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Lessons learned from an interdisciplinary evaluation of long-term restoration outcomes on 37 restored coastal grasslands in California

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ABSTRACT

Governmental and non-governmental organizations spend considerable funding on restoring ecosystems to counter biodiversity loss, yet outcomes are often not assessed at a regional scale. Monitoring is done ≤5 years after project-implementation, if at all, and rarely assesses the effects of management practices on project success. We combined vegetation surveys and management interviews to compare long-term restoration outcomes of 37 California coastal grassland projects (5–33 y post-implementation) that spanned a 1000-km north-south gradient. We found that coastal grassland restoration is largely successful at reaching project goals (95 %) and a standard performance metric (80 %) to restore native cover, but land managers preferentially use a small number of well-tested, "high success" species, potentially at the expense of regional diversity. Medium and high maintenance intensity resulted in lower non-native cover and improved native cover and rarefied native richness. Managers of voluntary (non-statutory) sites were more open to assessing outcomes and spent less per hectare compared to legally mandated (statutory) projects but achieved similar plant cover and even higher rarefied richness. Statutory project managers indicated that regulatory agencies sometimes lowered compliance goals for native cover if the initial targets were not met. Additional funding for greater maintenance intensity and incorporating more locally distinctive species (*i.e.*, endemic or range-restricted) may help counteract potential unintended consequences from preferential plant selection, and inter-agency coordination of species selection could reduce biotic homogenization. We recommend delegating funds to a third-party monitoring group to ensure legally mandated compliance and consistency in assessment.

1. Introduction

Governments, conservationists and land managers make large expenditures to restore ecosystems (BenDor et al., 2015; Bernhardt et al., 2005; Menz et al., 2013) but outcomes vary greatly, and projects are seldom monitored after implementation (Bernhardt et al., 2005; Li et al., 2019). Project assessment is important to ensure goals are reached, adaptive management applied, and successful practices identified (Dickens and Suding, 2013; Mönkkönen et al., 2009). Project assessment of restoration outcomes typically only occur for legally-mandated (statutory) projects over the short-term (≤5 years), and rarely compare multiple sites (Bernhardt et al., 2005; Wyżga et al., 2021). Yet restoration project evaluation at a regional scale can help elucidate the effects of management on outcomes that cannot be observed at a single site (Bernhardt et al., 2005; Holl et al., 2022), and long-term data are required to develop strategies for adaptive management. For example, Matthews and Spyreas (2010) found initial recovery after wetland restoration but in later years found the plant community became homogenized by reinvasion.

Biotic homogenization across multiple levels of diversity after ecological restoration is a growing concern (Holl et al., 2022; Matthews and Spyreas, 2010; Zhang et al., 2022). Restoration practitioners make intentional choices during plant selection to maximize success and minimize risk (Lesage et al., 2020), however, these choices may have unintended consequences. For example, Lesage et al. (2018), found that practitioners tend to use perennial species that are more likely to persist over multiple years, resulting in loss of annual species diversity. Similarly, Talal and Santelmann (2020) found that land managers sometimes have multiple goals related to aesthetics and human safety that may exclude the use of certain native species to ensure all goals are met.

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Furthermore, biotic homogenization can be compounded by changing climatic conditions and land uses that promote biological invasions and fast-growing species (Holl et al., 2022; Matthews and Spyreas, 2010).

Restoration management decisions affect ecological outcomes (Burnett et al., 2019; Guiden et al., 2021; Lesage et al., 2018) but are often not considered (Cabin et al., 2010; Dickens and Suding, 2013) simultaneously with ecological data (Bernhardt et al., 2005; Wyżga et al., 2021). These management decisions are influenced by individual management ideologies, project-based goals, local habitat conditions, and legal requirements (Cabin, 2007; Hagger et al., 2017; Kull et al., 2015). For example, risk averse land managers may avoid species that grow slowly or have low survival due to a desire for achieving timebound project goals (Lesage et al., 2018). In addition, propagation methods often are not documented for the vast diversity of species that could be used for restoration (Bartholomew et al., 2022; Ladouceur et al., 2018). Integrating management perspectives and local ecological knowledge can improve understanding of restoration outcomes by providing context and justification for the use of certain species and management choices (Bernhardt et al., 2005; Cabin, 2007; Wagner and Davis, 2003).

