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REVIEW AND SYNTHESIS

An ecological perspective for analysing rural depopulation and abandonment

Francisco Lloret[1,2] | Adrián Escudero[3] | Joan Lloret[4,5] | Fernando Valladares[6]

[1]Ecology Unit, Department of Biologia Animal, Biologia Vegetal i Ecologia, Edifici C, Universitat Autònoma Barcelona (UAB), Barcelona, Spain; [2]CREAF, Edifici C, Universitat Autònoma Barcelona (UAB), Barcelona, Spain; [3]Biodiversity and Conservation Area, Universidad Rey Juan Carlos, Móstoles, Spain; [4]Instituto de Geociencias, CSIC-UCM, Madrid, Spain; [5]Departamento de Geodinámica, Estratigrafía y Paleontología, Facultad de Ciencias Geológicas, Universidad Complutense, Madrid, Spain and [6]Departamento Biogeografía y Cambio Global, Museo Nacional de Ciencias Naturales, (MNCN-CSIC), Madrid, Spain

Correspondence
Francisco Lloret
Email: francisco.lloret@uab.cat

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Abstract

1. Population loss in rural areas is rapidly increasing in high-income countries, raising concerns and debate, given its socio-economic consequences. Despite the evident environmental dimension of the phenomenon, ecological knowledge has been neglected in the analysis of actions aiming to reverse rural depopulation. Particularly, cultural landscapes reflect memories of ecological processes that have configured current patterns of biodiversity and ecosystem services.
2. Based on ecological principles, we present a conceptual procedure to assess the behaviour of social–ecological systems subjected to depopulation, projecting their expected trajectories through time within a framework defined jointly by demographic and environmental–ecological dimensions (in our case, biodiversity, carbon storage, pollution control, water resources and soil conservation).
3. We applied this procedure to various alternative interventions designed to confront depopulation in Spain: (1) non-intervention, (2) maintenance of the historical landscape configuration, (3) active conservation, (4) extensive, sustainable land use and (5) intensified land use.
4. We conclude that extensive, sustainable land use better optimizes criteria of demographic consolidation, environmental impact, resilience and implementation of actions confronting depopulation. We highlight the need to incorporate ecological knowledge into the assessment and application of actions confronting rural depopulation.

KEYWORDS
biodiversity, depopulation, ecosystem services, fireworks graphs, Spanish rural areas

1 | INTRODUCTION

The rural exodus and abandonment registered in many countries, with special intensity in high-income countries (Hospers & Reverda, 2015), has given rise to social concerns and strong tensions, due to the important economic, political, cultural, demographic and environmental consequences (Hospers & Reverda, 2015; Laborda Soriano et al., 2021). Reversing the depopulation of rural territories is a critical issue on political agendas, and it is a policy priority for the EU (ESPON, 2017). However, the economic, logistical and social incentives mobilized to stimulate the repopulation of these territories have not been successful so far (Navarro-Valverde et al., 2021), and, moreover, they may even exacerbate socio-economic and environmental problems (Hermoso et al., 2022). Significantly, these initiatives have been mainly aimed at promoting economic and social development, while neglecting the ecological dimension (Bruno et al., 2021; MacDonald et al., 2000; Martínez-Abrain et al., 2020; Regos et al., 2016), despite the current dramatic context of climatic crisis and biodiversity loss (IPBES, 2019; IPCC, 2021). In fact, ecological and social dimensions should not be separated when assessing rural depopulation (Hospers & Reverda, 2015). For instance, both social and ecological systems operate at spatial and temporal scales that go beyond the territory experiencing population decline. Actions to overcome the urban–rural dichotomy will benefit from considering depopulation and rural abandonment from an ecological perspective, reinforcing the conception of a common system in which rural and urban subsystems mutually nourish each other (Davoudi & Stead, 2002; Tacoli, 1998).

Ecological memory is essential to understand the ongoing ecosystem transformations in areas experiencing massive depopulation. This memory corresponds to the degree to which ecological processes are shaped by its history (Peterson, 2002) and, in our case, they are determined to a great extent by previous human activities (Balaguer et al., 2014) that have configured current landscapes, which, therefore, can be considered as cultural landscapes (Farina, 2000) (Figure 1). In these cultural landscapes subject to intervention, crops and human-driven open habitats prevail to the detriment of extended forests (Sabatini et al., 2017)—whose remnants have also largely been simplified in terms of their structure and biological composition (Lloret, Solé, et al., 2009)—and other habitats such as wetlands and natural grasslands. The historical legacy of human societies is particularly evident in European landscapes, especially in the Mediterranean region (Blondel et al., 2010). However, a 'primeval' situation of naturalness is frequently used as a reference value for range management and restoration (Aronson et al., 1993). But, the primeval ecosystems that were set in these landscapes cannot be recognized exactly in current landscapes, and any faithful restitution of non-modified ecosystems is elusive (Balaguer et al., 2014). Nevertheless, ecological processes that maintain biodiversity, primary productivity and carbon fixation, soil maintenance and the regulation of water and nutrient flows keep operating in these cultural landscapes, which, in fact, provide many of these essential services at an appreciable extent (Bernues et al., 2014; Gómez-Baggethun et al., 2011).

The history of cultural landscapes has determined the complex relationship between humans and biodiversity. A complex balance appears between, on the one hand, extinctions, impoverishment of food webs and disruption of some ecosystem functions (Root-Bernstein & Ladle, 2018). On the other hand, there is a maintenance of diverse and rich semi-natural habitats (Blondel & Aronson, 1999; Carrión et al., 2010), some of them with difficult access, but also in non-remote lands where low intensive agricultural and pasture uses, such as in 'dehesas' —that is, cultural grazed savanna-like ecosystems, common in the western Iberian Peninsula—(Ramírez-Hernández et al., 2014) are established. In these habitats, selective and evolutionary processes have operated in many lineages, thus permitting the persistence of many species and intricate biological interactions (Talle et al., 2016; Figure 1). These agroecosystems not subjected to intensification perform a set processes functionally mimicking natural ecosystems (Power, 2010), although a replacement of natural complex food webs controlled by large vertebrates or water and nutrient cycles cannot be achieved. For example, pressure from livestock has replaced at some extent wild herbivory selective forces, while extensive cultivation has substituted natural habitats affected by recurrent wildfires and herbivore foraging (Navarro et al., 2015). In consequence, a relevant part of the diversity of these cultural landscapes is confined to semi-natural habitats that require active human presence and active management (i.e., grazing, extensive agriculture; Estrada-Carmona et al., 2022; Maurer et al., 2022). When crops, grasslands and forests cease to be exploited, well-known ecological processes operate, largely resembling secondary succession (Escribano et al., 2014). This transformation of vegetation leads to new thickets, woodlands and forests, while open habitats such as grasslands and old fields diminish. A balance is established between the advent of conditions favouring biodiversity that is unable to life in human-driven habitats and the loss of conditions for biodiversity directly associated with these habitats. So, some species may benefit from the derived increase in semi-natural shrublands and forests following human depopulation (Escribano et al., 2014; Estrada-Carmona et al., 2022). This is the case of some components of biodiversity that have been

