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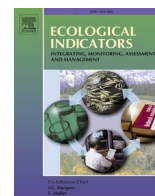
Choice of biodiversity indicators may affect societal support for conservation programs

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A B S T R A C T

Preservation and sustainable use of biodiversity brings multiple health, societal and economic benefits, including life-supporting services. Biodiversity indicators are important in framing the benefits of conservation and management programs and monitoring progress toward their outcomes. Biodiversity indicators therefore provide useful tools for policymakers in helping to communicate the benefits of conservation to society but also in garnering public support for conservation. This research aimed to help improve our understanding of the role of biodiversity indicators in the way that they influence preferences towards conservation programs. A discrete choice experiment was used to estimate relative societal preferences towards multilevel dimensions of biodiversity in relation to the conservation of pine forests in the Spanish Iberian Peninsula. Results show that (i) the level of biodiversity indicator (within species, between species and within ecosystems) matters, (ii) indicators related to the biodiversity within ecosystems are valued the most, and (iii) the use of several biodiversity indicators together is generally better at delivering benefits to society, but the value of these is reduced where there is redundancy between them. Overall, the most preferred indicators were the area of land covered by the conservation project, the status of keystone ecosystem components, and the number of native species. Some indicators such as invasive alien species and genetic diversity are least preferred and may be less helpful to how conservation efforts are perceived by the citizens. By careful consideration of which biodiversity indicators to use, policymakers and conservation managers can maximize societal acceptability of public investments in conservation efforts.

1. Introduction

Preserving and restoring biodiversity is a necessity for societies in the context of the biodiversity and climate crises (Díaz et al., 2020). Biodiversity underpins the resilience of ecosystems contributing to secure the flow of multiple benefits from ecosystem services, including human health protection (Mace et al., 2012; Cardinale et al., 2012; Kilpatrick et al., 2017; Díaz et al., 2020), crucial under the uncertainties of changing global environmental conditions (Finger and Buchmann, 2015; Martín-López et al., 2007; Millar et al., 2007). However, biodiversity is a complex term with many attributes and acceptations in ecology (Hamilton, 2005; Mace et al., 2012). This complexity means that no single ecological indicator captures all the dimensions of

biodiversity and an extensive literature defines a wide set of indicators to gather it (Pereira et al., 2013). Biodiversity indicators are essential to reinforce successful conservation policies and improve the detection of significant changes in global biodiversity. They play a surveillance role in monitoring the status and trends of biodiversity towards agreed political targets (Butchart et al., 2010; Turak et al., 2017; Mace et al., 2018), can guide decision making by generating predictions on potential impacts of alternative policy options (Collen and Nicholson, 2014; Costelloe et al., 2016), and allow for rigorous evaluations, characterized by causal mechanisms, of how the implementation of conservation policies made a significant difference in biodiversity (Ferraro and Pattanayak, 2006; Fisher et al., 2014; Baylis et al., 2016). However, selecting appropriate indicators to focus on for this range of operational

uses is not straightforward (Heink and Kowarik, 2010; Otto et al., 2018), and conservation programs can be adversely affected if their objectives and approaches are in conflict with societal preferences (Tanentzap et al., 2009). Here, we contribute to this literature by focusing on the indicators ability to trigger public interest in supporting conservation programs.

There is a substantial body of the literature that employs environmental valuation techniques to estimate the effects of changes in biodiversity on individuals' well-being to inform decision-making (Hanley and Czajkowski, 2019; Hanley and Perrings, 2019). Most often, this literature uses one indicator of biodiversity, typically, species richness or species abundance, to provide evidence on the social benefits of biodiversity by examining the choices people make between different levels of species conservation and the cost of provision (Bartkowski et al., 2015). There are also some works that focus on valuing multi-level changes in the characteristics of biodiversity itself (Nunes and van den Bergh, 2001; Christie et al., 2004; Czajkowski et al., 2009; Finger and Buchmann, 2015; Laurila-Pant et al., 2015; Bartkowski, 2017). These studies show that people's preferences for protecting biodiversity depend not only on how the policy is delivered (Austin et al., 2014; Martínez-Jauregui et al., 2020), but also on the perceived consequences of the policy, for example whether the outcome is framed in terms of habitat restoration versus habitat creation (Christie et al., 2006), which natural ecosystem processes and components such as dead wood, natural ponds, and clearings, are enhanced (Czajkowski et al., 2009), and what role biodiversity is perceived to play in relation to ecosystem services supplied (Martínez-Jauregui et al., 2019). Overall, this suggests that people are concerned with the way in which biodiversity change is delivered; and the protection or enhancement of a particular biodiversity indicator is an important element of the perceived value attached to an environmental protection program.

Here, we evaluated how people respond to the process of 'labelling' conservation programs based on a range of biodiversity performance indicators. In particular, we investigated the effect of using different indicators on the values people place to support biodiversity conservation programs using a discrete choice experiment. Following the definition of biodiversity given by the Parties to the Convention on Biological Diversity, we defined biodiversity at three levels of organisation and chose two common indicators used in the literature per each level, thereby exploring the economic preferences towards attributes of biodiversity in a more comprehensive manner than previous research. Finally, we recognised that conservation interventions often strive to achieve multiple biodiversity targets and at multiple scales, and that the independence between multiple ecological indicators cannot be guaranteed. Hence, we conducted a follow-up analysis of how survey respondents processed the decision of selecting a set of indispensable indicators and their preferences for a unique performance indicator.

2. Material and methods

A discrete choice experiment establishing a hypothetical market to value environmental changes (Carson and Louviere, 2011; Johnston et al., 2017) was used to gather information on social preferences towards indicators of multilevel dimensions of biodiversity changes resulting from hypothetical conservation programs in pine forests in Spain. Spanish pine forests are important in the territory and rural development, because they are widely distributed ecosystems, they have traditionally supported a wide range of human activities such as timber extraction, harvesting of pine nuts and mushrooms, hunting, and production of resin and derived products, they provide opportunities for leisure and enjoyment, and they generate a range of habitats that are important for conservation (Serrada et al., 2008).

