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RESEARCH ARTICLE

Prioritizing areas for ecological restoration: A participatory approach based on cost-effectiveness

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Abstract

1. Landscape-scale prioritization models are powerful decision-making tools in ecological restoration. Yet, they often fail to integrate multi-stakeholder perspectives and socio-ecological criteria.
2. We designed a new methodology to identify high-priority areas for landscape-scale restoration. This participatory cost-effectiveness analysis model is based on execution and maintenance costs and the potential increase in the supply of multiple ecosystem services.
3. We tested the model in a 181,000 ha heavily anthropized semi-arid landscape in southeastern Spain. Restoring the whole area would cost 221 million EUR and enhance the supply of ecosystem services by 39%. The cost-effectiveness of restoring pine forest and abandoned and irrigated crops were higher than restoring other Landscape Units. Restoring the least degraded sites was more cost-effective than the most degraded areas or randomly selecting sites, even when potential recovery was incomplete.
4. *Synthesis and applications*. The cost-effectiveness of restoration actions depends on the type of ecosystem and degradation state. Visualizing the outcomes of alternative restoration scenarios needs participatory prioritization maps based on financial costs and the potential supply of ecosystem services. We propose a participatory prioritization protocol that is flexible and adaptable and can help government agencies, environmental managers, investors, consultancies and NGOs' plan restoration actions at the landscape scale and optimize the effectiveness of restoration programs.

KEYWORDS
cost-effectiveness analysis, ecosystem services, land-use planning, landscape-scale restoration, participatory process, stakeholders, UN decade of ecological restoration

1 | INTRODUCTION

The growing human population and demand for natural resources have directly affected 75% of the Earth's terrestrial surface, threatening biodiversity and compromising ecosystem functions and services (hereafter ES; Butchart et al., 2010; Venter et al., 2016). Ecological restoration has the potential to protect biodiversity and balance the supply of ES (Rey Benayas et al., 2009), mitigate climate change (von Holle et al., 2020) and guide economic progress toward sustainability (Blignaut et al., 2014). This potential has been recognized internationally, underpinning continental and global restoration initiatives aiming to upscale ecological restoration (Aronson et al., 2020). Yet, the urgency of the environmental crisis calls for the efficient use of resources and the design of consensual prioritization strategies (Bodin et al., 2021).

Landscape-scale prioritization models reflect alternative approaches to assessing the effectiveness of restoration actions. These models have used multiple criteria to define priority areas for restoration, including disaster risk reduction (Vogler et al., 2015), past and future species distribution (Yoshioka et al., 2014), vulnerable ecosystems (Etter et al., 2020), multiple ES (Comín et al., 2018), landscape connectivity (García-Feced et al., 2011) and socioeconomic benefits (Torrubia et al., 2014). In addition, some prioritization models integrate land ownership, legal and logistical considerations (Holzmueller et al., 2011; Orsi & Geneletti, 2010), restoration costs and cost-effectiveness (Adame et al., 2015; Birch et al., 2010; Strassburg et al., 2020).

Cost-effectiveness analysis (CEA) allows the comparison and evaluation of projects with similar goals and avoid the economic valuation of natural resources (Gómez-Baggethum & Ruiz-Pérez, 2011; Robbin & Daniels, 2012). The choice of criteria for measuring the effectiveness of restoration varies from project to project. Habitat type, socio-ecological conditions and the interest of experts are the most common criteria (Kumar et al., 2021). Studies of cost-effectiveness have often considered a small number of objectives as a measure of effectiveness. Common goals include carbon sequestration, biodiversity (Strassburg et al., 2020), spatial connectivity (Molin et al., 2018) and potential hazards (Wada et al., 2017). Crucially, this leaves behind other services, like provision and cultural services, and may fail to respond to societal needs and aspirations. Social participatory processes may help to integrate multiple perspectives to solve these complex decisional processes (Ainscough et al., 2018).

Using prioritization models based on participatory CEA of restoration is essential to achieve the multiple objectives of the UN Decade of ecological restoration. They can help balance the protection of biodiversity and supply multiple ES, including carbon capture and fixation, helping with climate change adaptation and mitigation. Furthermore, shared diagnosis enables the incorporation of different types of knowledge, integrating multiple needs, reducing uncertainty and inequalities in access to ES and overcoming restoration barriers (Cortina-Segarra et al., 2021; Dufour & Piégay, 2009; Felipe-Lucia et al., 2015).

In this study, we established priority areas for ecological restoration using a CEA along a multi-criteria and multi-agent participatory process. Our model, which we applied to a large and complex landscape in southeast Spain, followed five steps: (i) establish societal preferences for ES, (ii) define a conceptual framework to obtain reliable estimations of cost-effectiveness of restoration actions, (iii) define priority areas for restoration based on the results of the CEA, (iv) analyse the relationship between cost and effectiveness and (v) measure the cost-effectiveness of different restoration scenarios. Our model can substantially improve restoration planning, increasing the transparency of the process and providing new insights into guide collective decisions.