We assessed restoration outcomes against project goals and a standard performance metric for 37 coastal grassland restoration projects across a 1000-km span in California, USA (Fig. 1; Barbour et al., 2007) to answer (1) whether restoration projects are meeting site-level targets for native cover and richness; and (2) how successful projects are in restoring plant diversity at a regional scale. We combined vegetation surveys, document analysis, and interviews with land managers to (3) determine which ecological, financial, and management factors most strongly affected (1) and (2). We hypothesized that most projects would not achieve ecological targets due to strong competition from non-native species (Matthews and Spyreas, 2010; Pearson et al., 2016) and a lack of funding for ongoing site management. At the regional scale, we anticipated that land managers would preferentially use a subset of species that have been demonstrated to establish well in many projects due to concerns about regulatory compliance (Lesage et al., 2018).

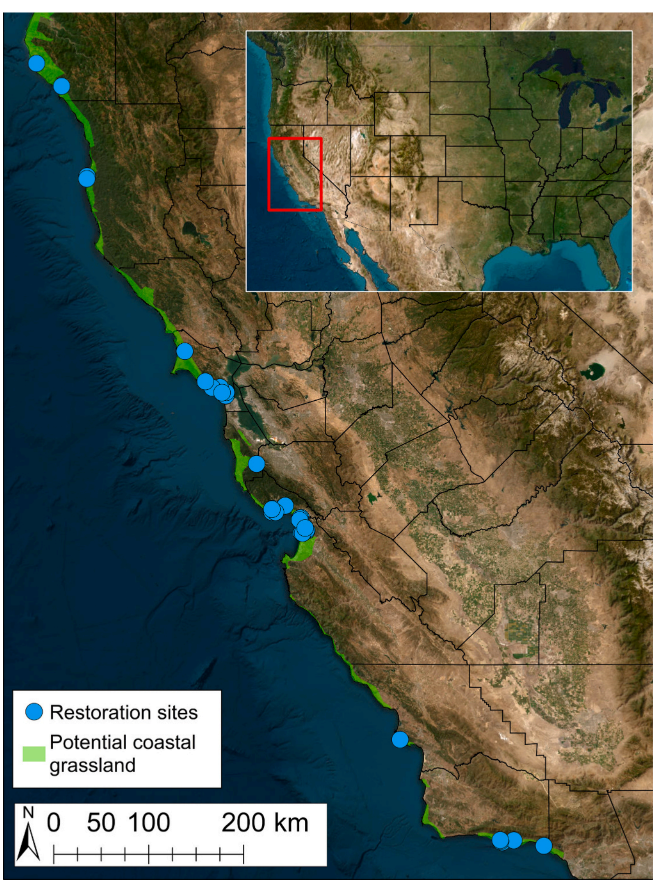

Fig. 1. Study region, restoration sites, and extent of historic coastal grassland habitat in California, USA.

Finally, we highlight unexpected differences in results from statutory and voluntary projects. Restoration projects are motivated by a range of goals, including compliance with legislation (Holl, 2020). Some grassland restoration projects in coastal California are legally-mandated by county general plans or regional regulatory agencies ("statutory projects") and others are undertaken voluntarily ("voluntary projects") when a manager had a keen interest in restoration or discretionary funds (Hagger et al., 2017). Due to their inherent differences, statutory and voluntary projects have different project constraints, approaches, and monitoring goals. For example, past studies suggest voluntary projects tend to have limited monitoring due to budget limitations (Brancalion et al., 2019; Mönkkönen et al., 2009). However, practitioners who create voluntary restoration projects may have greater intrinsic motivation for undertaking the project compared to mandated statutory projects (Bittmann and Zorn, 2020; Mönkkönen et al., 2009) and may use innovative methods for habitat restoration due to limited funding and fewer legal requirements (Hagger et al., 2017).