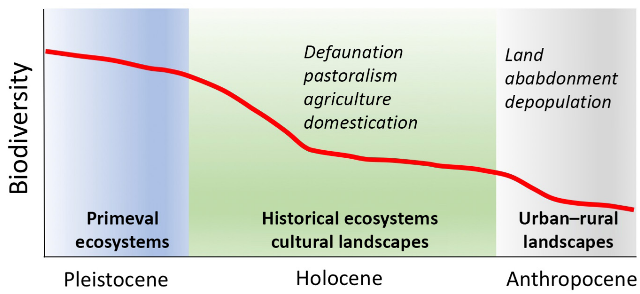

FIGURE 1 Biodiversity decay associated with human activity occurring during the last millennia in cultural landscapes of Europe that have originated novel historical ecosystems and species assemblages in territories currently experiencing depopulation.

largely decimated (Martínez-Abrain et al., 2020) or can hardly survive in regularly altered landscapes, but may find an opportunity to recover when human intervention is limited (Hirt et al., 2021). In contrast, rural abandonment may bring plants and animals typical from agriculture and steppe-like habitats such as some specialized bird species (Traba & Morales, 2019), to the brink of extinction (Moreno Saiz et al., 2019). Moreover, legacy effects of past anthropic use often remain through time in depopulated areas; for instance, invasive species introduced by humans (Guillerme et al., 2020) and the absence of now-extinct key megaherbivores may impede the recovery of biodiversity assemblages, which would need top-down trophic regulations (Svenning et al., 2019).

Here, our main aim is to address rural depopulation from an ecological viewpoint, assessing the effect of different management alternatives that potentially induce 'depopulation reversal', while simultaneously considering their impacts on biodiversity and various ecosystem services. To do this, after establishing different alternatives of intervention in rural depopulated areas, we present a simple conceptual procedure to assess, on the one hand, changes in human demography and, on the other hand, loss of biodiversity and ecosystem services. This procedure aims to provide a roadmap for the assessment of demographic-environmental systems. We qualitatively exemplify our assessment in Spanish territories suffering drastic rural depopulation, mainly in mountain areas (ca. 700–2000 m AMSL; https://atlasau.mitma.gob.es/#c=indicator&i=pobvar.pobvar010&view=map4, accessed 8 June 2023, Atlas Digital de las Zonas Urbanas, Ministerio de Transportes, Movilidad y Agenda Urbana), where a lot of studies are addressing the topic (e.g. Caceres-Feria et al., 2021; Larraz & San-Martin, 2021; Regos et al., 2016; Rodrigez-Rodriguez et al., 2021) and the phenomenon is gaining space on the political agenda. Specifically, our objectives are to (i) build an operational framework for assessing massive rural depopulation in a given territory incorporating an ecological perspective; (ii) establish a methodology to evaluate different intervention alternatives; (iii) analyse the consequences of some of these alternatives on the interface defined by human population (demographic) and different environmental dimensions; and (iv) provide recommendations for intervention and planning in such depopulated territories.

2 | METHODS

2.1 | Assessing alternative interventions to confront rural depopulation

Our procedure is based on the analysis of temporal trajectories of a social–ecological system under different management alternatives in a two-dimensional space defined by (i) a human demographic dimension and (ii) an environmental dimension characterizing ecosystems in such depopulated territory. Here, the demographic dimension considers two variables: mean population density and the inequality Gini index of the number of people in inhabited nuclei (farmhouses, hamlets, villages and towns), which informs on the spatial patterns of human occupancy in the territory. Thus, a trend towards high inequality in the size of populations in the inhabited nuclei of the depopulated territory under consideration would indicate an increase in the people living in cities or mid-size towns, coupled with a decrease in non-urban land. In turn, the environmental dimension is adequately addressed by ecosystem services, when considering social–ecological systems.

The procedural steps are as follows:

1. An evaluation of historical changes in the selected demographic parameters in the depopulated territory, in our case for the ca. 1900–2000 period. Next, the evolution of these demographic parameters is projected from the present to the future (until ca. 2100 in our case, a period which approximately encompasses three human generations) in response to different alternative interventions to confront depopulation (see below section '2.3. Alternative interventions to confronting rural depopulation' for the analysed Spanish case; Figure 2).
2. An evaluation of the trends of critical environmental parameters under the above-mentioned alternatives, considering the same time frame used to estimate the evolution of demographic parameters (see above).
3. A projection of the temporal trajectory of the social–ecological system into the future for each of the alternatives. This trajectory

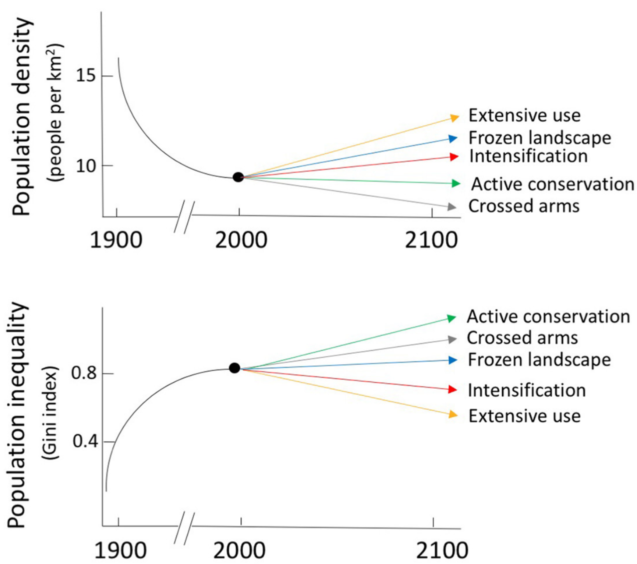

FIGURE 2 Demographic trends (density and inequality estimated by the Gini index of inhabited nuclei) in Spanish mountainous areas subject to rural depopulation. The Gini index is a measure of the inequality of the population distribution in the territory, with values ranging from 0 to 1, with 0 indicating perfect equality—all nuclei have the same population—and 1 maximal inequality, with a single nucleus houses the entire population. Different vectors represent expected future trends under different alternatives for confronting rural depopulation, from now (dot) to the end of the 21st century, following changes in the recent past (black line) (see main text for a description of the alternatives). Source for historical records: Goerlich et al., 2015.