2 | MATERIALS AND METHODS

2.1 | Study area

The study area is the Crevillent Forest Management Unit (CFMU), a 224,000 ha operational unit managed by the Regional Government of Valencia in southeast Spain (38°13′21″N, 11°11′14″W, Figure 1). The climate is Mediterranean arid and semi-arid, with mean annual precipitation of 307 mm, and temperature ranging between 12 and 23°C. The area includes a wide range of natural and semi-natural ecosystems covering 181,000 ha: pine forests dominated by *Pinus halepensis* Mill. plantations and small patches of *Quercus rotundifolia* Lam., shrublands and steppes–dominated by *Stipa tenacissima* L., sand dunes, rivers, wetlands and abandoned, rainfed and irrigated crops. Semi-natural ecosystems have high ecological value, with 18% of the territory protected in 19 Natura 2000 sites. Significant drivers of degradation are agricultural intensification and abandonment of crops, soil salinization, overexploitation of aquifers and increasing urbanization and tourism. In CFMU, we identified eight landscape units (LU) based on Land Use/Land Cover maps (IGN, 2014) corresponding to the ecosystems named above. Our study did not integrate urban areas, quarries, watercourses and continental waters because their restoration is regulated by specific legislation.

2.2 | Societal demand for ecosystem services

Through a participatory process, we identified a wide diversity of groups of interest with the primary goal of integrating as many perspectives and demands for ES as possible. We included government, political parties, non-governmental environmental and social organizations (NGOs), scientists, farmers and mining, tourism and service sector representatives. To identify stakeholders, we adopted the chain referral method for intentional sampling (as in Bautista et al., 2017), selecting a preliminary group of stakeholders who provided further contacts until each interest group was saturated (i.e. when stakeholders proposed no new name). When suitable, we prioritized contact with representatives of organizations (e.g. leaders of NGOs, CEOs, high-ranking managers, etc.) as they commonly feel/

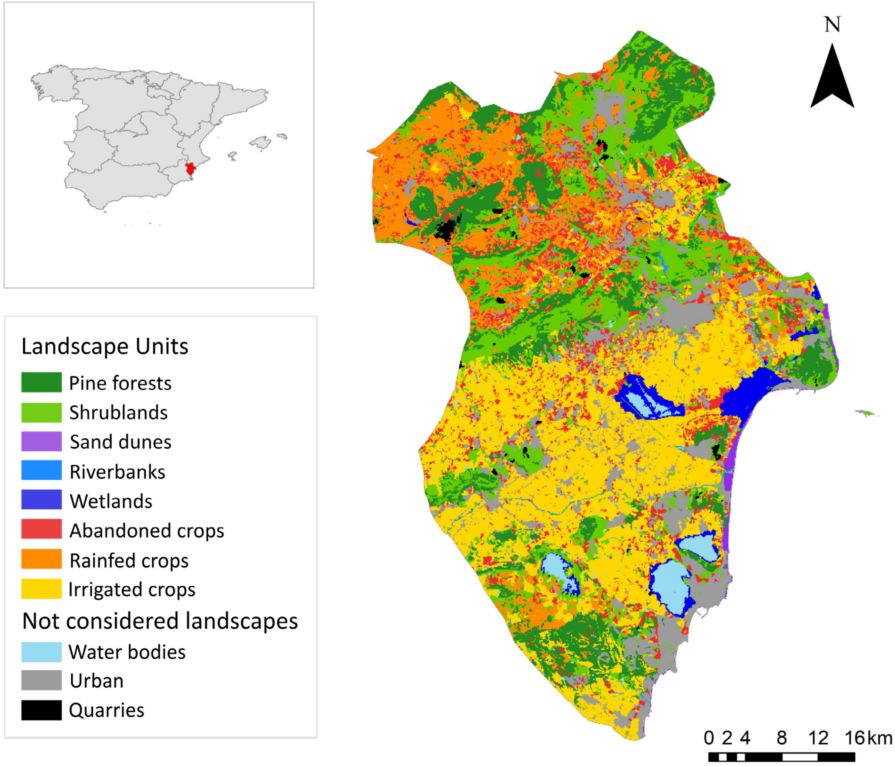

FIGURE 1 Distribution of the Landscape Units in the Crevillent Forest Management Unit (southeast Spain). The inset map shows its location within Spain. Water bodies, urban matrix and quarries were not considered in this study.

are responsible for representing the common ideas of their group, and they are more likely to collaborate in participatory processes.

The participatory process followed three phases (Figure 2). In the first phase, we conducted semi-structured personal interviews to generate lists of ES provided by each LU. We informed stakeholders of their anonymity and counted with their verbal consent for participation in the project (see Table S1). We used multiple stimuli, such as photos of landscapes and ES examples, to illustrate the concepts, obtain more information and foster the thinking of stakeholders and focus. Stimuli were carefully selected to elicit interviewees' responses and minimize influences on them (see Figure S1), avoiding differences in colour, quality and composition of the images. We completed the stakeholders' profiles by asking about their educational level, age and gender. Later, we conducted a qualitative analysis of the concepts of ES expressed by the stakeholders and classified them into four groups (Sections), following the Common International Classification for Ecosystem Services, CICES (Haines-Young & Potschin-Young, 2018).

In the second phase, we asked the stakeholders to rank ES in decreasing order of relative importance. For this, we divided ES into corresponding Sections to facilitate comparing them. Then, we asked stakeholders to propose an ordinal classification of ES within and between each Section using Qualtrics (Qualtrics, 2016). The total number of ES within each Section ranged from four to seven. For simplicity, ranking did not allow ES and ES Sections to have equal importance.

The standardization of the order of importance of each ES allows them to be comparable, transforming their values into a continuous numeric dimensionless scale between 0 and 1. We used a linear scale transformation as a standardization procedure: we applied the sum method, obtaining the standardized weight, in which each value is divided by the sum of values for each ES within and between each Section (Chakraborty, 2007). We multiplied the Section weight value by the weight obtained for each ES within each Section to obtain the final standardized weights across sections. Due to unbalanced Sections in terms of ES, we divided the weight by the ratio between the number of ES in each Section and the total number of ES (Saaty, 1994).