The choice experiment approach relied on setting alternative hypothetical conservation programs, which are presented to respondents, who are asked to choose their most-preferred alternative. This method allowed us to assess the relative values people attach to the different

indicators of biodiversity targeted in those programs. Three dimensions of biodiversity (within species, between species and within ecosystems levels) were used in our discrete choice experiment to characterise the complexity in measuring biodiversity (Noss, 1990; Mace et al., 2012), and at the same time addressed the necessary simplicity in the presentation of the 'good' in a choice experiment in a manner that is comprehensible and familiar to the general public (Czajkowski et al., 2009; LaRiviere et al., 2014; Hanley and Perrings, 2019). Thus, in the questionnaire, respondents were initially provided with a detailed definition of biodiversity and its multidimensionality following the Convention of Biological Diversity; and informed that: (1) there are many types of programs of biodiversity conservation that improve certain aspects of the biodiversity in pine forests, but to carry them out more funds are necessary; (2) to ensure the funding of the conservation program presented to them, a specific tax is proposed, i.e. the money will be used exclusively for the conservation of the biodiversity of Spanish pine forests; (3) these funds would come from the taxes citizens would pay, and thus it is desirable to consult members of society regarding which aspects of pine forest biodiversity these conservation efforts should focus on over the next ten years. In order to describe potential changes in biodiversity indicators in the discrete choice experiment, we used quantitative empirical data collected from previous ecological research in Spanish pine forests when available (Martínez-Jauregui et al., 2016a; 2016c; 2018a; 2018b). Indicators were presented visually to respondents on choice cards using icons of mammals, birds, and plants randomly to avoid taxon bias (Martín-López et al., 2007; Ressurreição et al., 2012); we avoided using icons of charismatic species to prevent flagship and irreplaceable feeling bias (White et al., 1997; Christie et al., 2006; Jacobsen et al., 2008). The indicators used in each of the three levels (within species, between species and within ecosystems levels) were as follows:

- *Genetic variation* (GEN; within species level indicator). Genetic variation is used frequently as an indicator in the captive breeding literature. There are very few environmental valuation studies which use genetic diversity as a proxy for biodiversity (Feld et al., 2009; Bartkowski et al., 2015; Soliño et al., 2020). However, the use of this indicator is justified as it is associated with the adaptability of species to changes in the ecosystem (Reed and Frankham, 2003). We distinguished two attribute levels for this indicator within the conservation initiative: (*i*) genetic diversity is not targeted, and (*ii*) conservation measures are targeted to maintain genetic diversity.

- *Population structure* (POPSTR; within species level indicator). Population structure is often used because it captures an important dimension of a population, such as old trees or trophy animals, which are missed in simple indicators of population size, and it is simple to measure (Gao et al., 2015; Nielsen et al., 2007; Bullock et al., 1998). In our case POPSTR addressed the structure balanced of ages and sexes for each species which can have important consequences in population viability (Ginsberg and Milner-Gulland, 1994; Martínez-Jauregui et al., 2016b). We distinguished two levels for this indicator: (*i*) population age structure is not balanced, and (*ii*) there are conservation measures to ensure that populations are balanced.

- *Number of native species* (NNS; between species level indicator). The number of native species is frequently used to indicate biodiversity as a whole in biodiversity valuation studies (Feld et al., 2009; Nordén et al., 2017; Bartkowski et al., 2015) with native bird species richness commonly used as an indicator of ecosystem biodiversity (Gregory et al., 2008; Gao et al., 2015; Martínez-Jauregui et al., 2018b). NNS was defined as the number of native birds in the pine forests. Taking into account existing data on the number of native bird species in pine forests in Spain (Martínez-Jauregui et al., 2016a; 2016c), we defined three levels for this indicator: (*i*) 24 native bird species, (*ii*) 25 native bird species, iii) 26 native bird species.

- *Number of invasive alien species* (NIAS; between species level indicator). This indicator captures the threat that invasive alien species

(IAS) pose to biodiversity and ecosystem services (Bellard et al., 2016; Doherty et al., 2016; Early et al., 2016; Rai and Singh, 2020). Including this attribute extends previous valuation studies that focus on the willingness to pay for different actions to control an invader (Adams et al., 2011; Fleischer et al., 2013; Bithas et al., 2018). We defined three levels of this indicator based on estimates of the range of invasive alien plants, birds and mammals species in Spanish pine forests (Martínez-Jauregui et al., 2018b): (*i*) no IAS control and presence of two invasive alien species, (*ii*) moderate IAS control, presence of one invasive alien species iii) high control, with no IAS present. Examples of plant, bird and mammal IAS (*Acacia dealbata* Link, *Carpobrotus edulis* L., *Estrilda astrild* L., *Psittacula krameri* Scopoli, *Mustela vison* Schreber, and *Procyon lotor* L.) commonly found in the case study area were used on the choice cards.

- *Keystone elements* (KEY; within ecosystems level indicator). Keystone elements have a profound effect on the functioning of entire ecosystems (Simberloff, 1998). In our case study, keystone elements were represented by the presence of either dead wood (Rondeux and Sanchez, 2010), rabbit (*Oryctolagus cuniculus* L., Delibes-Mateos et al., 2007) or the European green woodpecker (*Picus viridis* L., Gregory et al., 2008), since their absence could indicate that the ecosystem services of pine forests are in poorer condition. We distinguished two different levels of this indicator in the conservation programs: (*i*) conservation measures are not established to preserve keystone elements of the pine forest, (*ii*) conservation measures are included to preserve keystone elements of the pine forest.

- *Geographical area involved in the program* (EXT; within ecosystem indicator) was included as a variable in order to gather preferences regarding the extension and distribution of the pine forest areas conserved. Taking into account the extension of areas being currently protected within the Spanish territory, three levels were presented for this indicator: (*i*) 1% of the pine forests; (*ii*) 21% of the pine forests, and iii) 100% of the pine forests. The first two levels, i. e., 1% and 21% of pine forest, would correspond approximately to the area covered by the National Parks, and Red Natura in Spain, respectively. This information was used to help respondents to visualize these percentages in the territory. The third level was visualized using the spatial distribution of pine ecosystem in Spain. It is reasonable to assume that these levels allow for the possibility that respondents show their preferences towards either conservation of biodiversity clustered in few pine areas (i.e., sparing lands for exclusive conservation use in a small proportion of these ecosystems) or scattered throughout the territory including some degree of biodiversity-conservation practices (i.e., land sharing for conservation and productive uses) (Meli et al., 2019; Edwards et al., 2014; Fischer et al., 2014). Thus, the proposed enlargement of conservation efforts to all pine forest in Spain was necessary to inform respondents that at this level, any program involved managing both planted and natural pine forest with some biodiversity targets. This allows us to test the findings of previous studies of social preferences for managing forest for multiple uses including biodiversity (Giergiczny et al., 2015), in a vision where existing natural parks areas work synergistically with surrounding commercial, but biodiversity-friendly, pine forests, to increase connectivity for wildlife. Based on previous analysis of social preferences for forest management in Spain (Oviedo and Yoo, 2017; Soliño et al., 2018), we hypothesized an inverted U-relationship between extension of conserved area and value for an additional increase in area conserved.

