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Pricing risk and planning restoration: An examination of management costs and actions on privately conserved land

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Abstract

Market-based conservation programmes rely on private land managers to deliver ecological gains, but the planned costs underpinning those commitments are rarely examined. We analysed planned costs for conservation works on 57 properties (28,352 ha) participating in a biodiversity offset program in New South Wales, Australia to characterise how costs are structured and how they vary with site characteristics. Total planned costs per hectare were higher for properties that had restoration works than for other properties (median total planned cost per ha: 136.9%; 95% CI: 37.9 to 307). After adjusting for property area, vegetation condition and high-threat weed cover, this difference reduced to 44% (95% CI: 5 to 99%). Administrative costs (e.g., monitoring and reporting) averaged 35.4% of total planned expenditure. Costs are affected by property-level factors (e.g., same consultant, landholder, property history) and are allocated to operational units (e.g., weed management zones, burn units). Operational units often contain multiple vegetation zones. Together, operational structure and property dependence create uncertainty in zone-level cost estimates, with implications for how the costs are assessed and budgets are planned. Weed management was the largest and most variable cost category, yet the biophysical characteristics (area, condition and weed cover) explained only about 20% of per-hectare cost variation when the model was validated across held-out properties. This indicates that planned weed costs are structured mainly by property-level allocation rather than by measurable site characteristics or actions. Policies that account for these factors will be better placed to deliver on restoration commitments.

Highlights

· Planned costs arise from property-level processes and are allocated to vegetation zones through operational unit reporting. Improvements in ecological condition are reported for vegetation zones at a finer scale. Scale mismatch limits what inferences can be made about the true costs of restoration on private properties.

· Administrative overheads averaged 35.4% of total planned expenditure representing a fixed cost burden that may constrain enrolment for smaller properties.

· The difference between restored vs non-restored properties was 136.9%. After adjusting for property characteristics, this attenuated to 44%, consistent with restoration actions being expected in more degraded areas.

· Hand planting was the most common restoration action, suggesting a focus on woody structural attributes.

1 Introduction

The primary focus of restoring and conserving lands for biodiversity is to maintain and improve ecological condition. Costs associated with both protection and restoration influence both participation rates (Archibald et al. 2025) and outcomes (Waldron et al. 2020; Craigie & Pressey 2022). Understanding the relationship between costs and management decisions is critical to determining whether financial plans align with or compromise ecological objectives (Smallhorn-West & Pressey 2022). Yet policy-linked conservation interventions may (inadvertently) embed cost structures that reflect institutional requirements rather than ecological requirements constraining management decisions independent of site conditions (Conway & Brudvig 2024).

Although initial ecological condition (Conway & Brudvig 2024), landscape context and threatening processes (Visconti et al. 2010, 2016) determine what management actions are ecologically appropriate, the choice to implement certain management actions reflects interactions between ecology, landholder capacity (Gray et al. 2015), financial constraints (Carwardine et al. 2019), risk perception, resource availability and institutional factors (Ruoso et al. 2025). In practice, this may result in short-term lower cost actions (Mañas-Navarro et al. 2023). Compounding these challenges, poorly estimated costs can create significant implementation challenges (Iacona et al. 2018).

Biodiversity offsetting provides an opportunity to examine the relationships between policy settings and planned budget relationships because offsetting explicitly links financial mechanisms to measurable ecological targets within a structured policy framework (Gardner et al. 2013; Pope et al. 2021). In New South Wales, Australia, the Biodiversity Offsets Scheme (BOS) establishes offset sites (Biodiversity Stewardship Agreements, BSAs) that secure the in-perpetuity management and protection of biodiversity on private land. Improvements (or 'gains') in vegetation condition are predicted by the ecosystem's recovery potential and the actions undertaken. Management actions are outlined by the Biodiversity Assessment Method (BAM) (Department of Planning & Environment 2020) enabling management gains to accrue through two pathways: 'required' actions to manage threatening processes (e.g., weed control and feral pest management) and 'active' actions to reinstate missing biodiversity elements (e.g., planting or reinstating functional attributes such as coarse woody debris), which from here we refer to as "restoration." Although there are other contributing pathways for gains (avoided loss and security benefits) under the BAM (Department of Climate Change, Energy, the Environment and Water 2024), we focus here on management gains. For an overview of alternative gain pathways, including threatened species habitat, see SI Section 1.1.

Features of the BOS's policy structure may influence planned costs, independent of whether restoration is undertaken. First, ecosystem credit prices are determined by location, vegetation type, habitat features (e.g., hollow-bearing trees), historical clearing, market demand and ancillary policy settings (e.g., 'like-for-like' rules) (Department of Planning and Environment 2023) high condition trading unit, creating a potential misalignment between management costs and revenue which is partly accounted for by the number of credits generated (**Section 1.1**). The policy is somewhat flexible in how management actions are implemented, enabling bespoke management. While such flexibility is important because it allows context-specific management, individual discretion may influence how management actions are planned and costed. For instance, practitioners may omit necessary preparatory phases of restoration, with consequences for both planned costs and predicted ecological outcomes (Smith et al. 2021).

The planned budgets of BSAs offer a unique opportunity to examine how costs are operationalised but carry important caveats. Properties may only enter the BOS when projected revenue exceeds planned costs (based on the market value of ecosystem credits) and properties where restoration was determined too costly or too risky may self-can filter (Dewally et al. 2025) either by not participating in the BOS or not undertaking restoration. This effect means that the observed planned costs (particularly between restored and non-restored areas) cannot be truly interpreted as independent of the BOS policy settings, but rather as circumstances where conditions for both entry into the BOS and the choice to implement restoration were deemed viable (Kedward et al. 2023).

