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Habitat Fragmentation in Arid Zones: A Case Study of *Linaria nigricans* Under Land Use Changes (SE Spain)

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Abstract Habitat fragmentation due to human activities is one of the most important causes of biodiversity loss. In Mediterranean areas the species have co-evolved with traditional farming, which has recently been replaced for more severe and aggressive practices. We use a methodological approach that enables the evaluation of the impact that agriculture and land use changes have for the conservation of sensitive species. As model species, we selected *Linaria nigricans*, a critically endangered plant from arid and semiarid ecosystems in south-eastern Spain. A chronosequence of the evolution of the suitable habitat for the species over more than 50 years has been reconstructed and several geometrical fragmentation indices have been calculated. A new index called fragmentation cadence (FC) is proposed to quantify the historical evolution of habitat fragmentation regardless of the habitat size. The application of this index has provided objective forecasting of the changes of each remnant population of *L. nigricans.* The results indicate that greenhouses and construction activities (mainly for tourist purposes) exert a strong impact on the populations of this endangered species. The habitat depletion showed peaks that constitute the destruction of 85% of the initial area in only 20 years for some populations of *L. nigricans.* According to the forecast established by the model, a rapid extinction could take place and some populations may disappear as early as the year 2030. Fragmentation-cadence analysis can help identify population units of primary concern for its conservation, by means of the adoption of improved management and regulatory measures.

Keywords Habitat depletion · Conservation · *Linaria nigricans* · Fragmentation cadence · Greenhouses · Human settlements

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Introduction

Land-use changes constitute one of the most important driving forces of biodiversity loss (Sala and others 2000), usually resulting from habitat destruction and habitat fragmentation. In fact, habitat fragmentation is considered the leading cause of species extinction (Pimm and Raven 2000) and the most serious threat to biological diversity (Wilcox and Murphy 1985).

In fragmented landscapes, plant populations become both smaller and more isolated from each other (Oostermeijer and others 2003). Fragmentation problems especially affect rare and/or specialist species, since the probability of having their ecological niche represented in the remaining habitat patches is lower (Henle and others 2004). It should be noted that connectivity of habitat patches is thought to be important for movement of genes, individuals, populations, and species over multiple temporal and spatial scales (Minor and Urban 2008). Therefore, understanding the causes and spatially assessing the different forms of population decline (past, present, and future) is important to design effective management techniques to reduce the rate of biodiversity loss ("Caughley declining-population paradigm"; Norris 2004).

Fragmentation and loss of natural and semi-natural habitats has aroused great interest in recent decades (e.g., Collinge 1996; Harper and others 2007; Kjelland and others 2007; Cousins 2009). A variety of experiments have used the "focal species" approach in conservation programmes aimed at reassembling fragmented landscapes (Melbourne and others 2004). The focal species can be used as a threat-based indicator to provide management guidelines for the reconstruction of fragmented landscapes (Lambeck 1997). Moreover, a variety of indices based on diverse criteria have been developed to objectively quantify the degree of habitat fragmentation (for a review of fragmentation indices, see Jaeger 2000), as well as different measuring techniques across spatial scales (Olff and Mark 2002), but only a few studies have applied these metric based methods to time-series data in order to assess the historical evolution of habitat fragmentation (e.g., Burnside and others 2003; Bogaert and others 2005; Quine and Watts 2009).

Fragmentation due to land-use changes strongly threatens habitat conservation in arid and semi-arid environments of south-eastern Spain. These areas have an exceptional concentration of species together with a high degree of endemicity (Sainz and Moreno 2002; Cabello and others 2003), thus constituting a biodiversity "microhotspot" within the large Mediterranean hotspot (Médail and Quézel 1997). However, these environments have been historically underestimated from an effective conservation viewpoint (Benito and others 2009; Lorite and others 2010). Natural habitats have coevolved over centuries with traditional farming in this area but, recently (since the 1960s), severe land use changes, such as greenhouses spreading or the construction of building complexes for tourist purposes are destroying and fragmenting the remaining habitats, inflicting a severe loss in biodiversity (Benito and Peñas 2008).