2.3 | Reference sites and effectiveness of restoration actions

We quantified and mapped the ES identified by the stakeholders using spatial multi-criteria analysis of biophysical and social indicators with a 100 m resolution (see Table S2) using ArcGIS 10.4 (ESRI, 2016). Maps were standardized into a 0–1 scale due to the multi-dimensionality of ES through a Fuzzy linear membership tool in ArcGIS, assigning 1 to the highest supply of ES within the CFMU (Koschke et al., 2012). We compiled spatial indicators from the most current databases and spatial information systems available When missing, we built ES raster, creating new indicators and combining

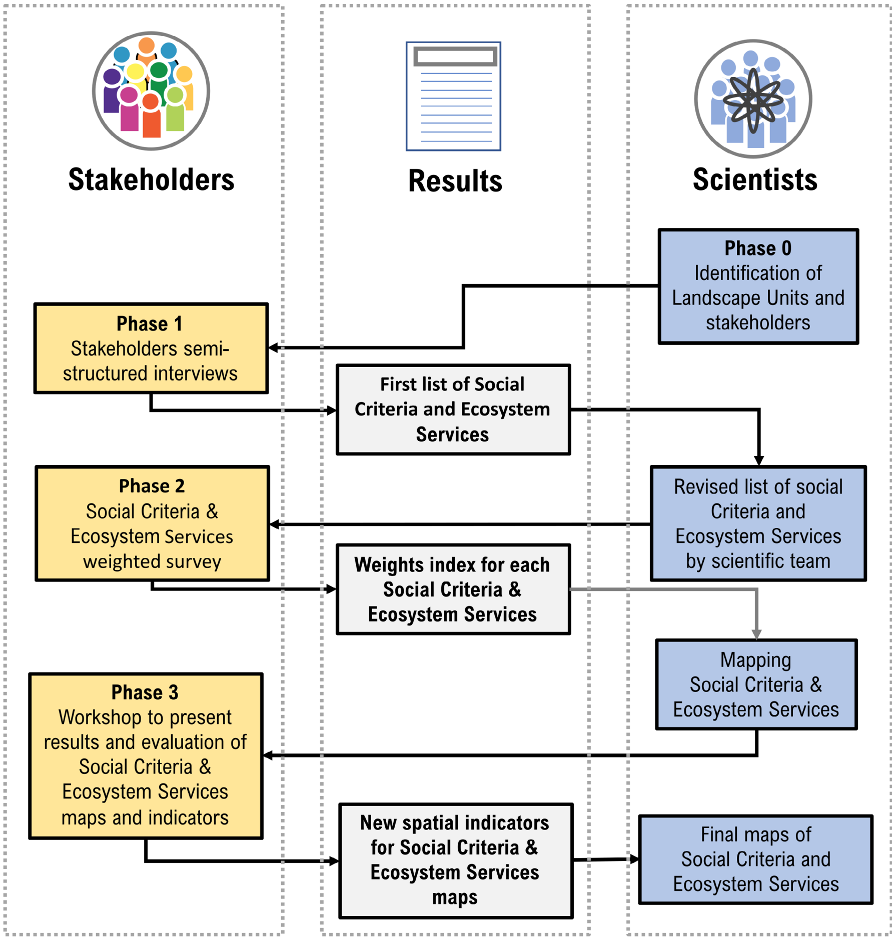

FIGURE 2 Phases of the participatory process. Yellow squares correspond to phases where stakeholders were involved; blue squares correspond to phases where only scientists were involved, and grey squares indicate results obtained in each phase of the process.

existing cartography. In addition, we chose indicators that presented the finer spatial resolution when possible.

In the third phase of the participatory process, we presented eight ES maps to the stakeholders for re-evaluation in a workshop. The project team previously selected these maps through internal voting, considering the chance of misinterpreting stakeholders' statements and the level of uncertainty of the chosen spatial indicator (Table S2). Stakeholders suggested modifications to better respond to their criteria and pointed out not considered areas that we later incorporated. The whole participatory process was completed between January 2016 and July 2018.

To estimate restoration effectiveness, we identified reference sites. Undisturbed sites are largely missing in highly anthropized ecosystems, as in the case of CFMU (Ruiz & Sanz-Sánchez, 2020). Thus, new functional and achievable reference site must be assigned (Gann et al., 2019; Perring et al., 2013). Because of this, we selected the reference sites as pixels that provided the highest levels of ES supply and similar proportions of ES as stakeholders demanded. For this, we used a Weighted Sum Similarity Index (WSSI) to estimate the restoration effectiveness for each LU. We first calculated the weighted sum of all ES, multiplying each ES layer by the specific weight obtained in the participatory process. Then, we multiplied this value by one minus the Euclidean distance between the weight assigned by the stakeholder group to each ES and the actual proportional contribution of each ES to the sum of ES in each pixel. The inverse distance informs the similarity between the proportion of the different ES supplied in each pixel and stakeholders' relative demands for the other ES, assigning higher weights to pixels whose proportional supply of different ES is closer to stakeholders' demands. Thus, Equation (1):

$$\mathrm{WSSI}_{ij} = \left(\Sigma W_i \times \mathrm{ES}_{ij}\right) \times 1 - \sqrt{\Sigma\left(W_i - \mathrm{ES}'_{ij}\right)}, \quad (1)$$

where i represents each ES, j each pixel, W the weights obtained in the participatory process and ES′ is the relative proportion of the current supply of each ES (i.e. the current supply of a specific ES divided by the sum of all ES supplied by pixel j). This aggregate method allows us to integrate multiple ES into a single value of restoration effectiveness and analyse potential trade-offs and synergies between ES. Then, we identified pixels corresponding to the reference site within each LU as the mean of pixels above the 95th percentile of WSS values (Schröter & Remme, 2016). By assuming the reference sites were located within the study area, we increased the probability that degraded sites share the same abiotic and biotic context, as well as a common land-use history. Conversely, if the whole study area were degraded, compared with similar areas beyond its limits, the selected reference sites would indeed be degraded, a possibility that we cannot exclude.