We explore planned costs through three research questions: First, to what extent does restoration explain the patterns and magnitudes of variation in planned management costs? We expect that restoration will impose a cost premium, particularly for costs associated with management actions. Because our sample spans both metropolitan and regional contexts across New South Wales, we provide cost patterns by regional versus metropolitan areas which we explore descriptively. Second, how do the frequencies of management actions differ between vegetation zones with and without restoration? Third, do ecological characteristics and policy settings explain cost variation? Here, we focus on weed management because it is common to both restoration and non-restoration areas. This study provides an empirical analysis of planned costs across a range of real-world offset sites.

2 Methods

2.1 Study context and Data Sources

The Biodiversity Assessment Method (BAM) provides the assessment method within the regulatory framework for biodiversity offsetting in New South Wales, Australia. Under the BAM, offset sites are mapped into vegetation zones, which are spatial units defined by combinations of Plant Community Type, Threatened Ecological Community and their condition. For each zone, ecologists accredited to apply the BAM assess baseline vegetation attributes, which are used to predict increases in vegetation condition under management. Improvements (or "gains") are converted into tradeable ecosystem and species credits (State of NSW and Department of Planning and Environment 2020a, See SI Section 1.1). We obtained data from the NSW Government's internal offset management system in March 2024 (Government of NSW 2024). Data were cleaned by eliminating test cases, duplicates, superseded or non-progressing cases and cases assessed under the previous NSW Biobanking Scheme (Dupont 2017) (number of total BSAs = 284).

2.1.1 Data sources

We compiled information from the following documents: i) Biodiversity Stewardship Site Assessment Reports (BSSARs) (initial ecological assessments of the property), ii) site management plans (detailed management actions) and iii) Total Fund Deposits (TFD; itemised planned costs).

2.1.2 Sampling procedure

To prioritise document retrieval and to restrict our dataset to zones where comparison between restoration and non-restoration is meaningful (i.e., excluding very-high-condition and very-low-condition zones, where restoration is respectively unlikely to be incentivised or unlikely to be feasible), we estimated a zone-level propensity score for receiving restoration (Austin 2011). The propensity score was derived from a logistic regression with restoration status as the binary outcome and three predictors: baseline vegetation integrity score (0–100), zone area (hectares, log-transformed), and high-threat weed cover (per cent), with bioregion (IBRA, (Thackway & Cresswell 1997) as a random intercept (Izem et al. 2020). Nearest-neighbour 1:1 matching with a calliper of 0.2 standard deviations (Kane et al. 2020) was used to select pairs of restoration and non-restoration zones with comparable baseline conditions. Match quality was assessed using standardised mean differences (SMDs) and Kolmogorov-Smirnov tests on each covariate (Figure S1; zone-level balance) (Zhang et al. 2019) and after aggregation to property-level. The propensity score functions in our analysis as a sample-restriction and data-prioritisation device, not as a causal adjustment.

Because documentation is filed at the property-level, we aggregated zone-level propensity scores to a single property-level score (the area-weighted mean of zone PS) to identify which properties to pursue for document retrieval. We prioritised document retrieval from 164 properties with the highest weighted propensity score; of these, 57 had complete and available documentation (Figure S2 shows their spatial distribution).

2.1.3 Cost categorisation and management actions

From within the BSSARs and management plans, we identified and classified 17 management actions for each vegetation zone. We allocated a binary (presence/absence) to each action where documentation indicated its use in each zone (See summaries in **Figures S3–S7 & Table S3**).

We identified 15 broad cost categories using unsupervised text mining for each of the TFD line items (n = 1,981) and supplemented this with manual classification aligned with the policies' classification of management actions (Department of Planning and Environment 2020). This ensured alignment between variable costs and the governing policy structure. All line items were manually attributed to cost categories by SH with support from JD.

2.1.4 Operational units and vegetation zones

Cost values for BSAs arise from property-level processes: a single consultant, each year, draws on the regional cost base and the property's management plan to construct cost estimates for the full set of management actions on that property. Cost values are allocated to vegetation zones via operational units. Operational units are organised around the logic of management delivery (e.g., weed management blocks group co-occurring weed species requiring similar control methods). Vegetation zones, by contrast, are organised around ecological condition and vegetation type, and are the units at which gains are calculated under the BAM. Operational units typically span multiple vegetation zones, producing a structural misalignment between the unit at which costs are reported and the unit at which gains are assessed. Per-hectare costs at the zone level were allocated by attributing the sum of the TFD line items for an operational unit to each overlapping vegetation zone in proportion to area share. For example, $10,000 AUD for mechanical removal in Operational Zone A overlapping vegetation zones 1 and 2 was allocated to each zone in proportion to its share of the combined area (Table S4). Where line items applied to all

zones within a property (e.g., property-wide administration, monitoring, and reporting), costs were analysed at the property scale. This allocation procedure, combined with the property-level source of the underlying cost values, has two consequences for the structure of the cost data. First, the per-hectare rate assigned to a vegetation zone is shared with the other zones in its operational unit, because all zones in an operational unit receive the same allocated rate by construction. Second, allocated rates across different operational units within the same property are not independent of each other, because all are outputs of the same property-level consultant, year, and total fund deposit. Distinct per-hectare rates are therefore coarser than either vegetation zones or operational units: a single rate is shared not only among the zones within an operational unit but often across operational units that draw on the same unit cost. For characterising cost variation, the relevant unit is thus the distinct per ha rate rather than the vegetation zone.

Within-property cost-sharing is substantial. Of the 1,712 unique per ha costs for each zone level cost-category, 87% (n = 1,515) were within groups of three or more zones sharing a common per-hectare rate within the same property (hereafter, 'rate groups'); 4% (n = 70) were paired with one other zone; and only 9% (n = 151) were uniquely priced at the zone level. For weed management, 85% of zone observations (n = 400) sat in groups of three or more zones sharing a per-hectare rate; for fire management, 93% (Table S4). This structural dependence is why we restrict formal cost inference to the property-level.