2. Materials and methods

2.1. Study area

California is a biodiversity hotspot (Myers et al., 2000) and its grasslands host nearly 90 % of the state's endangered and threatened plant species (Eviner, 2016). California coastal grasslands evolved with maritime fog during otherwise hot, dry summers, and are one of the most diverse grassland types in North America with numerous forb species (Ford and Hayes, 2007). The extent of these native grasslands has been reduced by 99 % due to urban development, conversion to agricultural lands, and altered disturbance regimes, and non-native species dominate most of the remaining coastal grassland (Ford and Hayes, 2007). Hence, they are the focus of extensive restoration efforts (Stromberg et al., 2007) and often designated as environmentally sensitive habitat areas (California Coastal Act, 1976).

2.2. Ecological field surveys

Study sites (SI Table 1) spanned a 1000-km distance from Carpinteria (Santa Barbara County) to Petrolia (Humboldt County), CA, USA (Fig. 1; Barbour et al., 2007), covering approximately 90 % of the extant range for California coastal grasslands. Average annual temperature and precipitation at the southern end of the gradient is 15.0 °C and 451 mm, as compared to 11.6 °C and 1002 mm at the northern end (30-year average from 1990 to 2019; SI Table 2). Precipitation was within 25 % of the long-term average for most sites during the first (2019) and third (2021) sampling years, but the second year (2020) had much lower precipitation (SI Table 2). For this study, we selected restoration sites that were: 1) actively "reconstructed" *via* planting or seeding native plants, 2) ≥3 years post-implementation, 3) ≥0.5 ha, and 4) experience summertime coastal fog (Ford and Hayes, 2007). Regular presence of coastal summertime fog was confirmed by land managers during site selection. We chose to focus on sites that were actively "reconstructed" *sensu* Gann et al., 2019, because we wanted to assess whether the most intensive grassland restoration efforts were successful, and because grassland plants tend to be strongly dispersal limited (Kiviniemi and Eriksson, 1999; Pinto et al., 2014). California grasslands are dominated by invasive non-native species, so invasive management alone is rarely successful, particularly in sites that have been used for agriculture and have depleted native seed banks (Hayes and Holl, 2003; Stromberg et al., 2007). As such, active reintroduction through planting or seeding is often required to recover local biodiversity. Moreover, invasive control methods and intensity vary widely making comparisons challenging.

We conducted vegetation surveys at 37 restored coastal grasslands during the peak growth season for Mediterranean climates (April–June) over a three-year period: 32 sites in 2019, 19 in 2020, and 34 in 2021 (SI Table 1). We monitored for multiple years because grassland ecosystems can show strong interannual variation (Zhu et al., 2016). The projects ranged from 5 to 33 years post-implementation by 2021. Through our exhaustive search for all restored coastal grasslands in California, we contacted 213 land managers, researchers and government officials to identify all potential study sites that met our criteria. In 2020 and 2021 we resurveyed the original 32 sites where possible given COVID-19 travel and access limitations (SI Table 1). We identified 16 additional sites that fit our surveying criteria through management interviews after 2019 surveys. We could not survey eight of these newly identified statutory projects because land managers would not permit access. We surveyed four additional projects (one statutory, three voluntary) in 2021 and did not survey the other four newly identified voluntary projects because they were executed by agencies for which we already had surveyed four or more sites.

At each site, we estimated absolute plant cover at the species-level in 0.25-m^2 quadrats every 5-m along 50-m transects (11 quadrats per transect). We estimated plant cover to the nearest 1 % for cover ≤10 %, and for cover >10 % we estimated cover into 5 % bins (*e.g.*, 10–15 % … 95–100 %). We used 3–16 transects scaled for project area which ranged from 0.5 to 13 ha.

2.3. Management data

We reviewed available documents to determine project: 1) restoration goals, 2) age and area, 3) planting composition, and 4) voluntary (projects that had no legal requirement or incentive) or statutory status. Documents included any plans or permit applications that were completed prior to implementation, but only 25 % of projects had documents. We asked land managers to provide information on these four topics during semi-structured interviews if a project did not have documentation.