In this study, we evaluate the historical evolution of habitat fragmentation in arid zones of south-eastern Spain regarding land use changes. *Linaria nigricans*, a critically endangered endemic plant has been used as a focal species. To achieve this objective, we have developed an evaluation index to monitor the fragmentation rate throughout a chronosequence.

Method

Model Species and Study Area

Linaria nigricans Lange (Scrophulariaceae) is a narrow endemic restricted to four populations in south-eastern Spain: Tabernas, Campohermoso, Salinas and Pulpí (Fig. 1; Table 1). It has been catalogued as "Endangered" (EN) (Cabezudo and others 2005) according to the IUCN criteria and is protected by the Andalusian Regional laws (Anonymous 2003).

Linaria nigricans is a short-lived therophyte (December–April) with a short flowering period (1–3 days per flower), and it has a high seed production (between 82 and 146 seeds per fruit), dispersed by wind. The number of individuals per populations shows great interannual variation related to rainfall (in terms of frequency and intensity) (Peñas and others 2011). The species characterizes diverse and original communities included in the Council Directive 92/43/EEC ("2230 *Malcolmietalia* dune grasslands" and the priority habitat "6120 Xeric sand calcareous grasslands"). These grasslands communities comprise a high plant diversity (Mota and others 2003) with a high number of mesogean taxa (i.e., taxa distributed throughout the Tethys sea during the Cretaceous period, sensu Quézel 1985), such as *Leysera leyseroides*, *Ifloga spicata*, *Ammochloa palaestina*, *Lobularia lybica*, *Filago* spp., etc., and other taxa of narrow biogeographic range, such as *Silene littorea* subsp. *adscendens* (also local endemic), *Asphodelus tenuifolius*, *Eryngium ilicifolium*, *Ononis sicula*, etc.

The characteristic habitat of this plant species consists of flat areas formed by alluvial fans, either of siliceous or volcanic origin, or beach sands in arid and sandy soils (arenosols). The traditional use of the habitat has been extensive farming and livestock, including fallow uses, with low rates of alteration (Benito and others 2009).

The study area is located in south-eastern Spain (Fig. 1) (2.9–1.6° west and 36.6–37.5° north). The climate is Mediterranean arid, with 200–300 mm average rainfall (frequently torrential), a high evapotranspiration rate (UNESCO humidity index, Ih < 0. 3), and average annual temperatures of between 16 and 17°C, with warm and dry summers and temperate winters (Mota and others 2004).

Historical Analysis of Habitat Fragmentation and Threats

The study was structured in two phases. First, an intensive field survey was conducted in 2004 and 2005 and the existing populations of *L. nigricans* were mapped using GPS (Garmin GPSMAP 60 device, 3 m error) and digital orthoimages. Moreover, following IUCN/SSC (2001), qualitative data were recorded, together with bibliographic data, to identify real or potential threats for each plant population.

In a second step, the historical development of the fragmentation and destruction of the suitable habitat for *L. nigricans* was assessed using temporal series of aerial photographs taken in 1957, 1977, 1984–1985, and

