Muñoz et al., 2010a, 2013), and the same is true in the case of reptiles (but see Atauri and Lucio, 2001). The Iberian Peninsula is one of the Mediterranean areas richest with regards to herpetofauna, and is the home to a considerable number of endemic reptilian and amphibian species (Barbosa et al., 2012; Sillero et al., 2014) that play an important role in the trophic web (Martín and López, 2002). The southern areas of the Iberian Peninsula have a higher genetic diversity than those in the north since the area acted as a refuge during last glacial period (Gómez and Lunt, 2007). Despite the vast area occupied by olives groves (1.5 million hectares in Andalusia), little is known about the effect of olive groves on the biodiversity pattern of reptiles and amphibians at large scales.

The principal goal of this study was therefore to evaluate the impact of olive groves on amphibian and reptilian biodiversity in Andalusia. We achieved this objective by employing GIS techniques and specific niche requirements to assess whether the habitat was suitable for each reptile and amphibian species in a UTM-grid of 10 × 10 km through the use of Ecological-Niche Factor Analysis (ENFA). Since most new olive orchards are farmed using an intensive irrigated system, the second objective was to identify the impacts of different olive grove management regimes on amphibian and reptilian communities. We expected that reptile and amphibians biodiversity patterns would be modulated by both climatic and land-use variables, and we hypothesized a herpetological biodiversity deficit in those areas with larger surface covered by olive groves.

2. Materials and methods

2.1. Study area

Andalusia is a large territory in the south of Spain covering 87,268 km^2. It is characterised by a Mediterranean climate, with oceanic features in the western area and arid features in the eastern area. The Andalusia relief is generally orientated in a SW–NE direction, and exceeds 3000 m above sea level in Sierra Nevada. Average annual temperature varies over a wide range, due mainly to altitude, from 9 to 10 °C in mountain enclaves to 18–20 °C in some areas along the Mediterranean coast, while precipitation ranges from a low of 250 mm on the eastern coast of Almeria to 2000 mm in Grazalema (Pita, 2003). Andalusia is the largest olive (*O. europaea* L.) oil producing region in the world, and this production is concentrated in the central-eastern area of Andalusia (Fig. 1). In Spain, olive trees have traditionally been cultivated in non-intensive or low tree-density orchards (around 100 trees/ha^{-1}) with no irrigation and with a ground cover (herbaceous vegetation) that rarely exceeds 25% (Villalobos et al., 2000). Drip irrigation has recently (in the last 30 years) been established in intensive orchards characterised by a high tree-density (300–400 trees ha^{-1}) and in super-intensive orchards characterised by a very high tree-density (400–1700 trees ha^{-1}) (Cameira et al., 2014), in which reduced tillage, high inputs of pesticides and fertilisers and mechanical harvesting are used in order to push up olive yields (Palomares et al., 2015).

2.2. Environmental data

The land use data were compiled using the “Map of land uses and vegetation cover of Andalusia 2007, scale 1: 25.000” (MUCVA, 2007), and were clustered according to study aim (olive groves) and study groups (Appendix 1). The eco-geographical variables (EGVs), which included climatic, geological and topographic variables, were obtained from different sources. The climate data were compiled from a number of databases and are available at the WorldClim website (http://www.worldclim.org/bioclim); altitude data were acquired from the Digital Elevation Model of Andalusia; while geological data were provided by the Spanish Geological and Mining Institute (http://www.igme.es/) (Appendix 2). All these variables were standardised by using ArcMap 9.3 to apply two procedures using: 1) the medium value of each climatic and topographic variable was calculated for each territorial unit; and 2) the percentage of the total area occupied by each type of geological and land use variable within each 10 km square in relation to the total area of the square, thus allowing us to derive an independent variable for each type of EGV. Multicollinearity among these environmental predictors may result in adverse effects in the modelling process, and collinear variables were therefore excluded using the variance inflation factor (VIF), in which 3 was considered to be the threshold cut-off value (Zuur et al., 2010).

The aforementioned objectives were achieved by following five steps: 1) The calculation of Observed Species Richness (OSR), which was carried out separately for amphibians and reptiles in the study area, and 2) GIS techniques and environmental niche modelling to evaluate the suitability of the habitat for each reptilian and amphibian species (excluding the land use variables), which were followed by an estimation of Potential Species Richness (PSR); 3) The calculation of the Subtracted Species Richness (SSR), defined as the difference between OSR and PSR; 4) The influence of land use on the SSR was evaluated using multiple regression; 5) Finally, we assessed how the species are associated to the different land uses by using a cluster analysis.

The data regarding reptilian and amphibian distribution were acquired from the Spanish Vertebrate Atlas (Ministerio de Medio Ambiente, 2014), which contains comprehensive information on the distribution of non-domesticated vertebrate (which includes data from 2009 to 2013). These databases yielded 9668 and 4498 records for reptiles and amphibians, respectively, whose data can be considered as high quality (Martins et al., 2014). With regard to herpetofauna, Andalusia can be considered to have been exhaustively sampled and the associated distribution databases are of a high quality (Martín and Avia, 2011).

The spatial resolution of the study was constrained by the data with the coarsest scale (10 × 10 km squares) – the distributional species data (see Moreno-Rueda and Pizarro, 2007). This resolution is enough to detect change in biodiversity patterns of amphibians and reptiles (Martins et al., 2014). We analysed 24 of 27 autochthonous reptile species and 15 of 16 autochthonous amphibians