corresponds to a vector whose components are the variation in the demographic parameters (e.g. population density, Gini index of population in inhabited nuclei) and the change in the environmental dimension (e.g. biodiversity, carbon storage, pollution, water resources and soil erosion) over a given time. In the case of the demographic parameter, we consider the difference between a future reference date (e.g. the year 2100 in our case) and the present. The component of the trajectory vector corresponding to the environmental dimension is an estimate of the effect that each alternative produces on the respective environmental dimension for the same period. This estimate should ideally be obtained from quantitative models, but, as in our case, it would often correspond to semi-quantitative trends supported by empirical studies undertaken in the region, or in areas with similar environmental and socio-economic contexts. To build the trajectories, it is essential to establish the territorial range in which the impacts of the proposed alternatives and the depopulation process occur. The result is a set of vectors that represent the foreseeable simultaneous change in demographic and environmental parameters. Here, we propose to name the resulting representation as 'fireworks graph', and it allows us to compare the vectors that describe the effects of the different alternatives (Figure 3).

2.2 | Implementation of the demographic–environmental space: The Spanish case

We illustrate the application of the proposed procedure for the assessment of rural depopulation in Spain, based in the published literature; so, the study does not directly involve human data or participants. We built the fireworks graphs as detailed above after considering published results addressing the relationship between depopulation and ecosystem functioning (related to ecosystem services) and biodiversity in inland and mountainous rural areas of Spain. For this purpose, we searched the literature in the Web of Science database using the keywords string 'depopulation AND (biodiversity OR carbon OR pollution OR water OR soil OR erosion OR ecosystem service) AND Spain' for title, abstract and keywords, and we incorporated additional publications from the author's records (see Data sources).

We considered two demographic parameters: mean population density and the inequality Gini index of the number of people in inhabited nuclei. In the Spanish case, the population density was low in the second half of the 19th century and early 20th century in many rural areas, with a slight-to-moderate tendency to increase, in contrast with a steeper increase in urban localities. Later on, rural areas experienced a slow population decline (e.g. Infante-Amate et al., 2016), which accelerated in the 1950s and, more particularly, the 1970–1980s (source: 'Instituto Nacional de Estadística' of Spain), a trend maintained to the present (Franch Auladell et al., 2013). Then, we qualitatively projected the trends of the trajectories of the two demographic parameters considered for the five proposed alternatives (see below section '2.3. Alternative interventions to confronting rural depopulation') until 2100.

We considered the following environmental surrogates for our assessment: biodiversity, above-ground carbon storage, pollution, water resources and soil conservation. They are linked to well-recognized supportive (biodiversity, soil conservation and absence of pollution), regulatory (carbon storage and water resources) and provisioning ecosystem services (water resources).

The selected environmental surrogates are deliberately wide. For instance, we consider biodiversity in a broad sense, without distinguishing its organizational levels—genetic, species, habitats and ecosystems—and also including biotic interactions and species functionality. This broad estimation of these environmental dimensions allowed us an illustration of the procedure without quantitative analyses, which would require modelling of the future trajectories of demographic and environmental parameters for a given territory, in accordance with the different alternative to face rural depopulation. Although a relationship between some of these dimensions is to be expected—for example, biodiversity is likely affected by the availability of water resources, soil conservation and pollution—we tried to avoid any highly covarying environmental dimension, such as invasions by exotic species—which are closely linked to biodiversity—or fire management—which is closely related to carbon storage.

2.3 | Alternative interventions to confronting rural depopulation

We consider five broad management approaches to facing depopulation and evaluate their impact on demographic and ecological dimensions. These approaches are assessed in virtual territories broadly corresponding to a county (ca. 500–1000 km^2) in mountainous areas of Spain (ca. 700–2000 m AMSL). They are not applied to specific locations because our aim is to provide a general picture of the phenomenon by integrating the available information for the region. These five alternatives do not cover the entire range of management options, but they represent contrasting and realistic strategies of intervention for depopulated territories. Note also that they are not exclusive and can be applied to different areas in the same territory. Furthermore, specific actions may correspond to more than one alternative.

2.3.1 | Alternative 1: Crossed-arms

This alternative corresponds to non-intervention, that is, it maintains the current actions oriented towards population, economic activity and biodiversity conservation and environmental quality. The capacity of the social–ecological system to organize itself in the absence of planned intervention is implicitly recognized. This alternative implies the absence of a sound strategy for the promotion of human populations, which are expected to maintain the current trend of abandonment (Figure 2). So, secondary succession associated with