Data were collected from 400 people from a stratified consumers' panel within Spain attending to gender, age, and rural–urban habitat. On-line structured surveys were conducted in 2016. This included a pilot study of 40 people, which was conducted to obtain the priors for a D-efficient design of the choice cards. Moreover, twenty-six individuals were eliminated since they were identified as protest responses

(Meyerhoff et al., 2014). Our analysis used therefore data from 334 individuals. The full choice experiment involved 12 choice cards shown to each individual, with each card including 3 conservation programs and an opt-out, i.e. individuals always had the option to choose the option of "No program", which did not imply additional taxes but assumed no measures to enhance biodiversity conservation. To estimate the marginal willing to pay (WTP) values for each attribute (i.e., biodiversity indicator) a specification of utility in the willing to pay in space model (WTP-space) was employed (Train and Weeks, 2005). In this modelling approach the individual's utility from the choice alternative j can be represented as:

$$U_j = X_j\beta + C_j\delta + \varepsilon_j \tag{1}$$

where X_j represents the biodiversity indicators, and C_j is the cost (in euros) of the choice alternative *j*. The econometric approach was based on a mixed logit model (Train, 2009). β and δ were assumed to be random parameters with a normal distribution, and ε_j was an i.i.d. type I extreme value error term. The individuals' preferences were estimated by simulation using a willingness to pay in space model with 500 Halton draws (Train and Weeks, 2005), using the software NLOGIT 6.0®. In this model the WTP was incorporated directly in the utility function, i.e.

$$U_j = X_j\delta\lambda + C_j\delta + \varepsilon_j \tag{2}$$

where λ is the ratio between β and δ. This re-parameterization assures finite moments for the distributions of the willingness to pay (Daly et al., 2012). This approach affords more control to the researcher about the distribution of WTP values in the population (Scarpa et al., 2008) than conventional approaches based on estimations on the preference-space. Thus, the sample mean WTPs (λ) were directly estimated using mixed logit specification with random parameters normally distributed using WTP-space, and individual mean WTPs are derived from individual parameters.

The questionnaire also asked respondents about their opinion regarding which indicators were either the minimum required (hereafter referred to as "indispensable") or additional (hereafter referred to as "accessory") to measure progress in biodiversity conservation in pine forests. Respondents were subsequently required to select a unique indicator that best represents for them the conservation of biodiversity in pine forests. With this information we were able to calculate *(i)* the total willingness to pay (WTPt), calculated by adding up marginal WTP values of the significant indicators in choosing a preferred conservation program; *(ii)* the indispensable willingness to pay (WTPi), corresponding with the linear aggregation of the individual set of marginal WTP values of the significant biodiversity indicators considered to be indispensable when assessing biodiversity in pine forest; and *(iii)* the minimum willingness to pay (WTPm), given by the marginal willingness to pay associated with the significant unique biodiversity indicator chosen by every respondent. In case the respondent was not able to choose a unique indicator (16 individuals of 334); the WTPm was assumed to be equal to the WTPi. Wilcoxon signed rank test with continuity correction was used to explore differences in the willingness to pay between these categories of indicators (WTPt, WTPi and WTPm). A sensitivity analysis of the total, indispensable and minimum willing to pay was performed when varying the geographically extent of the conservation program.

3. Results

Table 1 shows the results of the willingness to pay in space model. These results show that respondents distinguished among the different biodiversity indicators since all of them were statistically significant at the 1% level in explaining their choices of the conservation programs and have the expected signs. The positive signs are plausible considering that improvements in biodiversity indicators are associated with higher willingness to pay for conservation outcomes such as species and genetic diversity, balanced populations, keystone elements and area of land

Table 1
Results of the willingness to pay in space model (in Euros).

	Coefficient	Standard Error	z-value
Random parameters in utility function			
Genetic diversity	15.862***	2.042	7.77
Population structure	20.982***	2.157	9.73
Number of native species	10.683***	1.911	5.59
Number of invasive alien species	−7.979***	2.531	−3.15
Keystone elements	26.793***	2.032	13.19
Extent	5.818***	0.748	7.78
Extent^2	−0.043***	0.006	−6.76
Standard deviation of random parameters			
Genetic diversity	0.358	6.022	0.06
Population structure	16.614***	2.749	6.04
Number of native species	1.742	8.139	0.21
Number of invasive alien species	27.006***	2.478	10.9
Keystone elements	13.671***	3.479	3.93
Extent	0.086	0.140	0.62
Extent^2	0.005***	0.001	9.01

***, **, * are significance at 1%, 5%, 10% level.

protected. The quadratic relationship with the geographical area involved in the program indicates that the value attached for increasing the extent of the area where the conservation efforts are applied declines after the proportion of pine ecosystems protected reach a certain level. A negative marginal WTP is observed for increases in the number of alien invasive species in pine forests, i.e., the value granted to biodiversity protection programs increases with lower numbers of invasive species (higher levels of invasive species control). Individual specific mean WTP values vary with respect to the level of the indicators proposed to the respondent. The statistical significance of the standard deviation of random parameters signals the presence of heterogeneity across the respondents. In our case study, heterogeneity is shown for the population structure, number of invasive alien species, keystone elements and extent.

In the most complete conservation program, defined as the one with highest level of biodiversity attributes, the relative ranking of the estimated means of marginal WTPs, in regard to the no conservation program alternative, is as follows (see random parameters in utility function in Table 1). The extent of the land area over which the program is applied, is the indicator that has the highest marginal value of WTP (EXT model estimate = 146.79 euros when 100% of the pine forests in Spain are protected), followed by keystone elements (KEY = 26.79 euros, when program preserve keystone elements of pine forest ecosystems), and the number of native species (NNS = 21.37 euros, when program protects 26 native bird species). The biodiversity indicators with the lowest marginal WTP values are those metrics of biodiversity related to a balanced population structure (POPSTR = 20.98 euros, when conservation measures are in place to ensure that the populations are balanced), genetic diversity (GEN = 15.86 euros, when measures are established to maintain genetic diversity) and invasive alien species (NIAS = −15.96 euros, when measures are in places for having two invaders). These marginal values of WTP varied considerably with other hypothesised alternative conservation programs (i.e. attributes are defined with lower levels): marginal WTP for targeting the extent of land conserved as 21% of the pine forest area is 102.99 euros, while marginal WTP for conserving 1% of land is 5.77 euros. The marginal WTP is 10.68 euros when only one additional native species is protected (i.e., bird richness is targeted to 25 species), and 7.98 euros when only one invasive alien species is controlled.