We estimated the effectiveness as the potential increase in WSSI, measured as the difference between the mean WSSI of reference sites and the WSSI of a pixel. We, therefore, assumed that: (i) the reference sites exist and can be identified within the CFMU, (ii) through ecological restoration, the reference sites states can be attained and (iii) restoration allowed no shifts between LU, except for abandoned crops, whose reference was shrublands with the highest WSSI values. This assumption is realistic and conservative as the transformation from shrublands to forests in southeast Spain is unlikely under current and future climatic conditions (Felicísimo et al., 2011), and restoring forest areas is a management priority in the region. Yet, we cannot exclude the possibility of further land-use changes (e.g. abandonment of crops). Thus, Equation (2):

$$\text{Effectiveness}_{ij} = \mu\text{WSSI}_{95\%i} - \text{WSSI}_{ij}, \tag{2}$$

where i represent each LU, j represents each pixel and $\mu\text{WSSI}_{95\%}$ represents the mean of the 95% percentile of WSSI for each LU.

2.4 | Costs of restoration and cost-effectiveness analysis

We estimated the restoration cost per hectare as the sum of the execution and maintenance costs (Iftekhar & Polyyakow, 2021). For execution costs, we used budgets from recent projects done by the government of Spain over the last years in the area (see Table S3). We calculated the maintenance costs as 5% for terrestrial and 2.5% of execution costs for wetlands and riverbank ecosystems per year, from the second year after restoration, by recovery time (maximum time 20 years, see de Groot et al., 2013). We considered execution costs and recovery time to decrease linearly with the degradation state, which agrees with published studies (Adame et al., 2015; Holl & Aide, 2011). Thus, total costs were estimated following Equation (3):

$$\text{Total Cost}_{ij} = \text{Ecost}_i \times \text{DS}_j \times \left(1 + \text{Mcost}_i \times t \times \text{DS}_j\right), \tag{3}$$

where i corresponds to each LU, j represents each pixel, DS refers to the degradation state, Ecost and Mcost refer to the execution and maintenance costs, respectively, and t refers to the recovery time. Other costs, like opportunity and acquisition costs, were not considered because, in this study, we did not contemplate the shifts from productive to unproductive land uses.

Regulating and Maintenance ES are directly related to biodiversity and ecosystem functioning (Felipe-Lucia et al., 2015), have been degraded globally and are the major targets of ecosystem restoration projects (Santos-Martín et al., 2013). For this, we estimated the degradation state as the normalized (0–1) unweighted sum of these services for each LU. Thus, Equation (4):

$$\text{DS}_i = \left(\text{Max}\left(\Sigma\text{ESmr}_{ij}\right) - \Sigma\text{ESmr}_{ij}\right) / \left(\text{Max}\left(\Sigma\text{ESmr}_{ij}\right) - \text{Min}\left(\Sigma\text{ESmr}_{ij}\right)\right) \tag{4}$$

where i corresponds to each LU, j represents each pixel, DS refers to the degradation state and ESmr refers to Regulation and Maintenance ES. Finally, cost-effectiveness was calculated as the ratio between the effectiveness and total restoration cost. We mapped cost-effectiveness in the CFMU and defined priority areas for restoration as those with the highest values per LU.

2.5 | Effectiveness versus cost

We compared the effectiveness and total cost of restoration in each LU. We used ES data to calculate both effectiveness and cost. To avoid possible spurious correlations, we tested the significance of the effect size of the relationship by performing a randomization analysis (Brett, 2004). We run 100 swapping randomizations of each ES within the corresponding LU, keeping the total amount of each ES constant. From each randomization, we estimated the Standardized Effect Size of the effectiveness (SES_{eff}) as the difference between actual effectiveness and the mean of simulated effectiveness, divided by the standard deviation of simulated effectiveness (Gotelli, 2000; see also Soliveres et al., 2015 for similar approaches). We fitted linear models between SES_{eff} and restoration costs by LU to estimate their slope, intercept and significance. We used R for all the analyses (R Core Team, 2020).

2.6 | Restoration scenarios

Many countries are committed to restoring 15% of degraded ecosystems (CBD, 2004). Based on the 15% target, we used our cost-effectiveness model to explore the impact of different restoration scenarios on costs, cost-effectiveness and total gain of ES Sections (Provision, Regulation and Maintenance and Cultural services). For this, we defined 12 scenarios. First, we selected critical areas for restoration for each LU by assigning the highest priority to 15% of either the lowest levels of degradation (LOW), the highest levels of degradation (HIGH) or a random selection of pixels (RAN) (Kotiaho et al., 2016). For each of these three scenarios, we calculated costs and effectiveness for four levels of recovery: complete recovery (100% effectiveness) and partial recovery (75%, 50% and 25% effectiveness; Jakovac et al., 2020). Calculations were performed in R, and maps were prepared with ArcGIS.