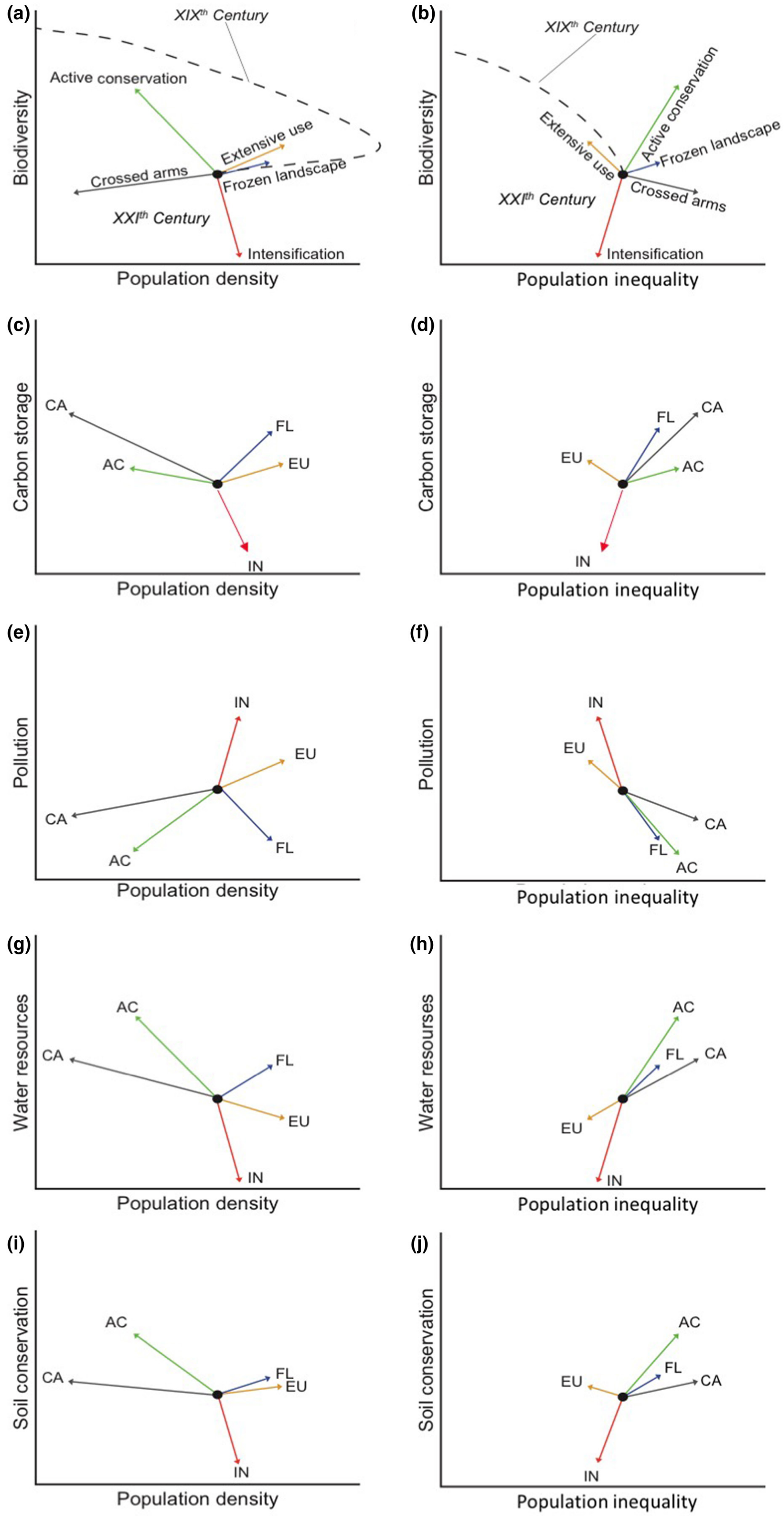

FIGURE 3 Expected temporal (21st-century) trends in taxonomic biodiversity (a, b), carbon storage (c, d), pollution (e, f), water resources (g, h), and soil conservation (i, j), as well as two demographic parameters—population density (a, c, e, g, i) and inequality of population size in inhabited nuclei (b, d, e, h, j)—under different alternatives for confronting rural depopulation (AC, active conservation; CA, crossed-arms; EU, extensive-use; FL, frozen landscapes; IN, intensified-use). Dashed line represents human population demographic trajectories in the recent past in the corresponding biplot.

the abandonment of crops, pastures and forests constitutes the dominant ecological process. This implies a predominance of encroachment and afforestation, which lead to the shrinking of open and semi-natural habitats such as pastures and *dehesas* and their associated biodiversity (Ramírez-Hernández et al., 2014), but, on the other hand, the appearance of habitat for some animals (f.e. birds, Regos et al., 2016), and an increase in C sequestration from the atmosphere (Velazquez et al., 2022). This alternative does not require any great additional financial or infrastructure investments, and economic activity is also expected to maintain current declines.

2.3.2 | Alternative 2: Frozen landscape

This alternative seeks the maintenance of the historical landscape configuration, with well-conserved remnants of natural habitats and mosaics of croplands and pastures. In Europe, this alternative is exemplified by the actions supported by the European Habitats Directive (1992), which provides a framework for biodiversity conservation by recognizing the value of historical habitats, many of them semi-natural and without trees. This alternative adopts a static view of ecological systems, in which vegetation units become references overlooking the role of secondary succession and other constituents of biodiversity and ecosystem dynamics. This static perspective has been questioned, since it does not consider population dynamics and community assembly processes, particularly responding to climate change (Normand et al., 2007). Actions that would potentially revert the trend towards depopulation are subordinated to inputs into the territory aimed at maintaining its spatial configuration, since historical landscapes depend on human activities and management (Figure 2). Promotion of economic activity derives from management plans for habitats, from agro-environmental measures implemented in cultivated areas to reconcile production, sustainability and the conservation of such habitats (Halada et al., 2011), and from ecotourism initiatives, which take advantage of traditional uses and the conservation of valuable semi-natural habitats.

2.3.3 | Alternative 3: Active conservation

This alternative is based on the premise that rural abandonment is an opportunity for biodiversity conservation and the recovery of healthy ecosystems. Therefore, actions promoting biodiversity and ecosystem health—for example, measures for the maintenance, restoration and recreation of habitats; species and habitat monitoring; reintroduction, translocation and conservation management of species populations; control of invasive species (Kati et al., 2015)—are enhanced, particularly in protected areas, such as the Natura 2000 network in Europe. Actions aiming to revert depopulation are accessory, given the secondary role of the human population for biodiversity maintenance, as it does not need to remain in the territory and may even be encouraged to abandon it. Actions derived from active conservation could favour the maintenance of even an increase of local human population (Schou et al., 2021), and they are expected to produce benefits to human societies as well, for instance by preventing diseases; but some evidence suggests negative effects on local populations in strongly protected areas (Rodrigez-Rodriguez et al., 2021; Schou et al., 2021; Figure 2). In this alternative, economic activity is minimized and related to biodiversity-focussed revenues, including ecotourism.