Our results show that the set of indicators used to value the social benefits derived from a biodiversity conservation program, matters. The total and indispensable willingness to pay, WTPt (213.12 euros, SD = 41,870) and WTPi (171.07 euros, SD = 72,135), calculated as the average of individuals' WTP in the most completed program, were found to be significantly different (Wilcoxon signed rank test with continuity correction, V = 20460, p-value < 0.001). Similar results were found for

the differences between the minimum and indispensable willingness to pay, WTPm (57.46 euros, SD = 69,690) and WTPi (Wilcoxon signed rank test with continuity correction, V = 53625, p-value < 0.001). A sensitivity analysis was performed when varying the geographically extent of the conservation program, because this was the indicator that showed higher marginal WTP values (Table 2). These results show that WTPt is approximately 1.2 or 1.3 times higher than WTPi and 3.0 or 3.7 times higher than WTPm.

Fig. 1 shows the numerical ranking of indicators using the information on the respondents' decisions regarding their perception of the indicators as indispensable, accessory, or selected as best unique performance indicator for pine forest conservation programs. The indicators more often chosen by respondents as indispensable were keystone elements (82% of respondents), geographical area involved in the program and number of native species (both 81% of respondents). These were followed by population structure (74%) and genetic diversity (72%) (Fig. 1a). Nearly 40% of our respondents (37%) indicated that four was the number of indicators to be included in the indispensable set of indicators to be used for biodiversity conservation programs (Fig. 1b), while five sets of indispensable indicators were chosen by 28% of the respondents and three by 18%, and an indispensable set of only one or two indicators was chosen by <3% of the respondents. Within ecosystem level indicators (keystone elements and geographical area involved in the program) were often represented when indispensable sets were higher than three (Fig. 1c). When faced with the decision to choose a unique indicator, 26% of respondents chose to characterize biodiversity in terms of the geographical extent of the conservation program, 22% chose keystone elements, 19% the number of native species, 18% genetic diversity, 12% population structure and only 4% the number of invasive alien species (Fig. 1d).

4. Discussion

No single indicator represents all the properties of biodiversity with respect to composition, the level of organization, and function; therefore a set of biodiversity indicators is desirable to capture the multidimensional concept of biodiversity (Noss, 1990; Gao et al., 2015; Haila and Kouki, 1994). The choice of metrics used to evaluate the progress of conservation programs can influence individuals' preferences and the societal benefits attached to these initiatives (Zhao et al., 2013; Bartkowski et al., 2015). This study reinforces the role that biodiversity valuation can play in informing the choice of appropriate indicators to maximize societal support for conservation programs. Here we illustrated that individuals do appreciate differences between biodiversity indicators. Our results showed that the extent of land over which biodiversity is conserved has the highest marginal value of WTP given the increases of 21% and 100% in area protected explored in the discrete choice experiment. However, the WTP for percent increases in area protected exhibited an inverted-U form. Therefore, our results suggest that people prefer in general to manage pine forest for multi-stakeholders and multiple uses, including biodiversity conservation, but they experience decreasing marginal WTP after a certain percentage of pine ecosystem dedicated to conservation is reached. Keystone elements attract the second highest marginal value of WTP, and the first

Table 2
Sensitivity analysis of the total, indispensable and minimum willing to pay (WTPt, WTPi and WTPm, respectively), attending to the levels of the geographical extent of the conservation program (with the rest of the attributes of the conservation program in the complete version).

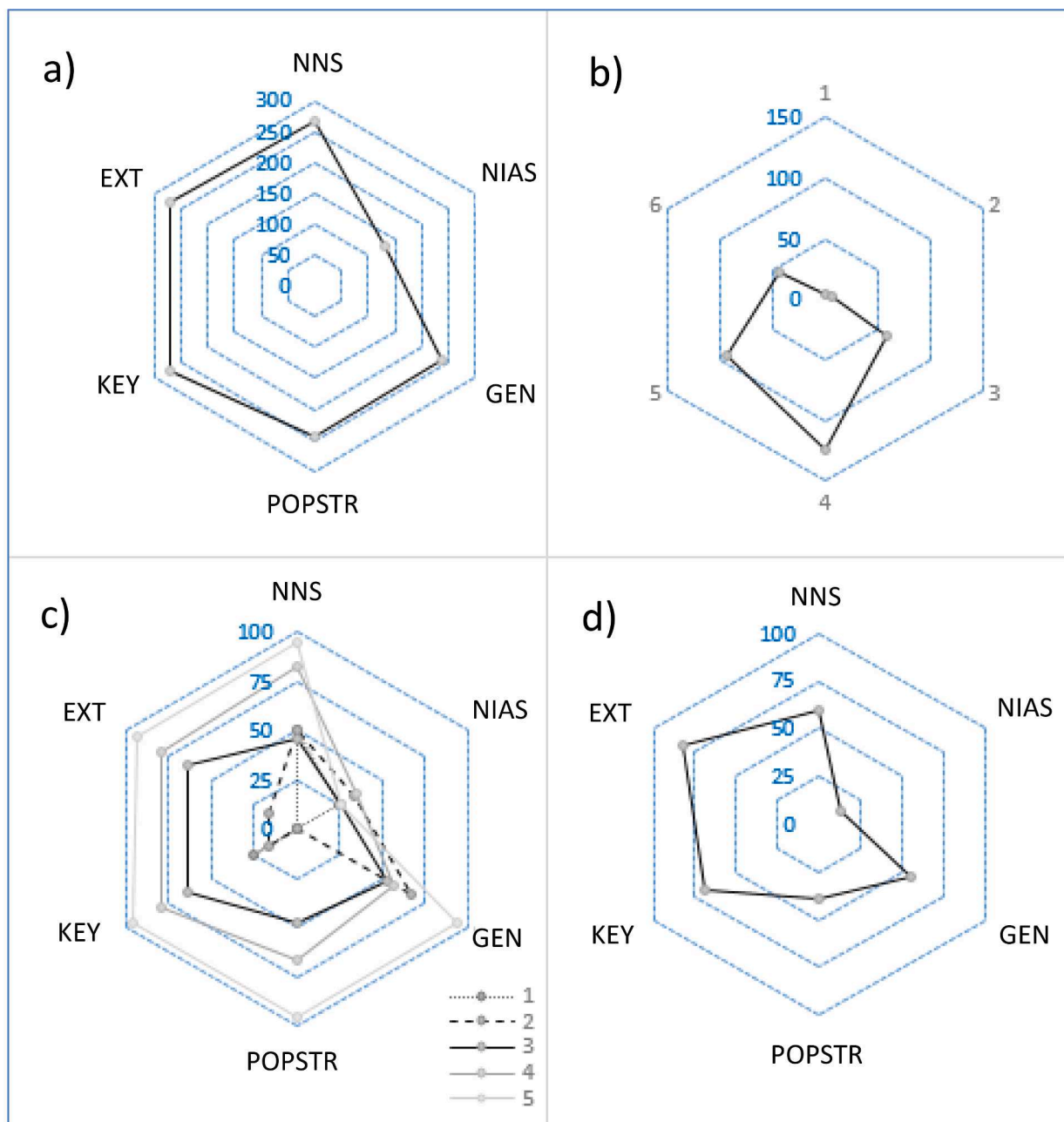| Percent of pine forest under conservation program	WTPt (euros)	WTPi (euros)	WTPm (euros)
1%	73.59	56.90	15.52
21%	170.73	135.49	42.99
100%	213.12	171.07	57.46