2.1.5 Cost standardisation and allocation

We focused on costs allocated to the first 20 years of property management (the period over which gains are accrued). Costs were standardised to AUD 2024 using the Consumer Price Index (Australian Bureau of Statistics 2025). For each property, we applied the ratio of the 2024 CPI index value from the June quarter, yielding year-specific adjustment accounting for variable inflation rates across the establishment years within our sample (2018–2024). Standardised costs were then discounted at 3.75%, representing a conservative inflation rate that aligned with the discounting rate at the time of analysis (Australian Centre for Evaluation 2025).

2.2 Statistical analysis

All analyses were conducted in R version 4.4.1 (R Core Development Team 2013). Data cleaning and manipulation were performed using the tidyverse and dplyr packages (Wickham et al. 2019a, 2019b).

2.2.1 Objective 1: Is restoration associated with higher costs?

2.2.1.1 Total cost differences: Property-level Costs

We considered three different levels of cost: 1) the total planned cost (total TFD over 20 years) per hectare at property-level (TotalCostperha$_j$); 2) administrative cost per hectare at property-level (AdministrativeCost perha$_j$); 3) management-specific costs related to land management actions at the property-level excluding those exclusively used for restoration and which cannot occur in non-restoration zones (e.g., planting and the emplacement of coarse woody debris; SharedCost perha$_j$).

EQ.1: $\log(\text{Costperha}_j) = \beta_0 + \beta_1 \text{Restoration}_j + \varepsilon_j$

EQ.2: $\log(\text{Costperha}_j) = \beta_0 + \beta_1 \text{Restoration}_j + \beta_2 \log(\text{Area}_j) + \beta_3 \text{VI}_j + \varepsilon_j$

EQ.3: $\log(\text{Cost perha}_j) = \beta_0 + \beta_1 \text{Restoration}_j + \beta_2 \log(\text{Area}_j) + \beta_3 \text{VI}_j + \beta_4 \text{HTW}_j + \varepsilon_j$,

where *j* is an index of individual properties, *Restoration* is a binary indicator for whether the property has at least one zone with restoration (1) or not (0). Area (hectares) was log transformed (natural log), VI was the area-weighted mean vegetation integrity score, and HTW was the area-weighted high-threat weed cover. All cost variables were log-transformed (natural log). Cook's distance identified potentially 4 influential observations (Fox 2019). Sensitivity analysis excluding these observations showed similar estimates (**Tables S5 & S6**). HC3 robust standard errors were used to account for heteroscedasticity (MacKinnon et al. 2023). Residual diagnostics are provided in **Figures S8-S10** and displayed non-normality and moderate heteroscedasticity at the property-level which are common characteristics of financial data.

2.2.1.2 Inferences, sensitivity and effective sample size

A description of the cost per ha for the properties' proportional area under restoration is provided in **Table S7**. We re-estimated EQ.3 using inverse-probability-of-treatment weighting (IPW) with stabilised, trimmed weights and using a continuous treatment (proportion of property area under restoration) in place of the binary indicator (**Tables S8 & S9**). The IPW estimate is consistent with EQ.3, producing a smaller point estimate at full restoration plausibly explained by regional asymmetries.

We report the minimum detectable effect (MDE) at 80% power (alpha = 0.05) given our realised sample size. At the property-level (n = 57), the MDE was estimated using a standard two-sample framework (Table S10).

E-values represent the minimum strength of unmeasured confounding that would be required to fully explain observed association. Results indicated moderate sensitivity to unmeasured confounding (**Table S11**;(VanderWeele & Ding 2017; Blum et al. 2020).

2.2.2 Objective 2: Patterns of common management actions

We compared counts of each management action for three groups: zones with restoration, counterfactual zones (matched zones without restoration, derived from Section 2.2), and high-condition zones (zones with a VI > 70). To characterise restoration, we grouped restoration activities into four sequential phases: site preparation (e.g., ripping, tilling, topsoil removal), plant establishment method (e.g., hand planting, direct seeding), follow-up actions (e.g., watering, weeding) and contingency actions, quantifying the variability of restoration planning across sequential activities (Mendez-Toribio et al. 2021).

2.2.3 Objective 3: Cost variability in Weed Management

To evaluate how policy settings, including site characteristics and choice of management actions influence cost variation, we focused on weed control. This cost category was the most variable and is common to both restored and non-restored zones. We used a random forest regression model (Liaw & Wiener 2002) with the randomForest package (Breiman et al. 2018) to model the log-transformed cost per hectare. We aggregated vegetation zones to the operational units at which weed costs are allocated (the property x operational unit), summing cost and area, taking area-weighted means of high-threat weed cover and vegetation integrity. We coded each control method as present (1) or absent (0) if it was applied to the operational unit. Operational units with no positive weed cost were excluded (n = 7). Variables accounting for site characteristics were high-threat weed cover, log operational-unit area, and area-weighted vegetation integrity.

We analysed the log-transformed per-hectare first by quantifying how much of the variation reflects property-level cost allocation versus operational unit-level predictors. We fitted a linear mixed model with a property random intercept (lme4; (Bates et al. 2015) and partitioned the variance using the intraclass correlation coefficient (ICC) and marginal (fixed effects) versus conditional (fixed effects plus property) R^2 (Nakagawa & Cuthill 2007). Second, to test whether nonlinearities, interactions and the specific control methods explain additional variation, we fitted a random forest (randomForest; (Liaw & Wiener 2002; Breiman et al. 2018). Because operational units are nested within properties, and because many share the same allocated rate (the ~ 300 weed operational units resolve to 116 distinct cost rates), within-sample (out-of-bag) error is optimistic, so predictive performance was assessed by property-grouped cross-validation, holding out whole properties (five folds). We report variable importance as the percentage increase in mean squared error on permutation (Gregorutti et al. 2017; Figure S11) and partial dependence for the two continuous predictors (Fig. 4)

3 Results

We examined 57 Biodiversity Stewardship Areas, comprising 480 vegetation zones and 28,352 hectares. Properties with restoration (n = 30) had lower weighted vegetation integrity (mean = 47.7, SD ± 18.4) compared to properties without restoration (mean = 63.4, SD ± 10.3) and were, on average, smaller (mean property size = 323 ha, SD ± 489 ha) compared to properties without restoration (mean = 691, SD ± 1,268 ha). Administrative costs associated with site establishment, monitoring and reporting and other overhead costs were, on average, 35.4% of the total expenditure and varied between restoration (mean = 29.7%, SD ± 9.5%) and non-restoration properties (42.6%, SD ± 11.3). More than half (57%) of properties included contingency funding, however, this was less than 6% of the 20-year budget.