Management interviews can help contextualize patterns observed from vegetation surveys that are not always readily apparent (Cabin et al., 2010; Homewood et al., 2001) and help guide better allocation of resources to improve future restoration efforts. We conducted semi-structured interviews with restoration managers individually through video meetings and asked about restoration practices, financial and labor investment, plant selection, and perceived barriers to restoration goals (full interview guide in Appendix A). For interview consistency, the same person (JCL) conducted all the interviews. Semi-structured interviews have guiding topics but are flexible to allow the participant to direct the conversation (Dunn, 2000). Semi-structured interviews were conducted after the first round of vegetation surveys in 2019 because we asked managers to reflect on their specific project outcomes, as measured by our field surveys. Although there were 37 projects, we conducted 26 interviews because, in some cases, multiple sites (up to five) were managed by one agency. In such instances, we interviewed two land managers when possible. Managers of two statutory projects elected to not participate in interviews. Interviews and document analyses were approved by the University of California Institutional Review Board.

2.4. Assessing restoration outcomes

Original project targets were used to determine whether restoration efforts achieved project-based goals using plant community data. Because projects had different targets, we compared project outcomes relative to a standard performance metric of ≥25 % native cover and ≥5 native species. Although 25 % cover may appear to be a low target, California grasslands are highly susceptible to invasion, making it difficult to achieve high native cover (Ford and Hayes, 2007; Stromberg et al., 2007), so statutory requirements typically require projects to achieve between 25 and 50 % native cover. Moreover, the classification of native grasslands in California only requires >10 % native cover (Barbour et al., 2007). A global review also indicated that 20 % native cover is a typical goal for working lands (Garibaldi et al., 2021). We used a singular numeric target for species richness to be consistent with how projects are designed and monitored for statutory compliance but acknowledge that this could contribute to a bias of higher success for larger projects due to higher sampling effort and a well-established species-area relationship (MacArthur and Wilson, 1967).

To determine site-level plant cover we first averaged cover by species identity across the 11 quadrats for each transect, and then averaged cover by species across all transects for each site. To determine native and non-native cover we summed cover of all native and non-native species within each quadrat along a transect and then averaged values of native and non-native cover the same as for site-level species cover. We quantified site-level native species richness ("raw native richness") by summing the total number of native taxa at a site, as we were only interested in native taxa. We calculated native rarefied species richness using `rarefy` through the VEGAN package (Oksanen et al., 2018; R Core Team, 2020) to correct for differences in the number of transect per site and potential undersampling. Rarefied native richness was calculated for each site at the quadrat level, which consisted of 33–176 sampling points, dependent on site size. For assessing whether projects reached targets, we compared plant cover and raw native richness (number of native taxa) with both project-based goals and our standard performance metric. For statistical models we used rarefied native richness (Oksanen et al., 2018), though results using raw and rarefied native richness were similar. All values were calculated per sampling year and compared at the site-level (n = 37). Trends in plant metrics (native and non-native cover and native richness) were similar across years despite differences in annual precipitation (SI Table 2). For simplicity, we use the most current annual (2021) vegetation data when possible and 2019 data for projects with no 2021 data. We used 2019 and not 2020 data for projects with no 2021 data because 2019 and 2021 were more climatically similar (SI Table 2).

We used generalized linear models (GLMs) to examine the relationships between cost per hectare and post-implementation project with native cover, non-native cover and rarefied native richness (SI Table 3). Using one-tailed Spearman's rank correlation tests, we evaluated the relationship between the number of restoration species used against both cost per hectare and rarefied native richness (SI Table 3). We used analysis of variance with a covariate (ANCOVA) of post-implementation project age to test the effect of our independent variable, maintenance intensity (low = no or annual non-targeted biomass control; medium = targeted invasive control annually twice or more and low-cost seeding; high = periodic invasive control, permanent staff, replanting efforts; Appendix B for more details) on plant cover, rarified native richness, and cost per hectare. We compared plant metrics between statutory and voluntary projects using *t*-tests (SI Table 3). Analyses were completed in R (v4.0.3; R Core Team, 2020) and maps were created using ArcGIS (v10.8.2; ESRI).