Fig. 1. a) Number of times that every indicator was selected as indispensable among the respondents. b) Number of respondents that selected different number of indicators as indispensable. c) Percentage of times that every indicator was selected depending on the number of indicators identified as indispensable. d) Number of the respondents that choose each indicator as unique.

place when the conservation program is smaller than 390,000 ha (4.5% of pine forest). This indicator can provide evidence that an ecosystem is in good conservation condition and its ecosystem functions are being maintained. Specific examples in the study area included rabbits in Mediterranean ecosystems, whose abundance is associated with the number of raptor species and the number of species of conservation concern (Delibes-Mateos et al., 2007), the presence of dead wood in forest ecosystems which is an essential factor in the nutrient cycle and provides habitat for numerous plants, animals and fungi in the forest (Rondeux and Sanchez, 2010), or the presence of woodpeckers, whose nest holes, chiseled out of tree trunks, provide habitat for numerous species (Gregory et al., 2008). The high level of social support to the conservation of within ecosystem levels is in accordance with a shift in emphasis of conservation ecology from species to landscape level (Mace, 2014). The number of native species received the third highest level of

social support, and is consistent with its common use in monitoring of the state of biodiversity in conservation and management programs (Varela et al., 2018). Birds are a large and varied group of species with different characteristics in their anatomy, physiology, growth form, life history, reproduction measures, feeding, behavior types and carry out multiple functions in the ecosystem. Therefore, native bird species has been used frequently in ecology as an indicator of biodiversity (Gregory et al., 2008), being able to capture the variety of life forms and the ecological complexities of which they are part. Balanced population structure, genetic diversity and invasive alien species were the biodiversity indicators with the lowest marginal WTP values. This probably reflects a combination of societal understanding and shifting ethical values. Balanced population structure is not a very accessible concept to the general public, and societal concepts of biodiversity do not necessarily link through to genetic diversity. In contrast, the impacts of

invasive alien species are widely realized. However, their management is increasingly controversial, with a growing proportion of the society is various countries being opposed in principle to culling of wildlife (Sharp et al., 2011), with the result that societal views can be at odds with those of conservationists (Lundberg, 2010). Emphasizing the impacts of conservation programs in controlling and eradicating invasive species may even be counter-productive in some instances, in terms of generating public support.

The results of a follow-up analysis of the respondents indicated that simply including different indicators does not per se guarantee public support increments. Moreover, this is more likely to result in non-independence between the indicators, with some of them being considered as redundant or providing accessory information. This does not imply that some indicators were perceived as having a zero value, but that their value could be embedded in the values of other dimensions of biodiversity (as the respondents stated in their answers). In particular, our results showed that four indicators (EXT, KEY, NNS and POPSTR) were most often selected as indispensable. If an indicator of each dimension of biodiversity is required, then our results indicated that EXT, NNS, and POPSTR could be the most preferable choices socially if program involves large areas; and KEY, NNS and POPSTR in conservation programs carried out in relatively small area.

Conservation programs that opt for within ecosystem elements as a single biodiversity indicator when communicating conservation efforts, such as the extent of the area to be conserved, have the highest WTP in our sample. This indicator, followed by keystone elements, was also most frequently selected by respondents when asked to choose a unique biodiversity indicator to assess the conditions of the case study ecosystem. The preference of respondents for this dimension of biodiversity may be because they perceive that it provides additional embedded information on other dimensions of biodiversity (Bakhtiari et al., 2014). Therefore, our results suggest the convenience to investigate in the role of 'cue attributes' in the individuals' decision process when addressing the WTP of complex multidimensional goods (Caputo et al., 2017) as biodiversity, which it is also consistent with a number of studies that use land use changes as an outcome metric to measure the effectiveness of conservation policies (Cuenca et al., 2016; Robalino et al., 2015). In practice, the implementation of conservation initiatives over large areas is challenging due to a lack of funding, the complexities of dealing with multiple landowners including land ownership and permissions, and diverse demands from the land in terms of ecosystem services. As a consequence, biodiversity conservation programs over relatively small areas are more common, and for these, our results suggest that keystone elements or native species are likely to be the best option for gathering social support to conservation programs if only a single indicator is used.

5. Conclusions

This work has extended the limited research on valuation of biodiversity in pine ecosystems in Spain (Soliño et al., 2018; Herruzo et al., 2016; Martínez-Jauregui et al., 2019; Campos et al., 2019), and it has deepened understanding of the importance of societal views in selecting appropriate indicators for biodiversity conservation programs. The results are context dependent and provide evidence that Spanish society supports biodiversity conservation programs in pine forests. However, people showed varied preferences over which dimension of biodiversity is best suited to serve as an indicator of success of these programs. The type of biodiversity indicator used can therefore affect the level of support for biodiversity conservation programs, and in our work the area of land covered, the status of keystone ecosystem components, and the number of native species are the indicators preferred by the public. The results also suggested that the use of certain indicators such as invasive alien species and genetic diversity may be less understood by citizens. The use of indicators that receive stronger support from society should complement rather than replace indicators that are most

ecologically relevant. Used in this way, they can provide a critical contribution to maximizing societal acceptability of public investments in conservation efforts.