3.1 Sample characteristics and cost overview

All cost figures are presented in 2024 Australian dollars (AUD) per ha. In general, property-level costs per ha were higher in Sydney Basin than they were outside this metropolitan region. Administrative costs (e.g., rates, bookkeeping, liability insurance, land taxes) were highest in the Sydney Basin for properties with restoration (median = 710, SD ± 1,664) but were similar to properties without restoration within the Sydney Basin (median 524, SD ± 507). Administrative costs were generally lower outside the Sydney Basin, and marginally higher on properties with restoration (median = 279, SD ± 535). BSA establishment costs (e.g., fencing, gate installation) were highest in the Sydney Basin (median = 982, SD ± 2,820) and were lowest in regions outside Sydney (median = 243, SD ± 257). Both annual reporting and 5-yearly monitoring and reporting were similar on properties outside the Sydney Basin, regardless of restoration, but were highest on properties with restoration within the Sydney Basin (annual reporting: median = 1,341, SD ± 1,124; 5-yearly reporting: median = 842, SD ± 1,007).

Table 2

Property-level costs per hectare over 20 years. Costs are presented for properties with restoration vs those without restoration. Values are in Australian dollars **(AUD ha^{-1},** *2024 equivalent, discounted at 3.75%). The Sydney Basin is a geographic area, classified under the IBRA framework, characterised by the highest population and longest history of degradation in NSW.*

	Other Regions								Sydney Basin							
	Non-restoration				Restoration				Non-restoration				Restoration			
Cost Category	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**
Administration	20	489	227	893	16	400	279	535	5	541	524	507	12	1477	710	1664
Annual Reporting	22	353	194	487	17	289	173	299	5	477	153	619	13	1447	1341	1124
Contingency	16	307	103	372	9	154	83	112	2	145	145	81	5	491	549	213
BSA establishment	22	557	465	527	16	285	243	257	5	843	360	1222	13	1843	982	2820
Five yearly Monitoring and Reporting	22	434	316	591	17	450	352	409	5	427	204	392	13	1354	842	1007
Vertebrate Control	22	568	244	1256	17	461	242	466	5	353	391	201	13	1802	901	1873

Zone-level costs varied by geographic region with higher costs observed in the Sydney Basin. The cost per ha for weed management was highest in the Sydney Basin for properties with restoration (median = 29,159, SD ± 22,046). Planting costs (present only on restoration zones) had a median cost per ha of $15,351 (SD ± 83,116) in the Sydney Basin compared to a median of 2,184 per ha elsewhere (SD ± 1,697). Fire management costs were similar for zones with and without restoration outside the Sydney Basin but were higher for properties with restoration within the Sydney Basin (median = 755, SD ± 1,954). There

were no grazing management or habitat enhancement actions in the Sydney Basin. Threatened species costs were similar across properties without restoration in the Sydney Basin (median range: 129–319).

Table 3

Zone management costs per hectare over 20 years by restoration status. Values are in Australian dollars (dollars **AUD ha^{-1}** *;2024 equivalent, discounted at 3.75% per annum).*

	Other Regions								Sydney Basin							
	Non-restoration				Restoration				Non-restoration				Restoration			
Cost Category	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**	**n**	**Mean**	**Median**	**SD**
Erosion Control	68	564	108	1158	11	680	360	793	-	-	-	-	2	14766	14766	0
Fire Management	215	274	107	415	47	217	81	419	70	448	248	720	36	1383	755	1954
Grazing	2	364	364	0	1	364	364	-	-	-	-	-	-	-		
Habitat enhancement	12	187	154	114	7	216	289	97	-							
Logs	-	-	-	-	5	516	558	96	-	-	-	-	17	3144	4815	2333
Maintenance	263	322	266	299	53	279	164	282	94	907	202	1602	46	3160	990	8761
Threatened species monitoring and management	106	442	319	551	22	259	202	197	37	434	129	523	24	365	286	313
Planting	-	-	-	-	51	2360	2184	1697	-	-	-	-	45	44138	15351	83116
Thinning	3	58	58	0	1	335	335	-	1	375	375	-	3	10062	655	16294
Weed Management	267	1685	826	3069	60	775	631	500	94	2555	1582	2482	49	25491	29,159	22046

Management costs were highly variable. There was a negative relationship between log-transformed property area and log-transformed cost per hectare for both non-restoration (R^2=**0.76** and restoration; R^2=**0.66** overall management costs). Furthermore, properties with a lower initial starting condition and those that were generally smaller showed higher per ha costings.

3.1.1 Estimating the association of restoration and planned costs

3.1.1.1 Property-level planned costs

In an unadjusted comparison, the median total planned cost per ha was 136.9% higher on restoration properties (EQ.1: CI 95% 37.9 to 307). After adjusting for property area and vegetation condition (EQ.2) this difference was attenuated to 40% (95% CI: 0.6 to 94.8%). An additional adjustment for high-threat weed cover (EQ3; our primary specification) returned a median difference of 44.2% between restored and non-restored properties (95% CI: 4.5 to 98.8%). This small increase indicates that (conditional on area and vegetation condition) restoration properties carry costs which may be associated with high-threat weeds. We interpret the attenuation from EQ.1 to EQ.3 as a descriptive decomposition, reflecting Gelbach (2016) rather than a causal identification of covariate relationships. Property size had a negative relationship with per-ha costs (Fig. 1B). For components of property-level costs, the adjusted proportional differences in median cost per ha between restoration and non-restoration properties were 8% for administrative overheads (95% CI: -21 to 48%) and 26% for all required management costs (95% CI: -10 to 78%), with confidence intervals crossing zero in both instances (Fig. 2A).