3 | RESULTS

3.1 | Societal demands for ecosystem services

The stakeholder group consisted of 124 members. They were biased toward males (72%), with an age of 49 ± 10 years (mean ± standard deviation). Members of the government, together with scientists, were the most represented groups (see Figure S2). Stakeholders selected 24 ES, seven Provisioning services, seven Regulation services, four Maintenance services and six Cultural services (Figure 3a). They favoured the supply of Regulation, Maintenance and Provision over Cultural services. The five most valued ES were habitat quality, carbon sequestration, water retention, temperature regulation and agricultural products, representing 34% of the total weight. In contrast, Cultural services like recreational activities, outdoor sports and hunting and fisheries were less valued. Preferences differed slightly between professional groups (Figure 3b), but cost-effectiveness maps built using the ES weights assigned by each group were similar (Spearman coefficients between 0.70 and 0.99).

3.2 | Restoration costs and cost-effectiveness

Restoring the entire CFMU could increase the supply of ES by 39% at a total cost of 220 million EUR, or 1216±962 EUR/ha (Table 1, Figure S3 and S4). The cost-effectiveness of restoring pine forests and abandoned and irrigated crops was higher than restoring other LU, increasing the ES supply per LU by 51%, 42% and 39%, respectively. On average, restoring these LU was less expensive than other LU (pine forest: 607±200 EUR/ha; irrigated crops: 650±283 EUR/ha and abandoned crops: 677±270 EUR/ha). Conversely, sand dunes, wetlands and riverbanks were more costly to restore (3697±1915, 3265±1042 and 3125±1435 EUR/ha, respectively), and their cost-effectiveness per hectare was lower (Table 1). The cost of restoring shrublands and rainfed crops was intermediate (1792±576 and 2227±1062 EUR/ha, respectively).

Differences in cost-effectiveness between and within the different LU created a complex spatial mosaic (Figure 4). We identified patches with high cost-effectiveness that may be considered priority areas for restoration. For example, the cost-effectiveness of restoring pine forests, shrublands, wetlands, and abandoned and irrigated crops in Santa Pola Cape, the easternmost point of the CFMU (Figure 4a), was high. In addition, the restoration of pine forests and rainfed and irrigated crops in the northwest showed high cost-effectiveness. In contrast, cost-effectiveness was low around highly populated areas.

The effectiveness of restoration actions (as estimated with the SES_{eff}) was positively related to restoration costs and degradation state (Figure 4b). These relationships were linear in all cases, and their intercept and slope depended on LU, which generated substantial variations in cost-effectiveness. Slopes in pine forests and abandoned and irrigated crops were highest (0.004, 0.003 and 0.002, respectively; Figure 4 and Figure S5), meaning that investments in these LU may generate the highest increases in the supply of ES per hectare. Conversely, riverbanks showed the lowest slopes (slope 0.0002).

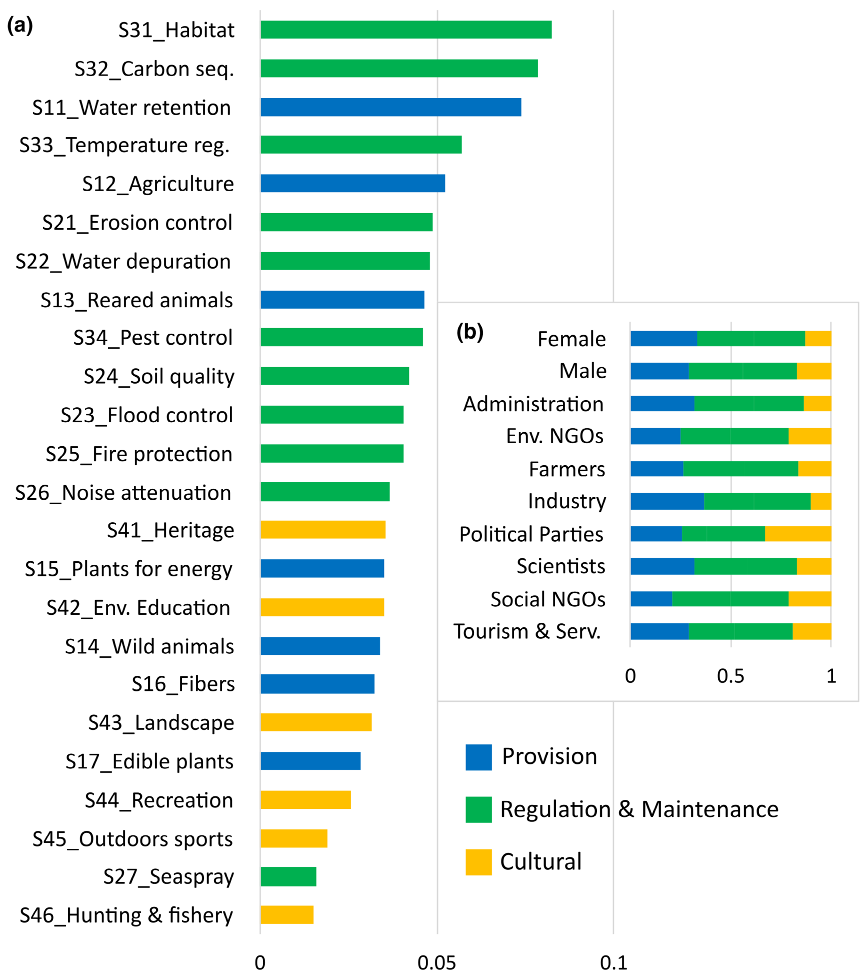

FIGURE 3 Weights of ecosystem services (ES) assigned by the stakeholder's group (a) and weights of the different ES sections dependent on gender and professional sector (b).