CRediT authorship contribution statement

María Martínez-Jauregui: Conceptualization, Data curation, Methodology, Software, Funding acquisition, Writing - original draft. **Julia Touza:** Conceptualization, Writing - review & editing. **Piran C.L. White:** Conceptualization, Writing - review & editing. **Mario Soliño:** Conceptualization, Methodology, Software, Funding acquisition, Writing - review & editing, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Adams, D.C., Bwenge, A.N., Lee, D.J., Larkin, S.L., Alavalapati, J.R., 2011. Public preferences for controlling upland invasive plants in state parks: application of a choice model. Forest Policy Econ. 13 (6), 465–472.

Austin, Z., Smart, J.C., Yearley, S., Irvine, R.J., White, P.C., 2014. Incentivising the collaborative management of mobile ecological resources. Land Use Policy 36, 485–491.

Bakhtiari, F., Lundhede, N., Gibbons, J., Strange, N., Jacobsen, J.B., 2014. How should biodiversity be presented in valuation studies? Testing for embedding and information bias, in: Fifth World Congress of Environmental and Resource Economists.

Bartkowski, B., 2017. Are diverse ecosystems more valuable? Economic value of biodiversity as result of uncertainty and spatial interactions in ecosystem service provision. Ecosyst. Serv. 24, 50–57.

Bartkowski, B., Lienhoop, N., Hansjürgens, B., 2015. Capturing the complexity of biodiversity: a critical review of economic valuation studies of biological diversity. Ecol. Econ. 113, 1–14.

Baylis, K., Honey-Rosés, J., Börner, J., Corbera, E., Ezzine-de-Blas, D., Ferraro, P.J., Lapeyre, R., Persson, U.M., Pfaff, A., Wunder, S., 2016. Mainstreaming impact evaluation in nature conservation. Conserv. Lett. 9 (1), 58–64.

Bellard, C., Cassey, P., Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. Biol. Lett. 12 (2), 20150623.

Bithas, K., Latinopoulos, D., Kolimenakis, A., Richardson, C., 2018. Social benefits from controlling invasive Asian tiger and native mosquitoes: a stated preference study in Athens, Greece. Ecol. Econ. 145, 46–56.

Bullock, C.H., Elston, D.A., Chalmers, N.A., 1998. An application of economic choice experiments to a traditional land use—deer hunting and landscape change in the Scottish Highlands. J. Environ. Manage. 52, 335–351.

Butchart, S.H., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J.P., Almond, R.E., Baillie, J.E., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., 2010. Global biodiversity: indicators of recent declines. Science 328 (5982), 1164–1168.

Campos, P., Caparrós, A., Oviedo, J.L., Ovando, P., Álvarez-Farizo, B., Díaz-Balteiro, L., Carranza, J., Beguería, S., Díaz, M., Herruzo, C., Martínez-Peña, F., Soliño, M., Álvarez, A., Martínez-Jauregui, M., Pasalodos-Tato, M., de Frutos, P., Aldea, J., Almazán, E., Concepción, E.D., Mesa, B., Romero, C., Serrano-Notivoli, R., Fernández, C., Torres-Porras, J., Montero, G., 2019. Bridging the gap between national and ecosystem accounting application in andalusian forests, Spain. Ecol. Econ. 157, 218–236.

Caputo, V., Scarpa, R., Nayga, R.M., 2017. Cue versus independent food attributes: the effect of adding attributes in choice experiments. Eur. Rev. Agric. Econ. 44, 211–230.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., 2012. Biodiversity loss and its impact on humanity. Nature 486 (7401), 59–67.

Carson, R.T., Louviere, J.J., 2011. A common nomenclature for stated preference elicitation approaches. Environ. Resour. Econ. 49, 539–559.

Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., Hyde, T., 2006. Valuing the diversity of biodiversity. Ecol. Econ. 58, 304–317.

Christie, M., Warren, J., Hanley, N., Murphy, K., Wright, R., 2004. Developing measures for valuing changes in biodiversity. Final Report to Defra" (Defra, London, 2004).

Collen, B., Nicholson, E., 2014. Taking the measure of change. Science 346 (6206), 166–167.

Costelloe, B., Collen, B., Milner-Gulland, E.J., Craigie, I.D., McRae, L., Rondinini, C., Nicholson, E., 2016. Global biodiversity indicators reflect the modeled impacts of protected area policy change. Conserv. Lett. 9 (1), 14–20.

Cuenca, P., Arriagada, R., Echeverría, C., 2016. How much deforestation do protected areas avoid in tropical Andean landscapes? Environ. Sci. Policy 56, 56–66.

Czajkowski, M., Buszko-Briggs, M., Hanley, N., 2009. Valuing changes in forest biodiversity. Ecol. Econ. 68, 2910–2917.

Delibes-Mateos, M., Redpath, S.M., Angulo, E., Ferreras, P., Villafuerte, R., 2007. Rabbits as a keystone species in southern Europe. Biol. Conserv. 137 (1), 149–156.

Daly, A., Hess, S., Train, K., 2012. Assuring finite moments for willingness to pay in random coefficient models. Transportation 39 (1), 19–31.

Díaz, S., Settele, J., Brondízio, E., Ngo, H., Guèze, M., Agard, J., Arneth, A., Balvanera, P., Brauman, K., Butchart, S., Chan, K., 2020. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (https://www.ipbes.net/sites/default/files/downloads/spm_unedited_advance_for_posting_htn.pdf, accesed in July 2020).

Doherty, T.S., Glen, A.S., Nimmo, D.G., Ritchie, E.G., Dickman, C.R., 2016. Invasive predators and global biodiversity loss. Proc. Natl. Acad. Sci. 113 (40), 11261–11265.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., Grosholz, E.D., Ibañez, I., Miller, L.P., Sorte, C.J., 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. Nat. Commun. 7 (1), 1–9.