3.1.1.2 Zone-level planned costs

Cost values arise from property-level processes and are allocated to vegetation zones through operational units, so per-hectare figures at the zone level are largely shared across zones within a property rather than varying zone by zone (Fig. 2B) or by vegetation condition (Fig. 2C). Panel 2B plots the number of distinct per-hectare rates against the number of zones receiving each cost on each property. Points cluster on the lower reference line at y = 1, indicating that most properties have one allocated rate across all their zones for a given cost. If costs varied zone by zone, points would follow the diagonal reference line. Figure 2C plots the log-transformed per-hectare cost at the zone level for each cost category, with each zone coloured by starting condition (vegetation integrity score). If costs tracked vegetation condition, a vertical colour gradient would be visible. The absence of a clear gradient indicates that zone-level ecological condition does not order zones by cost (e.g., higher costs are not consistently low condition zones).

This shared-rate structure means zone-level per-hectare figures are largely descriptive of operational allocation rather than of zone-specific cost variation (Fig. 2B, 2C).

3.1.2 Management action frequencies

Several management actions were infrequent including biological control for weed management (2.5%), ecological thinning (2.3%) and livestock grazing (3.8%). Among the remaining actions, management approaches differed substantially between zone categories (Fig. 3**A**). Weed control methods were similar across groups, although mechanical removal and spot spraying were more frequent in restoration zones compared to both counterfactual and high-condition zones. Foliar spraying and hand weeding were common across all zone types (occurring on approximately 60–70%) but were slightly less common on restoration zones. Cut and paint was slightly more common on restoration zones (occurring on approximately 53% of restoration zones). Coarse woody debris addition was more common in restoration zones (~ 40%) than in counterfactual (~ 5%) or high-condition zones (~ 10%) consistent with efforts to increase functional restoration gains. Nest box installation was infrequent (~ 10–15%). (See Table S1 for definitions of management actions)

Restoration actions are implemented exclusively on restoration zones with sequential phases exhibiting some variation (Fig. 3**B**). Site preparation activities (ripping, scalping, or topsoil management) preceded planting in approximately 85% of restoration zones. Within site preparation, weeding was the most frequent method (n = 30 restoration zones), followed by mechanical ground preparation (n = 35). To establish the plants, hand planting of tube stock was the most common approach (n = 65), with direct seeding used in fewer than half of restoration zones. Follow-up management was predominantly watering (n = 50) and supplementary weeding (n = 25). Contingency actions included supplementary or in-fill plantings and were present on about half of all restoration zones.

3.1.3 Cost variation by area and condition: weed management

Weed management was the largest and most variable cost category, and most of its variation was structured between properties. A linear mixed model with a property random intercept attributed 85% of the variance in per-hectare weed cost to differences between properties (ICC = 0.85); the unit-level predictors (area, vegetation integrity, high-threat weed cover and the control methods) together explained only 15% (marginal R^2 = 0.15; conditional R^2 =0.83). The random forest, evaluated by property-grouped cross-validation, explained about 20% of variation across held-out properties, and adding the six control methods did not improve out-of-property prediction (R^2 = 0.20 vs 0.19). Within the fitted model, predicted cost declined with operational-unit area and rose with high-threat weed cover (Fig. 4). Together, this suggests weed budgets are planned chiefly by property-level allocation rather than by measurable zone-level ecology or the specific control methods applied.

4 Discussion

Our analysis of 57 offset sites provides unique insights into how costs are planned within a market-based conservation program. Adjustment for property area, vegetation condition and high-threat weed cover attenuates the proportional difference in median cost per ha to 44% (EQ3, 95% CI: 5 to 99%). The remaining difference represents restoration actions themselves (e.g., hand planting) because these actions carry genuine additional per ha costs. Restoration properties in our sample were, on average, smaller and in poorer vegetation condition. For such properties, administrative costs may constitute a fixed burden (averaging 35.4% of total expenditure) inversely related to property size (Armsworth et al. 2011). Prospective properties may self-filter where the economics are unfavourable, ultimately reducing the opportunity for restoration where it may be most needed (Moilanen et al. 2009; Rampling et al. 2024). Such barriers may help explain the gap between interest and uptake. A recent statutory review of the scheme found that Stewardship (offset site) Expressions of Interest covered roughly 200,000 hectares in 2022–23, yet only about 22,000 hectares were secured as agreements, and concluded that higher enrolment is needed to realise the opportunity for offset sites to attract much-needed investment in restoration (Henry et al. 2023). Measures already introduced under the scheme's improvement program include biodiversity assessments at no upfront cost and the waiving of application fees. These measures target administrative costs directly, consistent with our finding that administrative overheads averaged over a third of the total planned budget.

Costs in our sample were highest in the Sydney Basin, where urban activity is concentrated (Tables 2 and 3). In such peri-urban landscapes, remnant vegetation is typically a mosaic of agricultural and suburban land uses (Polyakov et al. 2013) and patchy in condition (State of NSW and Department of Planning and Environment 2023), so assessors split otherwise similar vegetation into finer, condition-defined zones. Under the BOS, all zones of the same vegetation class receive the same credit price regardless of condition. Theoretically, a lower-condition zone therefore earns the same price per credit as a higher-condition one even though it costs more to manage. If a zone is in moderate condition, it is also predicted to generate larger gains, and hence more credits, before those gains are discounted by the BAM's risk weighting (Gordon et al. 2025). Whether a degraded zone is financially viable thus depends on whether its additional, risk-discounted credits offset its higher management cost; uniform per-credit pricing leaves that balance to credit quantity and the risk weight rather than to price. We did not observe credit revenue and cannot test where the balance falls, but the structure creates a clear tension between the sites that are most costly to restore and the price signal they receive. This tension is inherent to a deliberate choice for uniform pricing and illustrates the difficulty of building restoration incentives into a market instrument (Zu Ermgassen et al. 2026). It also reflects a feature of how the scheme values management: credits are priced, and actions costed, individually, whereas securing a condition outcome depends on a complementary set of actions whose contributions are not additive. Complementarity logic (long central to conservation planning; (Moilanen et al. 2009) and now being extended from individual to combinations of actions (Giakoumi et al. 2024) is the question our results point toward and we return to it below.