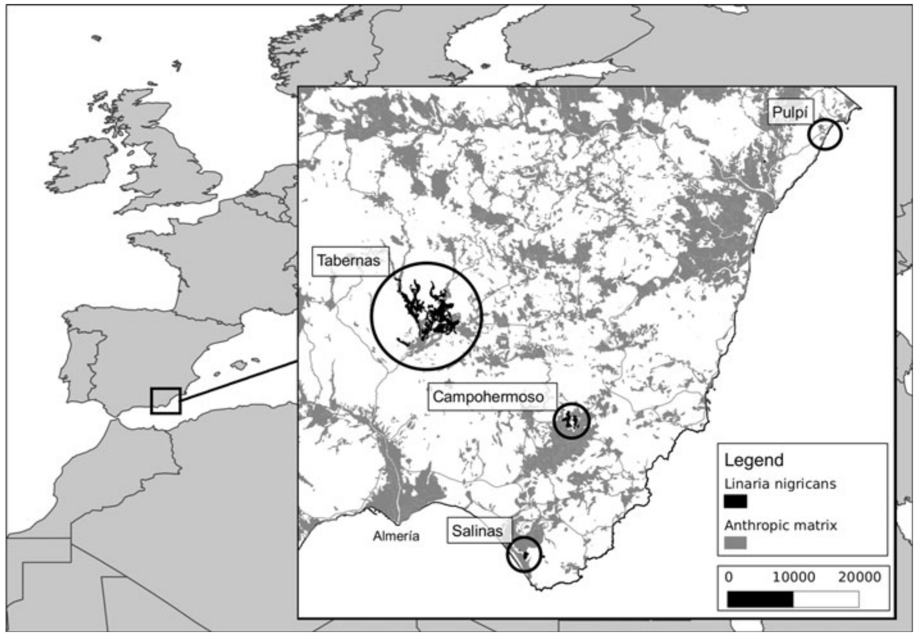

Fig. 1 Study area location in the south-eastern Iberian Peninsula (Spain)

Table 1 Extent (1957 and 2005), number of patches (1957 and 2005) and percentage of habitat lost in the assessed period (1957–2005)

Population	Area 1957 (km^2)	Area 2005 (km^2)	No. of patches 1957	No. of patches 2005	Lost habitat (%)
Salinas	0.629	0.467	2	7	25.6
Pulpí	0.070	0.022	6	7	68.3
Campohermoso	6.443	1.629	8	21	74.7
Tabernas	24.158	18.961	32	64	22.0
Total	31.300	21.079	48	99	33.0

1989–1990, as well as orthoimages 1999–2000, comparing across the time periods each of the four main areas occupied by the species (see Fig. 1; note that there are no aerial photographs available for Salinas before 1977). The aerial photographs were scanned, georeferenced and orthorectified using Idrisi Andes (Clark Labs) to obtain time-series of high-resolution images (1 × 1 m).

Since *L. nigricans* occupies characteristic habitats that are clearly identifiable and mappable, the available habitat patches appropriate for the species (H_n) were identified and delimited with polygons by means of photointerpretation for each time-series of images (chronosequence), considering the following parameters: terrain texture, orientation, slope, and proximity to points with confirmed presence of the species. Likewise, an area of "initial habitat" (H_0) occupied by the plant was established for each population corresponding to an estimated situation before any human fragmentation of the habitat, thus providing a reference to an original situation in order to carry out the fragmentation analysis. In addition, the area occupied by greenhouses and tourist buildings was mapped in all chronosequences in order to analyse the relative effect of the main threats in each population.

The maps of habitat availability were drawn using the software packages FRAGSTATS (McGarigal and others 2002) and Idrisi Andes. The areas of suitable habitat for *L. nigricans* corresponding to each year and the temporal evolution of the fragmentation indices were also calculated. The fragmentation indices, calculated from the remaining patch sizes, were the "splitting index" or SPLI (S) (Jaeger 2000), and a newly proposed index called "fragmentation cadence" (FC; see next section).

The SPLI index (S), calculated according to the formula $S = A_t^2/(A_1^2 + A_2^2 + \cdots + A_i^2)$, indicates the total number of patches of the same size in which the surface H_n should be theoretically divided in order to reach a value of the "degree of coherence" index (C) (Jaeger 2000) equal to H_0. S is independent of the scale, which is useful for comparisons between areas with different sizes, as is the case in our study.

An approximate date of extinction was estimated for each population of *L. nigricans* using the curve-fitting algorithm on the percentages of destroyed area, using the area of H_0 as a reference.