3. Results

3.1. Project outcomes

Native species cover ranged from 2 to 74 %, raw native richness ranged from 3 to 65 and rarefied native richness ranged from 5 to 107. Non-native cover ranged from 10 to 110 % and raw non-native richness ranged from 12 to 53. Forty-three percent of surveyed projects were statutory and 57 % were voluntary. Project related costs ranged from \$371 to \$66,718/ha with an average cost of \$26,579 ± \$24,031/ha.

Project-based goals for voluntary projects all were directional, either for increasing native cover or decreasing non-native cover or erosion. Prior to 2000, statutory projects mostly had directional goals, but projects initiated after 2000 all had numeric, time-bound targets (*e.g.*, 25 %

native cover after 5 years). All but two projects reached project-based goals (35/37 = 95 %). However, managers for 25 % (4/16) of statutory projects indicated that targets were reduced by the regulatory agency when they were not reached, so that a project would reach its new, adjusted project-based goal. In all three survey years, ~80 % of surveyed projects reached the standard 25 %-cover metric (2019: 82 %; 2020: 79 %; 2021: 79 %).

Projects with high and medium maintenance intensity had higher rarefied native richness ($F = 6.09$, $p = 0.007$), native cover ($F = 8.84$, $p < 0.001$) and lower non-native cover compared to sites with low maintenance ($F = 4.41$, $p = 0.020$; Fig. 2). However, there was no relationship between annual cost per hectare and plant cover metrics (SI Table 3). Cost per hectare did not differ as a function of maintenance intensity ($F = 1.77$, $p = 0.196$). On average, high intensity projects spent \$31,814 ± \$21,921/ha whereas, medium intensity spent \$36,242 ± \$29,926 and low maintenance projects spent \$16,593 ± \$20,178. Statutory projects spent more per hectare compared to voluntary projects ($t = 3.00$, $p = 0.007$) but the two types of projects did not differ in native and non-native cover (SI Table 3). However, voluntary projects had higher rarefied native richness compared to statutory sites ($t = 1.99$, $p = 0.027$). Not surprisingly, 81 % of project managers indicated that funding limited management decisions such as plant selection and maintenance intensity.

Project age (years post-implementation) was not significantly correlated with native ($t = 0.67$, $p = 0.509$) or non-native cover ($t = 1.74$, $p = 0.091$; Fig. 3A). Unsurprisingly, native species cover was negatively correlated with non-native cover ($t = -4.30$, $p < 0.001$; Fig. 3B) and positively related to rarefied native richness ($t = 4.79$, $R^2 = 0.379$; $p = 0.032$). As expected, all managers (100 %) indicated that invasive species management was a barrier to achieving project goals and diverted focus from other management activities that could further increase habitat quality. Seventy-eight percent of projects indicated they would have increased maintenance intensity or increased the number of species planted if they had additional financial resources.

All statutory projects undertaken after 2000 had post-implementation monitoring. No voluntary projects had post-implementation monitoring, but 78 % indicated they would monitor if given sufficient funding. Pre-2005 only 10 % of restoration managers believed they could achieve restoration goals but post-2005, 65 % were confident in reaching project goals.

Ninety-two percent of restoration managers preferentially use one or more of the same seven species (*Achillea millefolium*, *Bromus carinatus*, *Danthonia californica*, *Elymus glaucus*, *Festuca rubra*, *Hordeum brachyantherum*, *Stipa pulchra*) for restoration because they anticipate these species will have sufficiently high survival or growth to meet project goals. Half or more of all projects specifically used *S. pulchra* (69 %), *E. glaucus* (59 %), or *B. carinatus* (50 %) for this reason. All preferentially selected species were perennial bunchgrasses (Poaceae), with the exception of *A. millefolium* (Asteraceae), which is a circumboreal rhizomatous perennial forb present in a range of ecosystem types. These seven species comprised 50 % or more of the native cover at most sites that met the standard performance metric. Most managers indicated they used three to six species for restoration, with a limited number of projects that used more than nine species (Fig. 4A). Notably, seven projects only utilized one species, and none used two. The total number of species used for restoration was weakly, positively correlated with restoration costs per hectare ($r = 0.366$, $p = 0.039$; Fig. 4B) and rarefied native richness ($r = 0.361$, $p = 0.041$; Fig. 4C).