2.3.4 | Alternative 4: Extensive use of the territory

This alternative foments non-intensive land use. Biodiversity and environmental quality therefore benefit on a secondary level, although the maintenance of anthropogenic activity in large parts of a territory implies a certain environmental impact. This alternative likely helps stop or even reverse rural depopulation (Caceres-Feria et al., 2021; Figure 2), but it requires precise planning and a significant investment in public services and infrastructures for communication, while still valuing sustainability. The technological improvements achieved in telecommunications, together with the rise in teleworking, can make it easier to retain and attract people. The portfolio of economic activities for this alternative is significantly represented by ecological and regenerative agriculture—for example, pastoralism and sustainable agriculture based on the recovery of local plants' genetic resources and the maintenance of a heterogeneous agriculture-forest mosaic (Marull et al., 2014)—and other productive, sustainable activities with likely low environmental impacts, such as ecotourism, in combination with outdoor activities and others associated with the conservation of biodiversity and the improvement of ecosystem services (Caceres-Feria et al., 2021).

2.3.5 | Alternative 5: Intensified use of the territory

Rural depopulation provides an opportunity for activities that are not easily compatible with high population densities. This intensification alternative includes the installation of large power generation complexes, polluting industries, urban disposal, and industrial and mineral waste facilities, as well as the construction of transport infrastructure and reservoirs. Intensified land use may also be derived from large recreational resorts (Martin et al., 2018), large monospecific tree plantations, and the intensification of agriculture and farming (Gaglio et al., 2020), via, for example, large-scale irrigation projects. One of the main arguments in favour of this alternative is that economic activities with a high financial return eventually have a positive economic impact on local populations, while other arguments refer to global needs such as the mitigation of climate change. However, local environmental priorities are obscured, the effects on the ecosystems tend to be neglected, and the objectives and actions for biodiversity conservation are disregarded. Depopulation reversal dependent on economic activity would occur without specific actions. So, this alternative can help curb depopulation to a certain extent (Figure 2), but this effect is often only local and short-term (Collantes & Pinilla, 2004)—for example, while infrastructures are being built. The effect may disappear in subsequent phases of exploitation, when the demand for local workers declines and the remaining economic activity may not be sufficient to compensate for

TABLE 1 Qualitative assessment of alternatives for confronting rural depopulation according to several criteria: (i) effects on biodiversity and environmental quality, depopulation reversal and economic revenues, (ii) feasibility of implementation, in terms of economic investment, governance and set-up time, and (iii) capacity to promote the resilience of the social-ecological system (see text for further details). Colours indicate favourable (green), intermediate (yellow), or unfavourable (red) appraisal.

Alternatives to confront rural depopulation	Effect			Implementation			Resilience
	Biodiversity environmental quality	Depopulation reversal	Economic revenue	Economic investment	Governance	Time to set-up	
Crossed arms (CA)	Subordinated to conserved landscape	Business as usual	Business as usual	Low	Easy	Fast	Low
Frozen landscape (FL)	Static perspective	Subordinated to conserved landscape	Environmentally friendly	Medium	Medium	Medium	Medium
Active conservation (AC)	Strong prioritization	Accessory to conservation	Minimized, conservation focused	High	Difficult	Slow	Medium
Extensive use (EU)	Subsidiary of land use	Promoted to maintain land use	Sustainability focused	Medium	Medium	Medium	High
Intensified use (IN)	Disregarded	Dependent on economic activity	Strong prioritization	High	Difficult	Medium	Low

the depopulation associated with the loss of agricultural land (Larraz & San-Martin, 2021), while economic revenues from other activities, such as tourism, could increase (Silvestre & Clar, 2010). In this alternative, economic revenues become the priority, favouring a broad range of potential activities, as described above. Overall, intensification tends to consume large amounts of resources and land, and often pollutes watersheds, air and soils (Ladrera et al., 2019).

2.4 | Assessment of the alternatives for confronting rural depopulation

We have assessed the proposed five alternatives according to these criteria: (i) feasibility of the implementation of the alternative, which can, in turn, be assessed according to the economic investment needed (e.g. cost of health service in territories with low density), the estimated economic returns (e.g. revenues from irrigation plans or energy infrastructures), the governance required for implementation (e.g. approval of specific regulations) and the expected time needed to set all these up, (ii) effectiveness in reversing depopulation (i.e. demographic consolidation), (iii) environmental impact on biodiversity and ecosystem services, (iv) resilience, considered as the system's ability to absorb the impact of disturbances and stressors while maintaining its functionality (Folke et al., 2010; Walker et al., 2004). The assessments of the reversal of depopulation and the environmental impact are derived from the patterns observed in our fireworks graphs, as discussed above, while economic revenue, implementation feasibility, and resilience are estimated from the literature (see Data sources) as shown in the Results for each biodiversity and environmental dimension (Table 1).

3 | RESULTS

3.1 | Biodiversity

The alternative with the most positive effect on biodiversity in the Iberian Peninsula is expected to be active conservation (Figure 3a), given the long history of overgrazing, expansion of crops and forest thinning that have had a negative effect on some key constituents of biodiversity, such as large mammals (Martínez-Abrain et al., 2020). Overall, a general trend of biodiversity loss is likely to be found with increasing human population density (Underwood et al., 2009). When combining biodiversity and human demographic trends, this alternative is likely to lead to a certain demographic decline since it does not involve active social policies aimed at reversing depopulation. This demographic decline would, in fact, provide opportunities for the conservation of certain species (Hinojosa et al., 2018) and habitats (Regos et al., 2016) that are vulnerable to human pressures and disturbances (Henle et al., 2008). The extensive-use alternative, which shares elements of traditional land use, would also favour biodiversity to some extent by maintaining and promoting activities compatible with the persistence of multiple habitats and critical resources for some species (Beckmann et al., 2018) in certain landscape configurations (Franz et al., 2011; Varga, 2020; but see Lopez & Pardo, 2018 for the negative impacts on biodiversity of ecotourism and outdoor activities). At the same time, this alternative would promote economic activity favouring the maintenance of the human population, and even its increase (Caceres-Feria et al., 2021; Hospers & Reverda, 2015).