TABLE 1 Proportional increment in the supply of ecosystem services (ES), total costs and average cost-effectiveness (CE) per pixel of restoration actions in the Crevillent Forest Management Unit. Means and standard deviations (± SD) are shown.

Landscape units	Surface area (ha)	Increment of ES (%)	Total costs (million EUR)	CE/ha × million EUR^{-1} (relative units)
Pine forests	29,809	42	18.08	191 (±60)
Shrublands	35,562	38	63.74	54 (±20)
Sand dunes	932	40	3.44	32 (±39)
Riverbanks	657	39	2.05	33 (±19)
Wetlands	4739	41	15.47	25 (±6)
Abandoned crops	20,140	51	13.63	179 (±53)
Rainfed crops	29,165	31	64.95	40 (±16)
Irrigated crops	60,475	39	39.32	143 (±51)

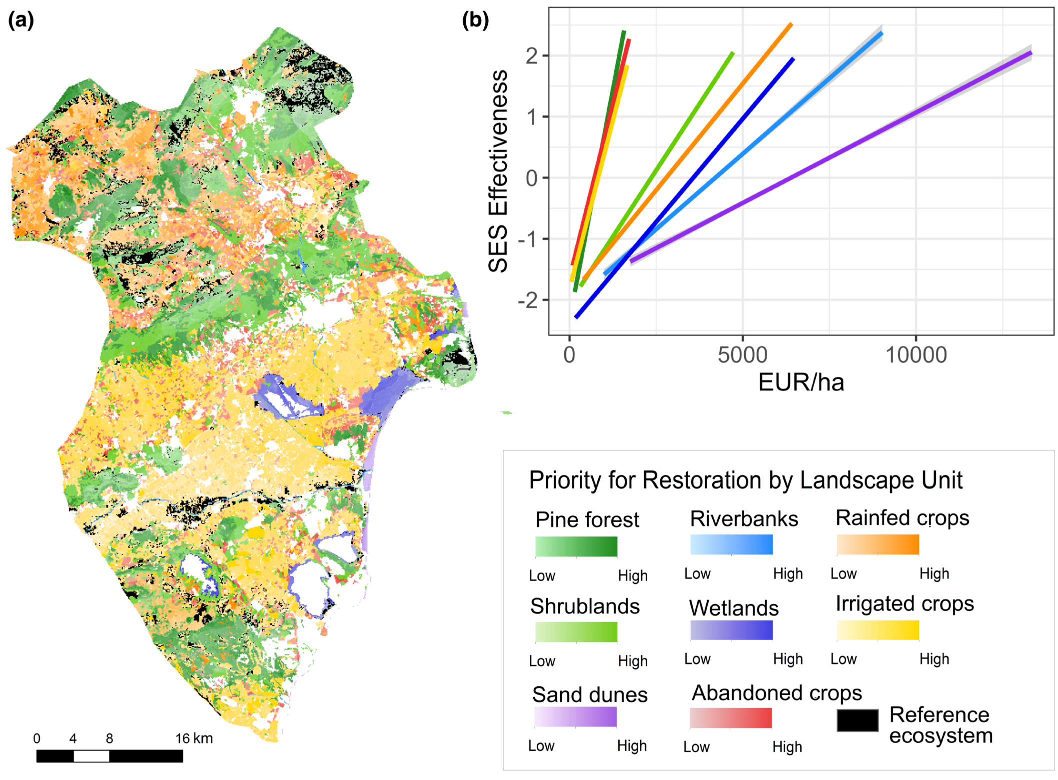

FIGURE 4 Prioritization of restoration areas by Landscape Units following a cost-effectiveness analysis in the Crevillent Forest Management Unit (a). Darker areas correspond to higher values of cost-effectiveness. Grey areas were excluded from the analysis. The relationship between standardized effect size of restoration effectiveness (SES effectiveness) and cost per ha is shown in (b), using the same colours to identify Landscape Units. Grey shadows correspond to 95% confidence intervals. All regressions were significant at $p < 0.001$.

3.3 | Restoration scenarios

The effectiveness and costs of different restoration scenarios were substantially different (Table 2). Restoring 15% of the less degraded areas (LOW scenario) required a 15.45 million EUR investment and increased the supply of ES by 18%. If 15% of the territory were randomly restored (RAN scenario), costs would almost double (33.08 million EUR), and ES supply would increase to 40%. By targeting the 15% most degraded areas (HIGH scenario), costs would attain 55.06 million EUR and the potential increase in the supply of ES would reach 85% (Table 2). Cost-effectiveness was highest in the LOW scenario, followed by RAN and HIGH. This pattern was maintained when complete recovery was not achieved. When only a 25% recovery was possible, the expected increment in ES supply would be 5%, 10% and 21% in the LOW, RAN and HIGH scenarios, respectively.

In terms of the impact of restoration actions on the different ES Sections, Regulation and Maintenance services showed the highest increase under all scenarios but the LOW (Figure 5). Thus, when restoration actions targeted relatively undegraded areas, reference sites selected by using stakeholders criteria supplied fewer Regulation and Maintenance services than areas showing a slightly smaller integrated and balanced provision of all ES. The HIGH scenario showed the highest increment of all ES Sections.