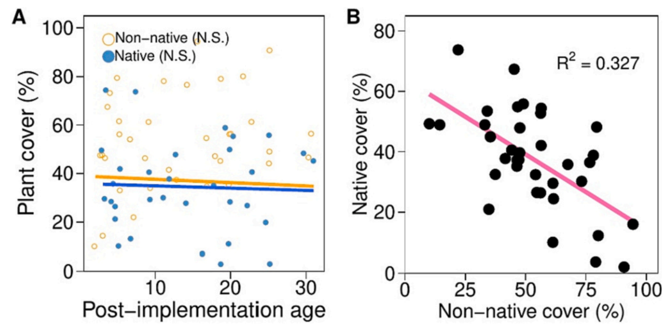

Fig. 3. Relationships (A) between post-restoration age and plant cover, and (B) native cover and non-native plant cover. Points represent restoration sites (n = 37).

4. Discussion

Contrary to our initial expectations, most coastal grassland restoration projects in California achieved their project-based goals, a standard performance metric, and statutory compliance for native plant cover. Interestingly, voluntary projects achieved similar plant cover and higher native richness compared to statutory projects despite spending less. At a regional scale, we found that managers commonly use a restricted subset of the species pool available for restoration, which can lead to habitat-wide biotic homogenization (Holl et al., 2022). Moreover, our study raised concerns about (1) the lack of openness to compliance monitoring by some statutory project managers; and (2) cases of lowering restoration targets to ensure that projects were compliant with permit requirements. We draw on the important insights and perspectives we gained from project documents, land managers interviews, and restoration in other ecosystems to suggest strategies to address these concerns and more effectively allocate limited financial resources to improve restoration efforts.

Our study uncovered some concerning issues regarding statutory restoration projects. We were given permission by land managers to survey every voluntary project but denied access to a third of identified

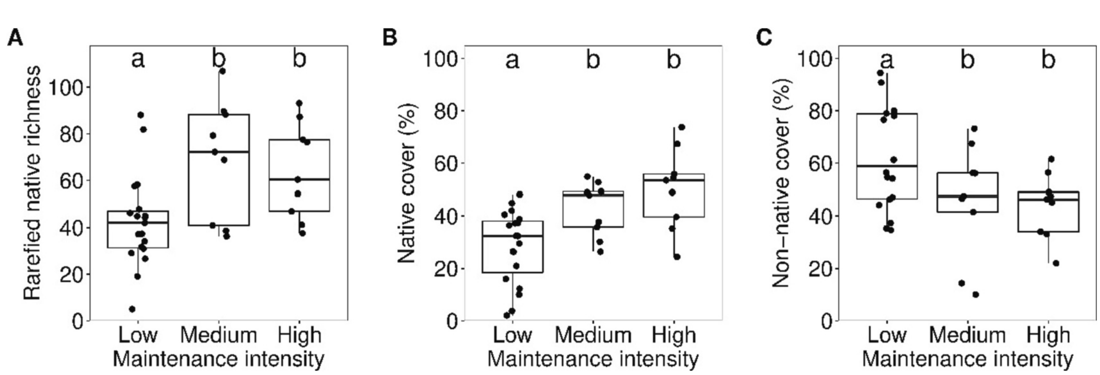

Fig. 2. Relationship of maintenance intensity with (A) rarefied native richness, (B) native cover, and (C) non-native cover across 37 sites using the most current annual data (2021) when possible or data from 2019 when not possible. Points represent restoration sites. See Appendix B for details about classification of maintenance intensity (n = 19 low, 9 medium, 9 high).