New Index to Estimate Habitat Fragmentation Rate

As a new fragmentation index especially designed to work with chronosequences we propose to use the slope of the index SPLI (*S*), which enables an objective quantification of the historical evolution of habitat fragmentation irrespective of the habitat size. It is calculated using the slope in degrees of the straight line joining the *S* values between consecutive time slices ($FC = \arctan((S_2 - S_1)/(t_2 - t_1))$). This new index was called *FC*. *FC* values higher than zero indicate active fragmentation processes, each *FC* value being higher whenever the fragmentation process is more severe. The graphic representation of the values of this *FC* index along any time-series, allows comparisons of the relative fragmentation speeds among different localities in a graphic way regardless of habitat size, and enables detailed examinations of the variations in the fragmentation rates during the historical period under study.

Results

Fragmentation History of the Habitat of *L. nigricans*

Linaria nigricans has lost a 33% of total habitat, being particularly severe in the populations of Campohermoso (74.7% of the local habitat area) and Pulpi (68.3% idem), whereas the number of patches have doubled (see Table 1). The temporal maps of available habitat of *L. nigricans* show the historical fragmentation processes that have taken place in the different populations (Fig. 2). In the initial situation (H_0), Campohermoso and Salinas showed strong cohesion, while Pulpí and Tabernas showed a geogenic natural fragmentation (i.e., in both cases the habitat has a scattered shape). Except for Salinas, the remaining populations have undergone a notable habitat decrease and habitat fragmentation since 1957.

The analysis using *S* index (Fig. 3) showed differences between population in the fragmentation process. All populations presented very similar *S* initial values (Campohermoso, $S = 2$; Tabernas, $S = 12$), but for Campohermoso an ascending exponential curve was found for the period between 1990 and 2005. Fragmentation analysis using *S* indicated Campohermoso as the most

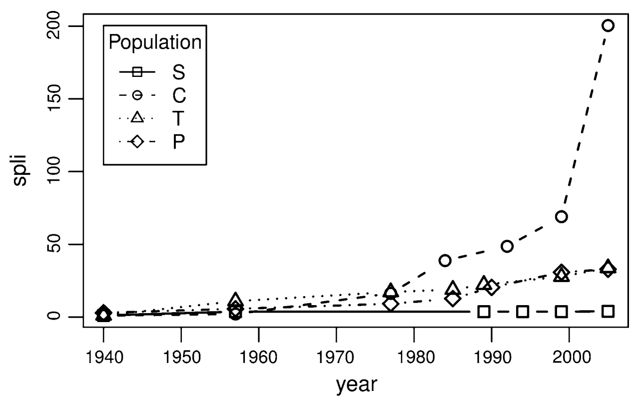

Fig. 3 Splitting index *S* calculated for the four populations of *L. nigricans* populations: S (Salinas), C (Campohermoso), T (Tabernas), P (Pulpí)