The frozen landscape alternative would have a less beneficial effect on biodiversity. Although human pressure has diminished in depopulated areas during the late 20th century, these biodiversity constituents seem unable to recover the same levels exhibited in distant historical times (Blondel et al., 2010; Navarro et al., 2015). It is worth noting that, although this alternative appears to ensure the status of certain habitats and species, it lacks any integrated vision of their dynamics at the landscape level. Furthermore, this frozen landscape would not be expected to substantially reverse any prevailing depopulation, given the limited impact on the human population of actions devoted to maintaining the landscape.

The crossed-arms alternative, in its turn, would maintain the current situation of population collapse, together with a certain deterioration in biodiversity, due to the loss of semi-natural open habitats by encroachment and afforestation (Halada et al., 2011). However, an incorporation of species inhabiting mature successional stages and vertebrates sensitive to anthropogenic pressure is also expected (Hirt et al., 2021; Regos et al., 2016). Finally, the intensified-use alternative produces the greatest negative impact on biodiversity (Beckmann et al., 2018; Carmona et al., 2020; Franz et al., 2011) by affecting habitats and altering the quantity and quality of water resources and soils, while it can counteract demographic decline, at least temporarily.

The component of the trajectories associated with the impact on biodiversity conservation would not vary substantially when comprehensively considering the inequality of the size of inhabited nuclei size (see fireworks graphs in Figure 3). However, the trajectories of the alternatives of extensive-use and active conservation would differ, since the former would promote a greater number of inhabited nuclei of intermediate size, and therefore, a greater inequality in the distribution of the population in the territory, than the latter, in which small nuclei are likely to disappear (Figure 3b). Similarly, the crossed-arms and frozen landscape alternatives would maintain the current population dilution and concentration in larger nuclei sometimes out of the territory, while the intensified-use alternative could retain some population in nuclei of intermediate size, through work in infrastructures, industries and new facilities. These demographic trends remain constant throughout the analysis of other environmental dimensions, and so their results will not be repeated further.

The trajectories of the different alternatives could be modified when the loss of population in the territory implies a greater deterioration of biodiversity, due, for example, to the loss of certain habitats. In that case, the extensive-use alternative would not be sufficient to reverse the loss of diversity, although it could alleviate it. This effect would be counterbalanced by biodiversity improvements driven by depopulation.

3.2 | Carbon storage

The carbon storage of Mediterranean-type ecosystems mainly relies on above- and underground woody vegetation (Vayreda et al., 2012), although the role of organic matter in grasslands is far from negligible (Doblas-Miranda et al., 2013). Therefore, forest cover and age constitute a proxy for this service. Overall, an increase in carbon storage is expected with the crossed-arms alternative (Figure 3c,d) since it reproduces recent trends of afforestation and carbon sequestration following land abandonment in Europe (Fuchs et al., 2016). However, carbon storage can be challenged by raising wildfires due to the increase of fuel and climatic risk (Duane et al., 2021). An increase in carbon storage is also to be expected with the frozen landscape and active conservation alternatives, as they favour the regeneration and growth of forests. This increase would be lower than in the crossed-arms alternative, since the maintenance of forests would be counterbalanced by the conservation of open habitats, which in the case of active conservation would be a crucial objective as they host some specialized species (Varga, 2020). Similarly, carbon storage is not expected to increase so substantially under the extensive-use alternative, due to the enhancement of open habitats for extensive livestock grazing; but no clear drop can be expected either, thanks to the conservation of forest remnants in the landscape. Finally, the destruction of forested and mountain grasslands by specific interventions associated with the intensified-use alternative would diminish carbon stocks (Doblas-Miranda et al., 2013).

3.3 | Pollution

Pollution in depopulated rural areas is expected to diminish due to the current reduction in the population and its associated economic activity (Bruno et al., 2021) and to the ability of ecosystems, particularly aquatic ones, to recover from both concentrated and diffuse sources of pollution (Donohue et al., 2010). This environmental improvement would reach a peak with active conservation and, to a lesser extent, with the frozen landscape alternative, due to its higher remaining activity, compared with the rest (Figure 3e,f). In contrast, intensification is expected to increase pollution in soils and water due to intensive farming (Ladrera et al., 2019), mining (García-Carmona et al., 2019), recreational resorts and installations for energy production (Heal et al., 2020). In fact, depopulated territories are often sought as sites for the disposal of hazardous products, such as industrial or radioactive wastes (e.g. Llurdes et al., 2003), representing intensive use of the territory. Human activity derived from extensive use is also expected to produce some levels of pollution, although much less than in this intensified-use alternative.

3.4 | Water resources

Local water input into the social–ecological system is directly determined by climate, aquifers and vegetation conditions. Soil quality and vegetation cover also play essential roles in regulating water supply. Water demand, in its turn, is determined by human population and economic activity. Water requirement is not constant over time, so water supply must be ensured in periods of peak demand or low availability—that is, drought periods. Therefore, depopulation and reduced economic activity will decrease the demand for water, while also enhancing water regulation by vegetation and soils (Pisabarro et al., 2019). Accordingly, those alternatives (crossed-arms, frozen landscapes, active conservation) involving low water demand derived from low levels of population density and reduced industry and agricultural intensification would, together with some degree of vegetation and soil preservation, increase both water resources and their regulation (Bruno et al., 2021). This increase would be proportional to the expected recovery of vegetation. This trend can be counterbalanced to some extent by the higher water requirements of forested lands, although this effect is limited in comparison with the impact of climate change (Otero et al., 2011). Overall, active conservation would represent the lowest water demand, while maintaining higher water stocks and groundwater recharge, as supported by the role of protected areas in providing regulating services (Castro et al., 2015). The crossed-arms and frozen landscapes alternatives would also maintain a positive balance of water resources to some extent (Figure 3g,h). In contrast, the intensified-use alternative would reduce water resources, due to high demand (e.g. for irrigation, Vidal-Legaz et al., 2013) and a likely reduction in vegetation and soil preservation. This reduction in water resources is expected to be less marked in the extensive-use alternative, due to lower demand for water and better preservation of vegetation and soils at landscape level (Ropero et al., 2019).