Edwards, D.P., Gilroy, J.J., Woodcock, P., Edwards, F.A., Larsen, T.H., Andrews, D.J., Derhé, M.A., Docherty, T.D., Hsu, W.W., Mitchell, S.L., 2014. Land-sharing versus land-sparing logging: reconciling timber extraction with biodiversity conservation. Glob. Change Biol. 20, 183–191.

Feld, C.K., Martins da Silva, P., Paulo Sousa, J., De Bello, F., Bugter, R., Grandin, U., Hering, D., Lavorel, S., Mountford, O., Pardo, I., 2009. Indicators of biodiversity and ecosystem services: a synthesis across ecosystems and spatial scales. Oikos 118, 1862–1871.

Ferraro, P.J., Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biol. 4 (4), e105.

Finger, R., Buchmann, N., 2015. An ecological economic assessment of risk-reducing effects of species diversity in managed grasslands. Ecol. Econ. 110, 89–97.

Fischer, J., Abson, D.J., Butsic, V., Chappell, M.J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H.G., Wehrden, H., 2014. Land sparing versus land sharing: moving forward. Conserv. Lett. 7, 149–157.

Fisher, B., Balmford, A., Ferraro, P.J., Glew, L., Mascia, M., Naidoo, R., Ricketts, T.H., 2014. Moving Rio forward and avoiding 10 more years with little evidence for effective conservation policy. Conserv. Biol. 28, 880–882.

Fleischer, A., Shafir, S., Mandelik, Y., 2013. A proactive approach for assessing alternative management programs for an invasive alien pollinator species. Ecol. Econ. 88, 126–132.

Gao, T., Nielsen, A.B., Hedblom, M., 2015. Reviewing the strength of evidence of biodiversity indicators for forest ecosystems in Europe. Ecol. Ind. 57, 420–434.

Giergiczny, M., Czajkowski, M., Żylicz, T., Angelstam, P., 2015. Choice experiment assessment of public preferences for forest structural attributes. Ecol. Econ. 119, 8–23.

Ginsberg, J.R., Milner-Gulland, E., 1994. Sex-biased harvesting and population dynamics in ungulates: implications for conservation and sustainable use. Conserv. Biol. 8, 157–166.

Gregory, R.D., Vořišek, P., Noble, D.G., Van Strien, A., Klvaňová, A., Eaton, M., Meyling, A.W.G., Joys, A., Foppen, R.P., Burfield, I.J., 2008. The generation and use of bird population indicators in Europe. Bird Conserv. Int. 18, S223–S244.

Haila, Y., Kouki, J., 1994. The phenomenon of biodiversity in conservation biology. Annales Zoologici Fennici 31, 5–18.

Hamilton, A.J., 2005. Species diversity or biodiversity? J. Environ. Manage. 75, 89–92.

Hanley, N., Czajkowski, M., 2019. The role of stated preference valuation methods in understanding choices and informing policy. Rev. Environ. Econ. Policy 13 (2), 248–266.

Hanley, N., Perrings, C., 2019. The economic value of biodiversity. Annu. Rev. Resour. Econ. 11, 355–375.

Heink, U., Kowarik, I., 2010. What criteria should be used to select biodiversity indicators? Biodivers. Conserv. 19 (13), 3769–3797.

Herruzo, A.C., Martínez-Jauregui, M., Carranza, J., Campos, P., 2016. Commercial income and capital of hunting: an application to forest estates in Andalucía. Forest Policy Econ. 69, 53–61.

Jacobsen, J.B., Boiesen, J.H., Thorsen, B.J., Strange, N., 2008. What's in a name? The use of quantitative measures versus 'Iconised' species when valuing biodiversity. Environ. Resour. Econ. 39, 247–263.

Johnston, R.J., Boyle, K.J., Adamowicz, W., Bennett, J., Brouwer, R., Cameron, T.A., Hanemann, W.M., Hanley, N., Ryan, M., Scarpa, R., 2017. Contemporary guidance for stated preference studies. J. Assoc. Environ. Resour. Econ. 4, 319–405.

Kilpatrick, A.M., Salkeld, D.J., Titcomb, G., Hahn, M.B., 2017. Conservation of biodiversity as a strategy for improving human health and well-being. Philos. Trans. R. Soc. B: Biol. Sci. 372, 20160131.

LaRiviere, J., Czajkowski, M., Hanley, N., Aanesen, M., Falk-Petersen, J., Tinch, D., 2014. The value of familiarity: effects of knowledge and objective signals on willingness to pay for a public good. J. Environ. Econ. Manage. 68 (2), 376–389.

Laurila-Pant, M., Lehikoinen, A., Uusitalo, L., Venesjärvi, R., 2015. How to value biodiversity in environmental management? Ecol. Ind. 55, 1–11.

Lundberg, A., 2010. Conflicts between perception and reality in the management of alien species in forest ecosystems: a Norwegian case study. Landscape Res. 35 (3), 319–338.

Mace, G.M., 2014. Whose conservation? Science 345, 1558–1560.

Mace, G.M., Barrett, M., Burgess, N.D., Cornell, S.E., Freeman, R., Grooten, M., Purvis, A., 2018. Aiming higher to bend the curve of biodiversity loss. Nat. Sustainability 1 (9), 448–451.

Mace, G.M., Norris, K., Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol. Evol. 27, 19–26.

Martínez-Jauregui, M., Delibes-Mateos, M., Arroyo, B., Soliño, M., 2020. Addressing social attitudes toward lethal control of wildlife in national parks. Conserv. Biol. 34 (4), 868–878. https://doi.org/10.1111/cobi.13468.

Martínez-Jauregui, M., Díaz, M., Sánchez de Ron, D., Soliño, M., 2016a. Plantation or natural recovery? Relative contribution of planted and natural pine forests to the maintenance of regional bird diversity along ecological gradients in Southern Europe. For. Ecol. Manage. 376, 183–192.

Martínez-Jauregui, M., Herruzo, A.C., Carranza, J., Torres-Porras, J., Campos, P., 2016b. Environmental Price of Game Animal Stocks. Human Dimensions of Wildlife 21 (1), 1–17. https://doi.org/10.1080/10871209.2016.1082682.

Martínez-Jauregui, M., Serra-Varela, M.J., Díaz, M., Soliño, M., 2018a. Mitigation strategies for conserving bird diversity under climate change scenarios in Europe: the role of forest naturalization. PLoS ONE 13, e0202009.