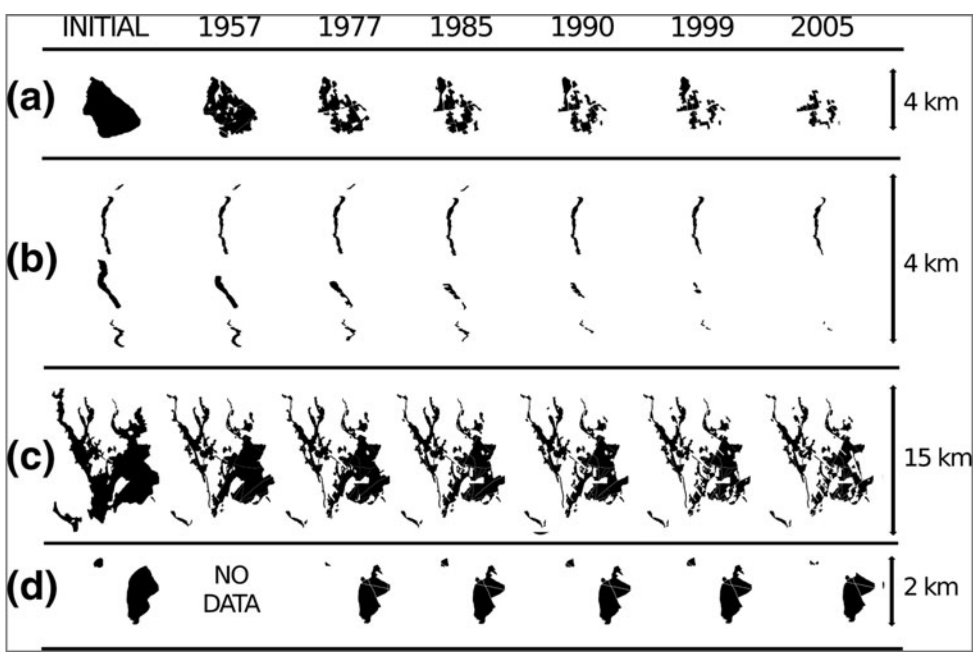

Fig. 2 Chronosequence showing the historical habitat fragmentation in the four populations of *L. nigricans*: (**a**) Campohermoso, (**b**) Pulpí, (**c**) Tabernas, and (**d**) Salinas

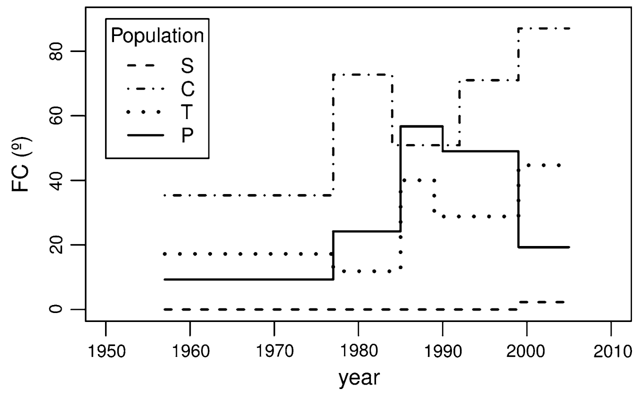

Fig. 4 FC calculated for the four populations of *L. nigricans*: S (Salinas), C (Campohermoso), T (Tabernas), P (Pulpí)

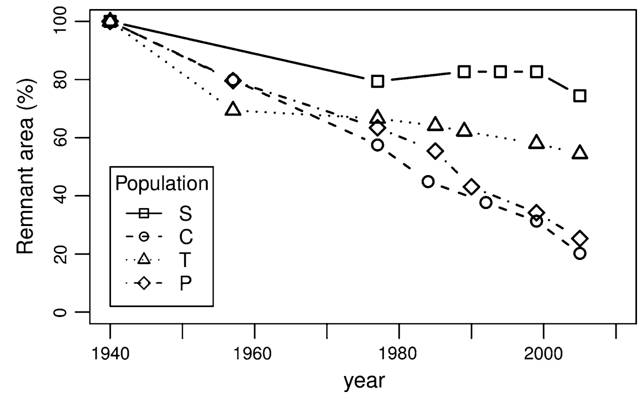

Fig. 5 Historical variation of remaining habitat in the four *L. nigricans* populations studied: S (Salinas), C (Campohermoso), T (Tabernas), P (Pulpí)

severely fragmented population, followed by Pulpí and Tabernas (Fig. 3).