4 | DISCUSSION

We developed a prioritization model for landscape-scale restoration based on financial costs and the effectiveness of restoration actions, integrating stakeholder demands for ES in a heterogeneous and highly anthropized landscape.

TABLE 2 Proportional increment in the supply of ecosystem services (ES), total costs and cost-effectiveness (CE) of restoration actions in the Crevillent Forest Management Unit under a combination of three strategies to restore 15% of the area considering the degradation state (LOW for less degraded areas, HIGH for highly degraded areas, and RAN for a random selection of pixels) and four levels of potential recovery (25%–100%).

Recovery level	Scenario	Increment in ES (%)	CE × million EUR^{-1}
100%	LOW	18	96
100%	RAN	40	83
100%	HIGH	85	80
75%	LOW	14	72
75%	RAN	30	62
75%	HIGH	64	60
50%	LOW	9	48
50%	RAN	20	41
50%	HIGH	42	40
25%	LOW	5	24
25%	RAN	10	21
25%	HIGH	21	20

4.1 | Social demand and supply of ecosystem services

Stakeholders identified a long list of ES, covering all ES Sections defined in the Common International Classification CICE. This long list is consistent with the vast diversity of social profiles involved in the participatory process and previous studies (Derak et al., 2016). Growing global concerns about climate change and biodiversity translated into a high value given to some services like habitat quality, carbon sequestration and temperature regulation. Surprisingly, other components of climate change received relatively low scores. Wildfires are of major concern in the Mediterranean region, but large extents of the CFMU are too arid to burn (Aragonses & Chuvieco, 2021). The low rate of wildfires in the area may explain stakeholders' low importance on fire protection. On the other hand, the lack of awareness of the relationship between Regulation and Provision services may be related to the low value given to flood control, despite recent and historical records of catastrophic floods in the region (Pérez-Morales et al., 2018) and the shared belief that deforestation is responsible for increased runoff (Calder et al., 2007). Provisioning services, mainly water supply, agriculture and reared animals, are usually highly valued, as they are tangible and easy to identify by the stakeholders, particularly in drylands (Castro et al., 2011).

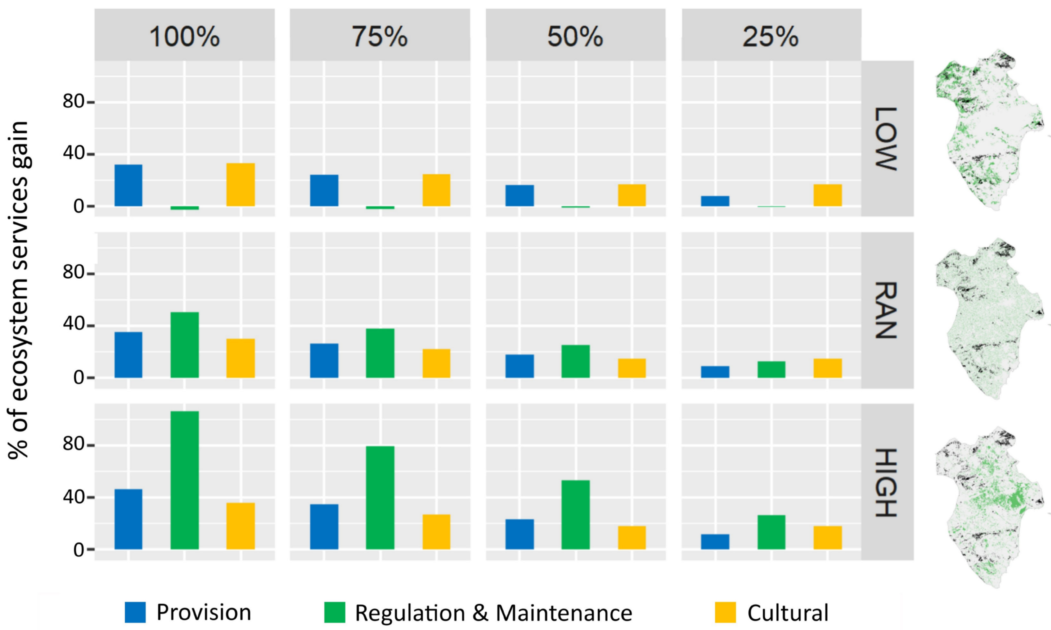

FIGURE 5 Total gain of ecosystem services Sections under 12 scenarios of prioritization. These are combinations of three strategies to restore 15% of the area considering the degradation state (LOW for less degraded areas, HIGH for highly degraded areas, and RAN for a random selection of pixels) and four levels of potential recovery (25%–100%). On the right side of the figure, areas selected for restoration are shown in green, and reference sites are shown in black.

4.2 | Restoration costs

The derived cost of restoration per hectare (≈1200 EUR/ha) felt within the costs estimated by the European Union for implementing the Biodiversity Strategy (between 800 and 5000 EUR/ha; Tucker et al., 2013). In some cases, our estimates were lower than those reported in other studies (Cuenca et al., 2016). Yet, we must note that our cost data comes from restoration projects implemented in southeast Spain, combining site-specific active restoration, and thus may be closer to actual costs. Restoring the whole CFMU would cost 221 million EUR. This investment parallels the highest funding allocated by Spanish regional governments to forest restoration (Cuenca et al., 2016). Although restoration costs are often lower than degradation costs (Nkonya et al., 2011), it may be unfeasible to allocate this amount of funding to this area in the short term, emphasizing the need to optimize costs through prioritization strategies (Molin et al., 2018).