3.5 | Soil conservation

Soil erosion has been exacerbated by past deforestation, overgrazing and recurring fires associated with pastoralism, particularly in Mediterranean mountain areas (Sanjuan et al., 2016), despite the retention of soil provided by traditional terraces (García-Ruiz, 2010). So, depopulation tends to improve soil retention and development (Beguería et al., 2003; Bruno et al., 2021), which, in turn, produce cascade effects by enhancing revegetation processes (Navas et al., 2008) and reducing off-site costs (e.g. quality of surface and ground water; Colombo et al., 2005). However, the abandonment or destruction of terraces on croplands would endanger soil, reducing any such improvement in the crossed-arms alternative (García-Ruiz, 2010; MacDonald et al., 2000), particularly with an increased frequency and intensity of wildfires (Cerdà & Lasanta, 2005). The risk of soil loss would be less pronounced in the frozen landscape and, to a lesser extent, extensive-use alternatives, as vegetation cover would be enhanced, and terraces could be somewhat better preserved. The most effective alternative for soil conservation will be active conservation, as it would increase the vegetated area; it would be even more effective if soils were specifically included as a target for conservation and restoration, particularly in drylands (Bautista et al., 2010). In contrast, soil conservation is put at risk in the intensified-use alternative, particularly when mining or agricultural intensification are involved (Bruno et al., 2021; Cerdà et al., 2012). In contrast, soil conservation could benefit in the extensive-use alternative if some of the soils affected by intensive land use revert to a non-intensive use. This shift would allow some soils to return to semi-natural conditions (Figure 3i).

3.6 | Assessment of the alternatives for confronting rural depopulation

Overall, our qualitative assessment of the alternatives reveals that the most favourable alternative is extensive-use, according to their capacity to reverse depopulation and maintain biodiversity and environmental quality, together with the feasibility of their implementation and their ability to promote resilience of the social–ecological system (Table 1). In this alternative, none of the criteria show low levels of achievement and the best demographic consolidation and resilience are found here. In contrast, the alternatives that obtain the worst rates are intensified-use and active conservation, with five criteria rated negatively (high environmental impact, low depopulation reversal, high economic investment, difficult governance, and low resilience in the intensified-use alternative; and weak demographic consolidation, low economic revenue after substantial economic investment, difficult governance, and long set-up time in the active conservation alternative). This assessment is supported by studies comparing the outputs of different land uses related to our alternatives in Spain (Bruno et al., 2021; Castro et al., 2015; Franz et al., 2011; Varga, 2020; Vidal-Legaz et al., 2013), and the assessments of biodiversity and ecosystem services shown above.

4 | DISCUSSION

4.1 | Integrating ecological and social dimensions into depopulation assessment

In the context of rural depopulation, social–ecological dynamics are historically determined by both natural and social processes, as reflected in cultural landscapes. This rationale implies that: (i) there are ecological legacies that must be interpreted and considered when planning and applying decisions affecting depopulation; (ii) previous historical situations do not provide exact references suitable for replication (Balaguer et al., 2014) but they provide valuable information on key ecological processes; and (iii) in a given historical context, management should aim to favour the resilience of biodiversity and ecosystem services (Connell & Ghedini, 2015).

One important issue when integrating social and ecological processes is the difference in the time and spatial scales at which they operate. For instance, Gosselin and Callois (2021) have found that current patterns of biodiversity in terms of proportion of extinct species show, at the European national level, a time lag of approximately one century compared to human population density—extinction debts. Moreover, the abandonment of agricultural practices results in a secondary succession of vegetation, which takes decades to produce an increase in the forest surface (Duane et al., 2021), and even longer to culminate in highly structured stands (Rozas et al., 2009) and balance carbon budgets with a significant contribution of soil organic matter (Fuchs et al., 2016). Also, regulation of water quality shows important lags between production and consumption, with substantial spatial mismatching (Fremier et al., 2013). Therefore, according with these studies, a general recommendation is that short-term economic benefits should be contrasted with environmental gains, which often only emerge in the medium and long term.

Consequently, contextual dependencies need some kind of spatial–temporal up-scaling. Actions and demands in a given environment have an impact on other parts of a territory, and on other components of the social–ecological system (Bruno et al., 2021; Caceres-Feria et al., 2021; Costa et al., 2017; Davoudi & Stead, 2002). This applies to environmental impacts such as pollution, as well as the demands of urban populations for agricultural, livestock and forest resources, and recreational use, as particularly reflected in the intensified-use alternative and, to a lesser extent, the extensive-use, frozen landscape and crossed-arms ones. Furthermore, rural populations also present a demand for technological resources created in urban areas. Therefore, any solutions for facing depopulation and integrating ecological conditions should optimize the cost–benefit balance at different local, regional, and global scales. It is thus important to apply the principle that rural and urban systems are part of a functional and ecological continuum (Tacoli, 1998). They are mutually dependent in terms of flows of energy, matter, goods, information, and people, as well as in terms of social and ecosystem services. Although this principle has been recognized by ecologists (Villarroel Walker & Beck, 2012), social perception and political practitioners often confront territorial and environmental challenges separately, thereby obstructing any overall assessment or application of effective solutions.

Dealing with these conflicts implies that we must (i) prioritize those actions with the greatest synergies and evaluate existing trade-offs, and (ii) consider different spatial and temporal scales in order to optimize the distribution of actions in space and time. Note that some actions, such as 'rewilding'—which aims to re-establish integrated ecological processes by improving homeostasis in ecosystem functioning (Svenning et al., 2019), by re-establishing key elements such as locally extinct herbivores or carnivores—shares objectives and procedures that we can recognize in more than one alternative (in our case, extensive-use and active conservation), albeit with different degrees of involvement. We have knowledge and tools (cost–benefit analysis, multi-criteria, modelling) that provide us with a well-founded and rational approach to these evaluations (e.g. Martínez-Sastre et al., 2017). The participation of the population in decision-making is essential here, and it must be sustained by the dissemination and understanding of scientific knowledge. In Spain, strong social conflicts are expected to appear with the intensified-use and active conservation alternatives, resulting in challenges to governance. While intensification will trigger environmentalist reactions (Costa et al., 2017), active conservation will face opposition from local populations seeking to maintain their control over land resources (Hinojosa et al., 2018). These conflicts become ambivalent when the proposed infrastructures in the intensified-use alternative promote local economic activity, particularly when the latter is apparently consistent with the historical economic activity or provides environmental benefits on a broader scale (e.g. renewable energy). In turn, any implementation of active conservation should be slow, in keeping with the pace of not only ecological processes but also administrative obstacles (Perez-Barberia et al., 2023), and social acceptance, which can vary from refusal to acceptance, such as in programmes involving large carnivores (Delibes-Mateos et al., 2022; Recio et al., 2020, respectively).