Fragmentation Rates and Loss of Initial Areas

The *FC* index shows the relative speed and the rate of the fragmentation process for each time interval for each population studied (Fig. 4). Campohermoso registered two maximums peaks in the *FC* values (1977–1985 and 1992–2001), both corresponding to periods of increases in agricultural pressure (the second is owed to the spreading of greenhouses; see Fig. 7b). In Pulpí the fragmentation rate was different, showing its maximum during the interval 1985–1992 due to the construction of housing and tourist complexes in the larger patch of habitat occupied by the species (see Fig. 7a). For Tabernas a similar pattern emerged, with a maximum value of the index *FC* during the period 1985–1992 probably due to the abandonment of traditional farming practices, followed by a slight decrease in the fragmentation rate during the next period (1992–1999) and by an increase between 1999 and 2005, in parallel to that observed in Campohermoso. As expected, *FC* values in Salinas did not show any variation.

The comparison of percentage loss of initial habitat available for *L. nigricans* among the different localities (Fig. 5) showed that the populations at Campohermoso and Pulpí lost most of their initial extent, and the trend pointed to their imminent total disappearance. Salinas showed a slight increase in the available habitat during the 1980s and 1990s, and afterwards, a decline (ca. 7%) between 2000 and 2005. Whereas Tabernas lost around 15% of the habitat surface area since the 1950s, although the rate of habitat destruction increased since the 1980s.

Main Threats and Prediction of Extinction Times

The qualitative data dealing with the factors that threaten the species habitat gathered during field and photointerpretation works are shown in Table 2, together with information on whether the populations of *L. nigricans* are included or not in currently protected areas. The main threats observed were abandonment of traditional agricultural management and intensification of modern and more aggressive agricultural practices (e.g., greenhouses spreading), followed by the expansion of human construction. Also the main threat factors that have affected the areas in the past, continue in the present and are likely to persist in the future (by estimation of the trend, see Table 2). Only the population at Salinas is included in a currently legally protected area, while the remaining three populations are not protected at all, despite that there are several threat factors involved.

According to our estimation using curve fitting (Fig. 6; equation: $\mathrm{AREA} = -9793 - 0.0026{*}\mathrm{YEAR}^2 + 10.12{*}\mathrm{YEAR}$; $R^2 = 0.9866$; $F = 258.2$; P-value: 2.765e−07), the habitat available for the species would disappear completely between 2050 and 2060, in the case that the ongoing process of habitat destruction continues. Figure 7 shows the comparison of the habitat destruction and the increase of the area affected by the construction of tourism settlements in Pulpí (Fig. 7a), as well as by the expansion of greenhouses in Campohermoso (Fig. 7b). According to the model, the habitat available for the current *L. nigricans* populations growing at both sites may disappear around 2030.

Discussion

The study shows the severe reduction of habitat available for the narrow endemic *L. nigricans*, thereby constituting a paradigmatic example of how active habitat destruction and fragmentation processes affect biodiversity loss in natural ecosystems of arid zones in Europe (SE Spain), increasing the extinction rates of animal and plant species

Table 2 Summary of main threat factors and legal protection figures, adapted from IUCN/SSC (2001), affecting the populations of *L. nigricans*

Population	Main threat factors	Legal protection figures
Campohermoso	Agriculture: Shifting of traditional management regime (past, present, future). Intensification: greenhouses expansion (past, present, future)	No
	Livestock: Shifting of traditional management regime (past, present, future)	
	Infrastructure development: Human settlements (past, present, future) and greenhouses' auxiliar buildings (past, present, future)	
	Invasive alien species: Competitors (present, future)	
Tabernas	Agriculture: Shifting of traditional management regime (past, present, future). Wood plantations: *Eucalyptus* (past)	No
	Livestock: Shifting of traditional management regime (past, present, future)	
	Extraction: quarries (past, present, future). Groundwater extraction (past, present, future)	
	Infrastructure development: Industry: solar farming (present, future). Human settlements (past, present, future)	
Pulpí	Infrastructure development: Human settlements: touristic resorts (past, present, future)	No
	Accidental mortality: Trampling; vehicles and people (past, present, future)	
Salinas	Agriculture: Shifting of traditional management regime (past, present, future)	Natural Park of Cabo de Gata-Níjar (since 1988)
	Invasive alien species: Competitors (present, future)	Biosphere Reserve (UNESCO)
		Special Area of Conservation (SAC)-European Network Natura 2000

in recent years (Peñas and others 2004; Laiolo and Tella 2006).