Martínez-Jauregui, M., Solino, M., Díaz, M., 2016c. Geographical variation in the contribution of planted and natural pine forests to the conservation of bird diversity. Divers. Distrib. 22, 1255–1265.

Martínez-Jauregui, M., Soliño, M., Martínez-Fernández, J., Touza, J., 2018b. Managing the early warning systems of invasive species of plants, birds, and mammals in natural and planted pine forests. Forests 9, 170.

Martínez-Jauregui, M., White, P., Touza, J., Soliño, M., 2019. Untangling perceptions around indicators for biodiversity conservation and ecosystem services. Ecosyst. Serv. 38, 100952.

Martín-López, B., Montes, C., Benayas, J., 2007. The non-economic motives behind the willingness to pay for biodiversity conservation. Biol. Conserv. 139, 67–82.

Meli, P., Rey-Benayas, J.M., Brancalion, P.H., 2019. Balancing land sharing and sparing approaches to promote forest and landscape restoration in agricultural landscapes: Land approaches for forest landscape restoration. Perspectives Ecol. Conserv. 17 (4), 201–205.

Meyerhoff, J., Mørkbak, M.R., Olsen, S.B., 2014. A meta-study investigating the sources of protest behaviour in stated preference surveys. Environ. Resour. Econ. 58 (1), 35–57.

Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Appl. 17, 2145–2151.

Nielsen, A.B., Olsen, S.B., Lundhede, T., 2007. An economic valuation of the recreational benefits associated with nature-based forest management practices. Landscape Urban Plann. 80, 63–71.

Nordén, A., Coria, J., Jönsson, A.M., Lagergren, F., Lehsten, V., 2017. Divergence in stakeholders' preferences: Evidence from a choice experiment on forest landscapes preferences in Sweden. Ecol. Econ. 132, 179–195.

Noss, R.F., 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conserv. Biol. 4, 355–364.

Nunes, P.A., van den Bergh, J.C., 2001. Economic valuation of biodiversity: sense or nonsense? Ecol. Econ. 39, 203–222.

Otto, S.A., Kadin, M., Casini, M., Torres, M.A., Blenckner, T., 2018. A quantitative framework for selecting and validating food web indicators. Ecol. Ind. 84, 619–631.

Oviedo, J.L., Yoo, H.I., 2017. A latent class nested logit model for rank-ordered data with application to cork oak reforestation. Environ. Resour. Econ. 68 (4), 1021–1051.

Pereira, H.M., Ferrier, S., Walters, M., Geller, G.N., Jongman, R., Scholes, R.J., Bruford, M.W., Brummitt, N., Butchart, S., Cardoso, A., 2013. Essential biodiversity variables. Science 339, 277–278.

Rai, P.K., Singh, J.S., 2020. Invasive alien plant species: their impact on environment, ecosystem services and human health. Ecol. Ind. 111, 106020.

Reed, D.H., Frankham, R., 2003. Correlation between fitness and genetic diversity. Conserv. Biol. 17, 230–237.

Ressurreição, A., Gibbons, J., Kaiser, M., Dentinho, T.P., Zarzycki, T., Bentley, C., Austen, M., Burdon, D., Atkins, J., Santos, R.S., 2012. Different cultures, different values: the role of cultural variation in public's WTP for marine species conservation. Biol. Conserv. 145, 148–159.

Robalino, J., Sandoval, C., Barton, D.N., Chacon, A., Pfaff, A., 2015. Evaluating interactions of forest conservation policies on avoided deforestation. PLoS ONE 10 (4), e0124910.

Rondeux, J., Sanchez, C., 2010. Review of indicators and field methods for monitoring biodiversity within national forest inventories. Core variable: deadwood. Environ. Monit. Assess. 164 (1–4), 617–630.

Scarpa, R., Thiene, M., Train, K., 2008. Utility in willingness to pay space: a tool to address confounding random scale effects in destination choice to the Alps. Am. J. Agric. Econ. 90 (4), 994–1010.

Serrada, R., Montero, G., Reque, J.A., 2008. Compendio de selvicultura aplicada en España (No. 634.95 C737). Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria, Madrid (España) Ministerio de Educación y Ciencia, Madrid (España).

Sharp, R.L., Larson, L.R., Green, G.T., 2011. Factors influencing public preferences for invasive alien species management. Biol. Conserv. 144 (8), 2097–2104.

Simberloff, D., 1998. Flagships, umbrellas, and keystones: is single-species management passé in the landscape era? Biol. Conserv. 83, 247–257.

Soliño, M., Alía, R., Agúndez, D., 2020. Citizens' preferences for research programs of genetic forest resources: a case applied to Pinus pinaster Ait. in Spain. Forest Policy Econ. 118, 102255.

Soliño, M., Oviedo, J.L., Caparrós, A., 2018. Are forest landowners ready for woody energy crops? Preferences for afforestation programs in Southern Spain. Energy Econ. 73, 239–247.

Tanentzap, A.J., Bazely, D.R., Williams, P.A., Hoogensen, G., 2009. A human security framework for the management of invasive nonindigenous plants. Invasive Plant Sci. Manage. 2 (2), 99–109.

Train, K., Weeks, M., 2005. Discrete choice models in preference space and willingness-to-pay space, in: Applications of Simulation Methods in Environmental and Resource Economics. Springer, pp. 1–16.

Turak, E., Brazill-Boast, J., Cooney, T., Drielsma, M., DelaCruz, J., Dunkerley, G., Fernandez, M., Ferrier, S., Gill, M., Jones, H., Koen, T., 2017. Using the essential biodiversity variables framework to measure biodiversity change at national scale. Biol. Conserv. 213, 264–271.

Varela, E., Verheyen, K., Valdés, A., Soliño, M., Jacobsen, J.B., De Smedt, P., Ehrmann, S., Gärtner, S., Górriz, E., Decocq, G., 2018. Promoting biodiversity values of small forest patches in agricultural landscapes: Ecological drivers and social demand. Sci. Total Environ. 619, 1319–1329.

White, P.C., Gregory, K.W., Lindley, P.J., Richards, G., 1997. Economic values of threatened mammals in Britain: a case study of the otter Lutra lutra and the water vole Arvicola terrestris. Biol. Conserv. 82, 345–354.

Zhao, M., Johnston, R.J., Schultz, E.T., 2013. What to value and how? Ecological indicator choices in stated preference valuation. Environ. Resour. Econ. 56, 3–25.