The integrated maintenance over time of ecological and social processes is well-exemplified by resilience (Folke et al., 2010; Walker et al., 2004). Here, this is expected to be prominent in the extensive-use alternative, thanks to a diffuse spatial distribution of human activities that permits the functioning of essential self-regulatory ecological mechanisms (i.e. those involving biodiversity and soil and watershed integrity). This implies, for instance, that decision makers confronting depopulation should favour diverse landscapes (i.e. hosting grasslands, woodland and forests) with high species diversity as a means to buffer the impacts of pests, extreme weather events and wildfires (Fischer et al., 2006; Varga, 2020). In contrast, alternatives such as active conservation and frozen landscape, which tend to dissociate human activities from cultural landscapes, can threaten some components of the ecosystems which have historically been modelled by human activity (i.e. diverse mountain pastures, or domesticated lineages of animals and plants). For instance, an effective fire suppression policy aimed at preserving forests can also contribute to an increase in fuel load and may favour large, highly intense wildfires that, in turn, reduce the resilience of burned forests (Lloret, Piñol, et al., 2009). The maintenance—or even promotion—of ecological self-regulating processes, such as herbivory by large mammals, could counterbalance any loss of the regulatory capacity associated with human activity. However, when this capacity is completely lost, as tends to occur in the crossed-arms and intensified-use alternatives, resilience is expected to be dramatically depleted. Current trends of rural abandonment, corresponding to the crossed-arms alternative, are also increasing the fuel load, which—combined with increasing climatic risk—is triggering wildfires with higher intensities and frequency (Duane et al., 2021), jeopardizing important ecosystem services, such as soil conservation. In the intensified-use alternative, the destruction of habitats and disruption of ecological processes are particularly liable to reduce resilience (Laliberté et al., 2010). For instance, soil losses due to such intensification dramatically deplete the capacity of vegetation to establish itself (Colombo et al., 2005). This effect is even more obvious when exploitative activities constitute harmful disturbances in themselves or amplify a pre-existing natural disturbance regime (e.g. García-Carmona et al., 2019).

4.2 | Fire-works procedure for an integrative depopulation assessment

Given its multidimensional conception, the proposed procedure based on the evaluation of land management alternatives and their impact on human demographic and ecological trends is useful for an integrative semi-quantitative assessment of rural depopulation, despite reservations about a lack of precise, specific information. Interestingly, the procedure can be applied to situations other than the Spanish case, corresponding to different territorial, socio-economic or historical situations addressing the environmental consequences of depopulation. In fact, it can become more comprehensive if it incorporates additional operations, by (i) combining different environmental dimensions, for example, by weighting and multiplying their respective vector components; (ii) incorporating alternatives other than those exposed here for the Spanish case, once their effects on demography and ecological dimensions have been evaluated; (iii) combining the consequences of different alternatives operating in distinct parts of a given territory, for example, by weighting the coordinates of the vectors that describe the respective trajectories for the proportion of the territory affected by each alternative; (iv) analysing trends at different future time lags, that is, these intervals may represent different periods marked by changes in the relative weights of different alternatives; and (v) estimating lagged effects and other nonlinear responses of demographic parameters and environmental dimensions.

Our procedure can also evaluate other properties of interest for the social–ecological system. For instance, resilience would correspond to the distance of the final coordinate of the vector on the axis of the environmental dimension, with respect to a historical reference without rural depopulation. The overall magnitude of the transformation of the social–ecological system under the different scenarios (response intensity or responsiveness) corresponds to the angle or slope of the vector, with respect to the axes that describe the demographic and environmental dimensions. Finally, the directionality of the change corresponds to the maintenance of the direction of the vectors in a given alternative over successive periods of time.

5 | CONCLUDING REMARKS

Planning and actions carried on to confronting rural depopulation must integrate the ecological perspective. In the current context of biodiversity loss, climate change and environmental degradation, rural depopulation has clearly acquired a dimension that goes beyond a merely socio-economic and cultural analysis. Land uses in depopulated territories range from promoting biodiversity conservation, which provides opportunities to confront the global environmental crisis and to subsequently implement mitigating measures, to intensification of land use. In addition to the specifics of the physical environment and the current social situation, the historical legacies determining the spatial–temporal contexts of each locality are crucial to any understanding of the patterns of change in ecosystem services and biodiversity in all their complexity. Furthermore, the current crisis of globalization and climate change, and the value being placed on the use of local natural resources and environmental health, are leading to a reassessment of the opportunities provided by these territories.

We provide a procedure for an integrated analysis of demographic trends and ecological properties and processes of the social–ecological systems, particularly those resulting in biodiversity conservation and ecosystem services. Our evaluation of these alternatives in Spain indicates that management based on the extensive sustainable use of natural resources—including activities related to recreation and biodiversity conservation, as well as the provision of ecosystem services—would be the option that best fulfils the criteria for demographic consolidation, lower environmental impact, increasing resilience, and feasible implementation. More detailed analyses should be carried out, based on local characteristics and a specific assessment of the performance of demographic and environmental parameters in terms of realistic alternative approaches. Crucially, any depopulation analysis should be based on a set of basic ecological principles that must be included in the implementation of any mitigating actions.

AUTHOR CONTRIBUTIONS

Francisco Lloret, Fernando Valladares, Joan Lloret, and Adrian Escudero conceived the ideas and designed the methodology; Francisco Lloret, Fernando Valladares, Joan Lloret, and Adrian Escudero implemented the proposed methodology in the Spanish study case from published literature and discussed the interpretation; Franciso Lloret led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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CONFLICT OF INTEREST STATEMENT

The authors have no potential sources of conflict of interest to disclose, as established by the policy of the journal 'People and Nature'.

DATA AVAILABILITY STATEMENT

The article is bibliographic, based on the published literature and the study has not generated data eligible to be stored in a public archive.

ORCID

Francisco Lloret https://orcid.org/0000-0002-9836-4069
Adrián Escudero https://orcid.org/0000-0002-1427-5465
Joan Lloret https://orcid.org/0000-0002-9070-4639
Fernando Valladares https://orcid.org/0000-0002-5374-4682

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Data S1. Supplementary material.

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