Also, we have shown that the processes of habitat destruction and fragmentation in ecosystems of arid zones can be assessed using a chronosequence (spatio-temporal variations), which provides historical hindsight of the fragmentation path operating at the population scale. The possibility of objectively quantifying the impact of any fragmentation process in a population helps to characterize the anthropogenic penetration of landscapes from a geometric standpoint (Jaeger 2000). In highly fragmented habitats the "splitting index" (S) enables the comparison among localities when the fragmentation processes are at

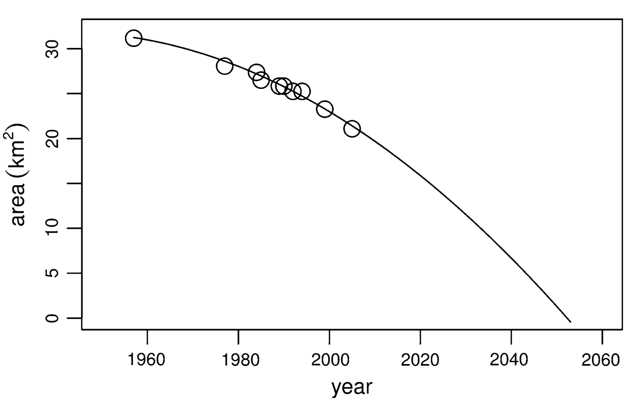

Fig. 6 Habitat decline estimated by curve fitting (considering the four populations)

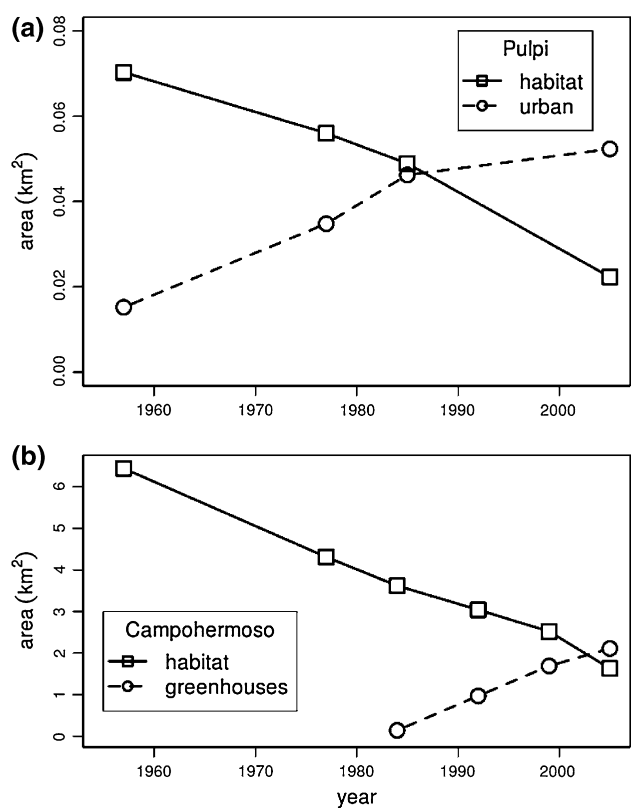

Fig. 7 Habitat depletion in Pulpí (**a**) and Campohermoso (**b**). *Projections* indicate the tendency of habitat loss

an advanced stage, since it has no upper surface limit and is independent of landscape scale (Jaeger 2000). We propose an additional new index (*FC*) that allows the estimation and easy comparison of fragmentation rates among different populations of the same species. It will be useful in cases similar to that of *L. nigricans*, where fragmentation processes taking place in different populations or habitat patches differ in scale, speed, strength, and constancy.