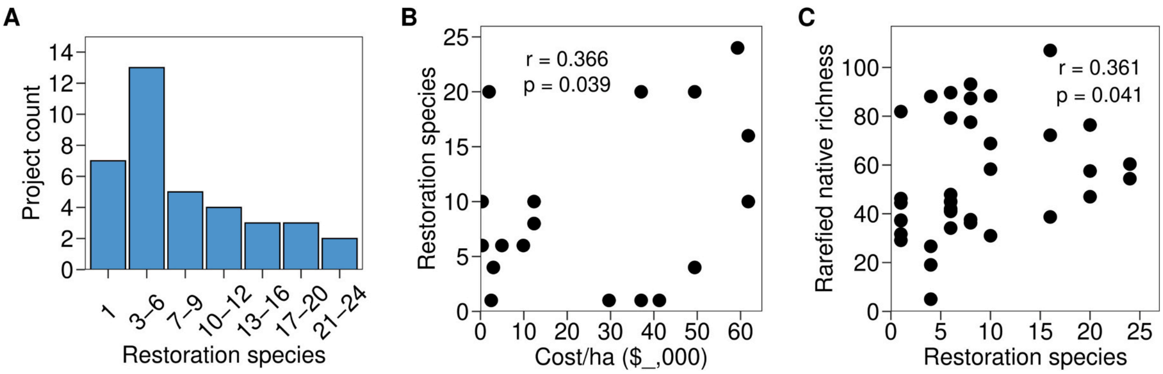

Fig. 4. (A) The binned number of native species planted or seeded ("restoration species") across surveyed restoration projects, (B) relationship between cost per hectare and the number of restoration species; and (C) relationship between the number of restoration species and rarefied native richness. Points in panels B and C represent restoration sites; *r* = Spearman's correlation efficient.

statutory projects. This raises serious concerns about the rigor with which biodiversity offsets are being monitored (Maron et al., 2016; Theis et al., 2020). Although we can only speculate on the outcomes of access-denied statutory projects, this result indicates that policies are needed to allow independent assessment of statutory projects in perpetuity (Skousen and Zipper, 2014). Although certain legal statutes permit this, the approval process can be time intensive and inconsistent across political boundaries. Indeed, we attempted to gain access to restricted sites, but by the time approval was granted by the responsible agencies, plant identification was not viable, as most species at potential study sites had already set seed, leaving mostly senesced or dormant standing vegetation.

To ensure consistency in statutory monitoring and to ensure ongoing compliance as mandated, we recommend delegating funds and responsibilities through legislation for independent monitoring to a regional agency. For example, under the U.S. Surface Mine Control and Reclamation Act, compliance with reclamation efforts following mining are monitored by trained inspectors who are employed by U.S. government state agencies (Skousen and Zipper, 2014). Government or non-governmental third-party professionals would follow a standard protocol for assessment, which minimizes conflicts of interest with demonstrating compliance with outcomes (Godwin et al., 2021). Ensuring these data are publicly available would further increase transparency in legal compliance when evaluating restoration success (Wallach et al., 2018), and provide information for land managers to adapt future practices.

Our interviews with land managers revealed a troubling result that regulatory agencies sometimes lower baselines for mitigatory statutory projects. The reduction of statutory plant cover targets to meet observed outcomes raises concerns about the widespread use of restoration to mitigate habitat destruction elsewhere (*i.e.* mitigation banking), as restoration efforts rarely reach similar function and diversity as remnant habitats (Bull et al., 2013; Moreno-Mateos et al., 2015; Theis et al., 2020). We have not seen the issue of adjusting targets discussed in the literature but suspect that it may occur in other ecosystem types and think that it is an important area for further investigation. If this is a common practice, then there need to be strict criteria for when these targets are adjusted, implications for offsetting environmental mitigation and a clear record documenting changed goals (Brandt-Hawley, 2021).

Insufficient funding was commonly viewed as a factor limiting restoration success across our interviews (Bayraktarov et al., 2015; Brancalion et al., 2019; Cabin et al., 2010). We, however, suggest that the relationship between the amount of money invested and outcomes is not necessarily linear (Bayraktarov et al., 2019) and funds need to be thoughtfully allocated both within and among projects. In our study, there was no relationship between direct monetary costs and plant cover or maintenance intensity, which is likely due to a few projects in which costs were inflated by other expenditures for consultants or construction (*e.g.*, removing concrete from a retired lumber mill). In contrast, we found that projects with medium and high allocation to maintenance had improved restoration outcomes, which highlights the importance of budgeting for long-term maintenance to increase restoration success (Kimball et al., 2015). Projects with high maintenance had an annual budget for management in perpetuity, which highlights the need for funding pools that focus on long-term stewardship.