More than half (57%) of properties included general contingency provisions for adaptive management in their planned budgets and an additional 86% of zones with restoration provided additional contingency provisions to support their restoration. Although this coverage is not uniform, this proportion reflects integration of adaptive management planning, which is notably important because ecological recovery is difficult to predict (Brudvig & Catano 2024). BSAs without contingency funding may have little or no capacity for adaptive management (Gorrod et al. 2017). Recent policy updates now require contingency funding as part of the total fund deposit (Department of Climate Change, Energy, the Environment and Water 2024b)

Hand planting was the most common restoration method, while direct seeding was rare, despite its known effectiveness (Cuneo et al. 2018) and cost-efficiency (Andres et al. 2024). While this choice of method may be explained by zone scale, the preference for hand planting suggests restoration primarily targets woody structural attributes (tree and shrub cover and richness) rather than herbaceous ground cover (which is typically introduced with direct

seeding). The BAM's gains calculations (which separately weigh composition, structure and function) may inadvertently incentivise woody planting in ecosystems where trees and shrubs carry the highest weight and offer the greatest potential gains. The preference for woody planting may also reflect that the restoration of herbaceous vegetation (e.g., establishing forbs and grasses) is typically more expensive, harder to source and considered riskier than planting (Campbell et al. 2017). Site preparation (e.g., soil remediation, scalping and initial weed management) and follow-up actions (watering, infill planting, post-establishment weed control) were infrequent despite their importance for establishment success. Hand planting provides a management solution that can be directly linked to gains in the BAM's metrics but does not directly incorporate how particular management actions address the underlying constraints to ecological recovery (Sheaves et al. 2021).

Management actions implemented on BSAs reflect interactions between ecological suitability (i.e., what is needed for the ecosystem to recover) (Perring et al. 2015), landholder preferences, consultant recommendations and regulatory approval processes. The interaction between these factors has long been acknowledged in restoration ecology (Miller & Hobbs 2007; Hobbs 2007, 2016). Regarding restoration actions, our results indicate that certain actions (e.g., hand weeding, tree planting) are more common than others. BOS policy specifies which management actions are considered 'restoration actions', making restoration actions implicit to policy design. Supporting guidelines also emphasise how actions are implemented and in what order to achieve objectives (Faith & Walker 1996), aligning with global standards for ecological restoration (Gann et al. 2019). Drawing on complementarity concepts, a core tenet in systematic conservation planning (Moilanen et al. 2009), guidance supporting multi-action prioritisation within the same site (Giakoumi et al. 2024) could be better developed, particularly with a focus on ameliorating threatening processes (Olivares-Rojas et al. 2025). Presently, guidance defines 'required management' and 'restoration' actions separately and does not specify how to assess the interaction between actions. For example, weed management was the largest and most variable cost category, yet only about 20% of its per-hectare variation was explained by operational-unit characteristics, and the control-method mix explained none; planned weed cost is therefore governed chiefly by property-level allocation.

How best to manage uncertainty has been the focus of long-standing discussions in policy development and evaluation (Pindyck 2007). In its calculations of predicted restoration gains, the BAM's future values for each ecosystem attribute (e.g., tree species richness) are specified. The BAM then applies a risk weighting to account for uncertainty in achieving those values. However, it does not mandate specific restoration methods or sequences of actions under restoration needed to inform risk, inadvertently creating a disconnect where risk is priced into credit calculations (through reduced gains) but not addressed in management planning. Risk weightings may, therefore, function as a financial penalty on restoration due to uncertainty in achieving outcomes. Alternatively, risk weightings could function as an incentive for risk management through staging the release of credits when implementation risks are clearly addressed (Rau et al. 2024). Recent studies suggest that, where information about the long-term outcomes of an environmental-economic policy is asymmetric (as is the case for restoration outcomes), policy options which provide flexibility may enable more participation (Bellassen & Shishlov 2017). A statutory review of the Biodiversity Assessment Method is currently examining whether gain predictions can be strengthened and credit yield increased for strategic restoration, and whether assessment complexity can be reduced to lower compliance costs (Department of Climate Change, Energy, the Environment and Water 2024b). Our results, however, locate the larger share of cost variation in on-ground management and property-level allocation rather than in assessment complexity, implying that reforms aimed at compliance cost will ease only part of the cost burden that restoration carries.

A central feature of the New South Wales Biodiversity Offset Scheme and of market-based offset programs more generally is structural misalignments between the unit at which management costs are reported and the unit at which ecological gains are calculated. Costs in the NSW BOS are reported by operational units (e.g., weed management zones, burn units and threatened species polygons). Operational units are organised around the logic of management delivery. Gains, by contrast, are organised by vegetation type and condition and calculated at vegetation zones. Literature on conservation costs has long emphasised that the scale and granularity at which costs are recorded influence the reliability of cost estimates (Armsworth et al. 2011; Adams 2024). Our results across 57 Biodiversity Stewardship Areas describe one illustration of this concern. Because most operational units span multiple vegetation zones, and because different operational units on a property often draw on the same unit cost, per-hectare cost figures are largely shared across zones within a property rather than varying zone by zone. Furthermore, the ecological condition of a zone does not order zones by cost (Fig. 2C). These descriptive patterns mean the effective sample size for any zone-level cost comparison approaches the number of properties (57) rather than the number of zones (480). This structure reflects operational planning and reporting but bounds the precision with which cost differences between restored and non-restored areas can be resolved at the zone scale.