The curve fitting analysis forecasts the total disappearance of habitat available for *L. nigricans* before 2060. Nevertheless, there are two populations (Campohermoso and Pulpí) under imminent extinction risk (by the year 2030) because the small remaining patches are under strong anthropogenic pressure mainly caused by housing and greenhouses expansion. The spread of greenhouses has been established as the main threat against the outstanding biodiversity in SE Spain (Martínez-Fernández and Esteve 2004; Benito and others 2009), frequently involving human occupation of natural habitats declared of European interest (Habitat Directive of the European Union). Both agricultural intensification and technological improvement cause sudden changes in the habitats, forcing species living in them to adapt or disappear (Rodríguez-Teijeiro and others 2009). That is the case of *L. nigricans*, which is threatened not only by the greenhouse expansion and the abandonment of traditional agricultural practices, but also by the construction of new tourist complexes.

The fragmentation rate of *L. nigricans* in the largest population at Tabernas is currently accelerating, since the abandonment of traditional farming practices is leading to a loss of available habitat for the species, since the periodic ploughing seems to favour conditions for this therophyte. Salinas, located within a Natural Park, is the only population with a fairly stable trend. Nevertheless, this lone population may not be adequate to guarantee the persistence of the species in the future, since it represents only 2.18% of the total area occupied by the species. Although there is no risk of land use changes in Salinas, the small population size makes it vulnerable to stochastic natural threats or competition with potential invasive species.

The analysis of habitat-fragmentation processes are crucial and may enable managers and conservationists to make better decisions about reserve design and to make predictions about the consequences of anthropogenic or natural disturbances for a variety of species (Minor and Urban 2008). For fragmented populations, there is a conservation interest in determining the size and the proximity of subpopulations in order to minimize their probability of extinction (Rodríguez and Andrén 1999), which may be used to decide where habitat management will best improve the population persistence (Rodríguez and Delibes 2003).

In this respect it is also important to emphasize that severe habitat-fragmentation processes may have other implications before habitat extinction. Small plant populations may undergo demographic stochastic effects (Oostermeijer and others 2003), be subject to genetic consequences, e.g., inbreeding or genetic drift, (Ellstrand and Elam 1993; Young and others 1996), reduce their fitness (Lienert 2004), or influence the outcome of plant-animal interactions (Kolb 2008). In some cases, as that of the population of *L. nigricans* growing in Pulpí, these effects would be aggravated by the marked edge effect of patches with a linear morphology. Furthermore, small plant populations tend to reduce their fitness since their seeds (wind-dispersed in *L. nigricans*) have a high probability of falling outside the appropriate habitat (Salisbury 1976). The loss of small habitat patches in *L. nigricans* may affect dispersion of propagules, eroding connectivity and genetic flow among populations and even within them among subpopulational units (Peñas and others unpublished results). In this regard, there is also a need to study the specific effects of habitat fragmentation on the population dynamics of *L. nigricans*, since only a few studies deal with population viability in fragmented scenarios for short-living species (Menges 1990; Nantel and others 1996).

The current situation of the habitat available for *L. nigricans* (heading towards its total disappearance according to our data) indicates the need to take urgent action for its conservation. Gómez-Campo and Herranz-Sanz (1993) suggested that small reserves are the best way to protect plant-rich spots in the Iberian Peninsula. Cowling and others (2003) have highlighted the value of the small reserves in the Cape region (South Africa), since they are capable of hosting plant populations as well as important ecological processes. The conservation of *L. nigricans*, its habitat and the evolutionary and ecological processes involved could also be approached in this way: at least a reserve should be designed for each population, since well-managed small natural reserves have been shown to be efficient for plant conservation (also efficient for invertebrate and small vertebrate preservation; Lesica and Allendorf 1992; Laguna and others 2004).

As a conclusion we remark that it is important to stress that the environmental legislation developed to protect species and their natural habitats in SE Spain is not effective for preventing habitat depletion. This situation causes the destruction of the natural resources that prevails over the sustainable development of these fragile arid ecosystems.

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