Our interview results were consistent with prior research showing that practitioners often plant or seed a small subset of a highly diverse regional species pool in an effort to reduce risk and cost while maximizing success (Barak et al., 2022; Brancalion et al., 2018; Lesage et al., 2018). Heavy reliance on just seven species at the expense of countless other species is cause for concern, as California coastal grasslands are one of the most diverse grassland types in North America with over 400 native plant species (Ford and Hayes, 2007). Over long temporal scales, coastal grasslands may support less regional richness (gamma diversity) as remnant habitat is gradually degraded, and restoration projects commonly reintroduce only a handful of well-tested species (Bartholomew et al., 2022). A growing body of literature suggests that typical restoration species selection practices can lead to biotic homogenization at multiple levels of diversity across a wide variety of ecosystems (Holl et al., 2022; Matthews and Spyreas, 2010; Zhang et al., 2022).

Given that the relationship between project cost and species richness was weak, we think that several strategies besides additional funding could help to increase the number of locally distinctive species (*i.e.*, endemic or range-restricted) used for restoration. Our interviews and other research suggest that the use of fewer species may be due to insufficient information about propagation protocols for a diverse suite of species (Bartholomew et al., 2022; Brancalion et al., 2012; Ladouceur et al., 2018; White et al., 2018). This lack of knowledge, combined with practitioner demand for "high success" species, means that native seed nurseries typically produce a restricted subset of the local and regional species pool (White et al., 2018). Funding for the co-production of scientific studies between scientists and restoration managers can improve knowledge of natural history, propagation protocols and reintroduction methods to address the science-practice gap (Bartholomew et al., 2022; Cabin et al., 2010), and in turn, increase the use of less utilized species (Ladouceur et al., 2018). Regional restoration networks and seed exchange programs can be useful in developing nursery propagation of a wider variety of species (Brancalion et al., 2012). Furthermore, legislative policies could be implemented for statutory restoration to require the use of locally distinctive native species (Chaves et al., 2015), or to designate experimental zones that allow managers to test rarely utilized species and learn through "intelligent tinkering" (Cabin et al., 2010) without risking noncompliance with statutory targets (Holl et al., 2022).

Despite spending less money per area restored, voluntary projects reached similar levels of native and non-native cover, and even higher rarefied native richness. This may be due, in part, to greater intrinsic motivation for undertaking the project compared to mandated statutory projects (Bittmann and Zorn, 2020; Hagger et al., 2017; Mönkkönen et al., 2009; Wagner and Davis, 2003). It also suggests the importance of sharing results from successful projects since they may have innovative methods to achieve similar outcomes with more limited resources. Additional polices that support tax-exemptions for voluntary projects or generate other financial incentives could be a powerful tool for increasing successful restoration efforts in a region (Barrett and Livermore, 1983; Jantz et al., 2007). Our interviews indicated that both voluntary and statutory projects received funding from government and non-profit grants. This funding was in addition to budgeted support from the restoration agency or developer (for statutory projects) responsible for restoration. Such funding could be tied to regional coordination of experimentation with locally distinctive species, ensuring the use of a diverse suite of species and sharing best practices to enhance the restoration success and regional biodiversity.

CRediT authorship contribution statement

JCL, KDH conceived research ideas; JCL, KDH designed field methodology; JCL led site selection and plant surveys; JCL, DMP designed document analysis and interview methodology; JCL led analysis and writing with input from KDH and DMP; JCL, KDH acquired funds.

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Declaration of competing interest

Authors declare no conflict of interests.

Data accessibility

Data used for analyses is available on PANGAEA Data publisher for Earth and Environmental Sciences (doi.org/doi.pangaea.de/10.1594/PANGAEA.945320).

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Appendix A. Supplementary data

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