4.3 | Potential increase in the supply of ecosystem services

Restoring the whole CFMU would increase the supply of ES by 39%. This estimate may be conservative due to the assumptions of the model. We defined reference sites as areas with the largest supply of ES within each LU. In contrast, we cannot disregard the possibility that reference sites could be located beyond the CFMU, where the supply of ES may be higher. In this case, the absolute increases in the supply of ES would change for a given LU, but their spatial distribution would be the same. In addition, we assumed that degraded areas could be restored with a partial or total recovery of the supply of ES, whereas degradation thresholds may have been crossed (Berdugo et al., 2020), making restoration unfeasible or too costly. as it does not consider structural degradation involving, for example, replacements of forests by shrublands or any other shift between LU.

4.4 | Cost-effectiveness and restoration scenarios

We found substantial differences in the cost-effectiveness of restoring different LU. The cost-effectiveness of restoring pine forests was the highest. Yet, restoring abandoned crops to shrublands would increase the supply of ES at a higher level (51% and 42%, respectively) and enhance resilience to future climatic conditions (Peñuelas et al., 2017). Cost-effectiveness and its components, as well as the impact of restoration actions on the different ES, and other criteria that were not considered in our study, such as connectivity for fauna and flora, the potential to create jobs and cultural priorities, to mention a few, must be regarded as complementary decision-making tools that to build consensus on the most suitable restoration options (Silva et al., 2021).

Our model predicts that restoring less degraded areas leads to more cost-effective options when recovery is partial or total. These results support the need to prioritize areas that are not severely degraded and still deliver a relatively high level of ES (Kotiaho & Moilanen, 2015). The same approach has been recommended to assist the Members States of the European Union (Lammerant et al., 2013) and has been integrated into European-wide restoration prioritization models (Liquete et al., 2015).

The supply of different ES sections varied between restoration scenarios. Regulation and Maintenance ES showed the highest increase in almost all scenarios, which agrees with Rey Benayas et al. (2009). Contrastingly, under the LOW scenario, Regulation and Maintenance ES decreased after restoration, as stakeholders weighted Provisioning services higher, indicating how trade-offs can arise in restoration scenarios based on stakeholder preferences (Turkelboom et al., 2018).

Our model is flexible and can be replicated in other areas after selecting the stakeholder group that is specific to the site and defining and weighing relevant ES, given the socio-ecological context. For example, we applied the same participatory cost-effectiveness methodology in Enguera Management Unit (south Valencia, Spain), a dry Mediterranean sub-humid area dominated by pine forests and severely affected by wildfires. Prioritization maps based on the ES selected and weighted by the Enguera stakeholder group were later used by the Regional Government of Valencia to design a restoration project in the area. The stakeholder group was consulted to produce the final version of the project (García-Pereira et al., 2020).

Applying the participatory cost-effectiveness model in contrasting landscapes such as Enguera and Crevillent Management Units has been beneficial in identifying caveats and ways to face them. Thus, the composition of the stakeholder group could be enlarged to integrate underrepresented groups, like youth, females and citizens with lower education levels. This bias reflects unbalanced societal power roles (Fondas, 2000) and can affect ES selection and weighting (Butler & Adamowski, 2015). It is important to note that our approach is built on the opinion of current leaders but may not reflect the needs and aspirations of the whole community. Beyond desirable societal changes, further efforts to integrate under-represented sectors are needed, forcing stakeholder sampling to build diverse and equally distributed groups. This can be achieved by selecting specific profiles as crucial informants to eliminate bias. In addition, stakeholders' long-term engagement through the participatory process could be improved by reducing the number of surveys, making ES databases available in advance and agreeing with stakeholders on modelling from the beginning. Furthermore, developing dynamic modelling tools showing the possible outcomes of restoration scenarios in real-time could make results more understandable and enhance the participation of stakeholders in higher levels of the decision-making process (Green et al., 2019; Hoftman et al., 2022). Finally, further studies should help quantify ecosystem restorability, identify restoration thresholds, and integrate climatic and land-use changes, thus enriching our assumptions on ecosystem dynamics and their impact on the supply of ES.

5 | CONCLUSIONS

We show here that a prioritization strategy based on participatory CEA of ecological restoration actions can maximize the outcomes of investments while responding to societal preferences and needs. Cost-effectiveness differs substantially between different LU and is higher when restoration prioritizes less degraded areas. Our approach can be used to visualize the potential outcomes of large-scale restoration programs and assess restoration in different degradation scenarios. In addition, this model can guide economically efficient management under different ecological and socioeconomic contexts. Far from purely theoretical, our tool has already proven useful as a decision-making tool for restoration planning in two regions in Spain.

AUTHOR CONTRIBUTIONS

Elysa Silva, Jordi Cortina-Segarra, Andreu Bonet, Germán López and Antonio Aledo conceived the ideas and designed the methodology together with Pietro Salvaneschi and Walid Naji. Elysa Silva, Pietro Salvaneschi, Emilio Climent-Gil, Mchich Derak and Walid Naji collected and analysed the data. Elysa Silva, Jordi Cortina-Segarra and Andreu Bonet wrote 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 declare that they have no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are available via Figshare Digital Repository https://doi.org/10.6084/m9.figshare.22233238.v1 (Silva et al., 2023).

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SUPPORTING INFORMATION

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

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