We note several limitations of our analyses and their implications. First, we analysed planned costs rather than actual expenditures. Planned costs represent the funding available, making them policy relevant regardless of whether funds are spent as planned. Second, our sample was constrained by document availability and is not representative of all BSAs. Manual allocation of costs to zones involves individual judgements mitigated through clear allocation protocols. Third, our sample is restricted to properties with fully accessible documentation which bounds the scope of our inferences. Properties for which restoration would be prohibitively expensive self-filter out of the BOS before enrolment, so the unobserved 'eligible but unenrolled' population would, plausibly, have higher planned costs. This implies that the estimate provided in EQ.3, if anything, likely underestimates the underlying cost premium of undertaking restoration across the entire eligible population of properties. E-values (**Table S11**) characterise the strength of residual confounding required to nullify the association but cannot diagnose such selection bias. We therefore interpret our estimates as adjusted associations within the selected population, not as causal effects. Finally, we cover a 20-year management period during which biodiversity gains accrue, but offset obligations exist in perpetuity. Costs may stabilise, decline or increase as ecosystems mature in their response to management or as unexpected threats and pressures emerge. Longitudinal studies tracking actual expenditure would address this gap.

4.1 Conclusion:

Across 57 Biodiversity Stewardship Agreements we show that planned conservation costs are organised by a spatial logic that does not align with how ecological gains (or credit prices) are defined. Costs are set at the property-level and allocated to operational units that each span several vegetation zones, so 87% of zone-level per-hectare costs are shared across three or more zones and the effective sample size for zone-level inference collapses toward the

number of properties. Restoration properties had a median 44% higher adjusted cost per hectare (95% CI 5–99%), but this reflects the degraded, weed-affected context in which restoration is undertaken rather than the restoration actions themselves, which account for only about a sixth of their total planned budgets. Weed management (the largest and most variable category) is explained almost entirely at the property-level (ICC 0.85); site characteristics and the specific control methods explain only about 20% of cost variation across held-out properties. Because credit prices are fixed at the vegetation-class level and do not track the zone-level condition that drives per-hectare costs, more degraded sites are progressively less cost-effective to offset. Whether this cost-revenue gradient shifts where restoration occurs cannot be resolved from planned costs on an enrolled-only sample; it is, however, consistent with the self-selection described above, whereby the most degraded and most expensive prospective sites may never proceed to enrolment. Furthermore, because costs are planned at operational units that sit above the vegetation zones (at which gains are measured), our data cannot yield a cost-benefit assessment of restoration. Property budgets are, in effect, set property by property rather than built from transferable, condition-specific unit costs, so the cost-effectiveness of restoration relative to other management cannot be resolved at the scale at which it is planned.

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Figures

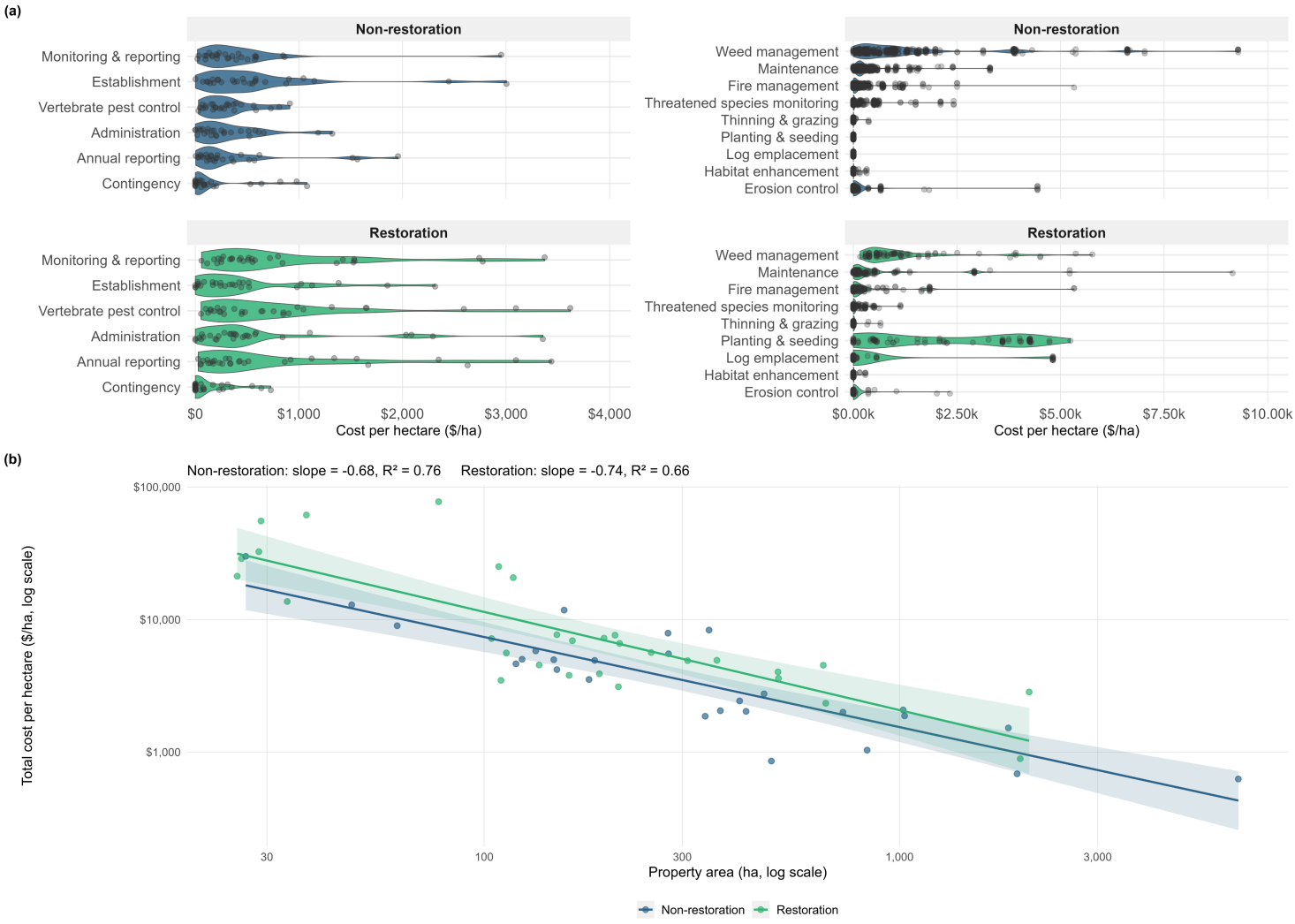


Figure 1

Overall summary of the cost relationship to area on the 57 sampled Biodiversity Stewardship Areas. A) Violin plots showing the distribution of cost in each cost category of the total fund deposit for properties with and without restoration. Costs are grouped by administrative and management costs. B) Relationship between the log-transformed property area and log-transformed total cost per ha for properties without (R^2 =0.76) and with restoration (R^2 =0.66).

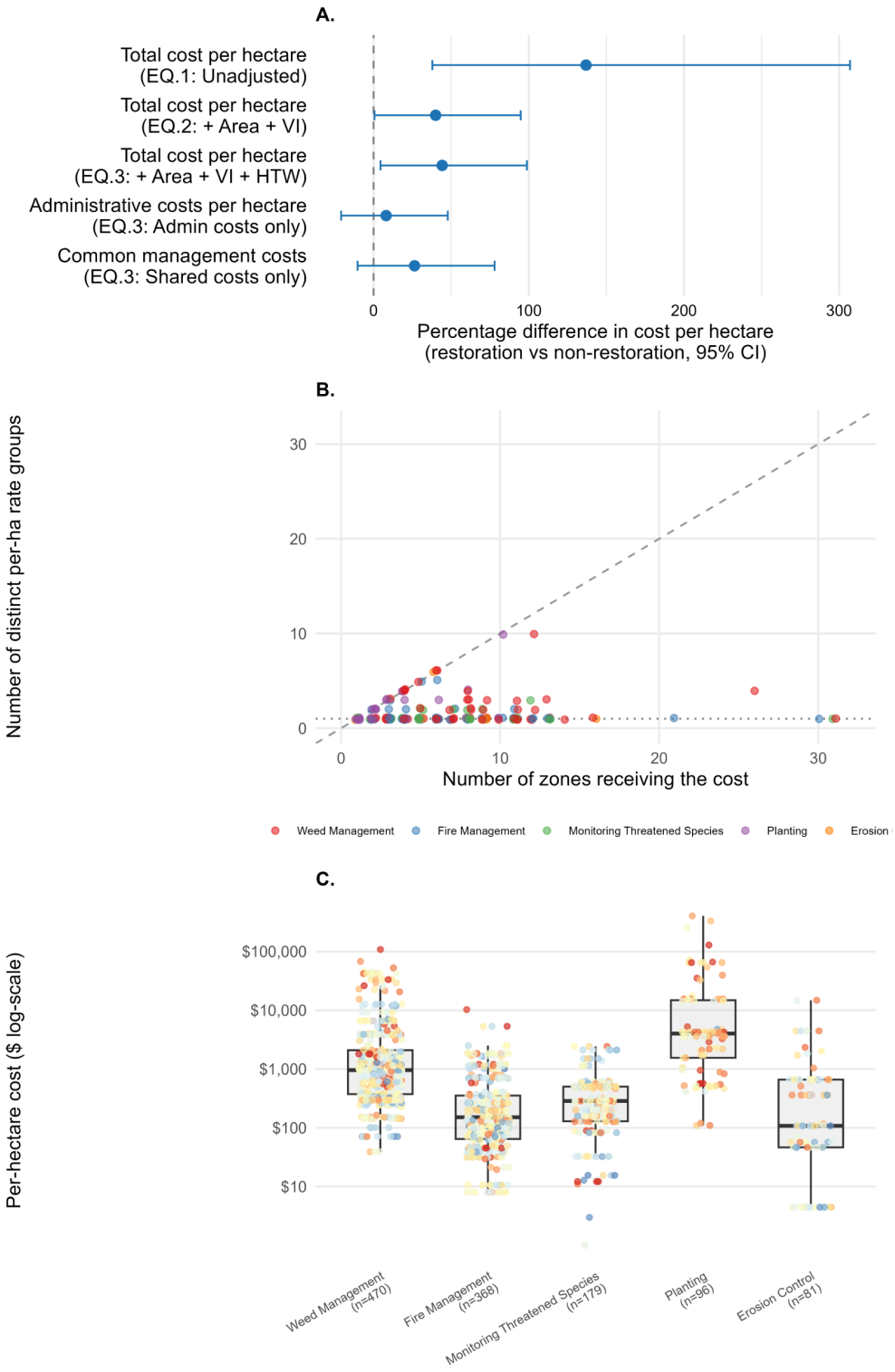


Figure 2

Patterns in planned costs across 57 Biodiversity Stewardship Areas. (A) Estimated differences in planned costs between properties with and without restoration. Point estimates represent the proportional difference in the median cost per ha from log-transformed OLS models (Eq. 1-3) for total, administrative and required management costs; horizontal bars indicate 95% confidence intervals derived from HC3 robust standard errors. (B) Scatterplot showing the number of distinct per-ha rates against the number of zones receiving each cost on a property, by cost category; points clustering on the lower reference line (y = 1) indicate a single allocated rate shared across a property's zones, whereas the diagonal reference line would indicate costs varying zone by zone. (C) Boxplots showing the distribution of log-transformed per-ha costs by cost category, with each zone coloured by starting vegetation condition (Vegetation Integrity Score); the absence of a clear colour gradient indicates that zone-level condition does not order zones by cost.

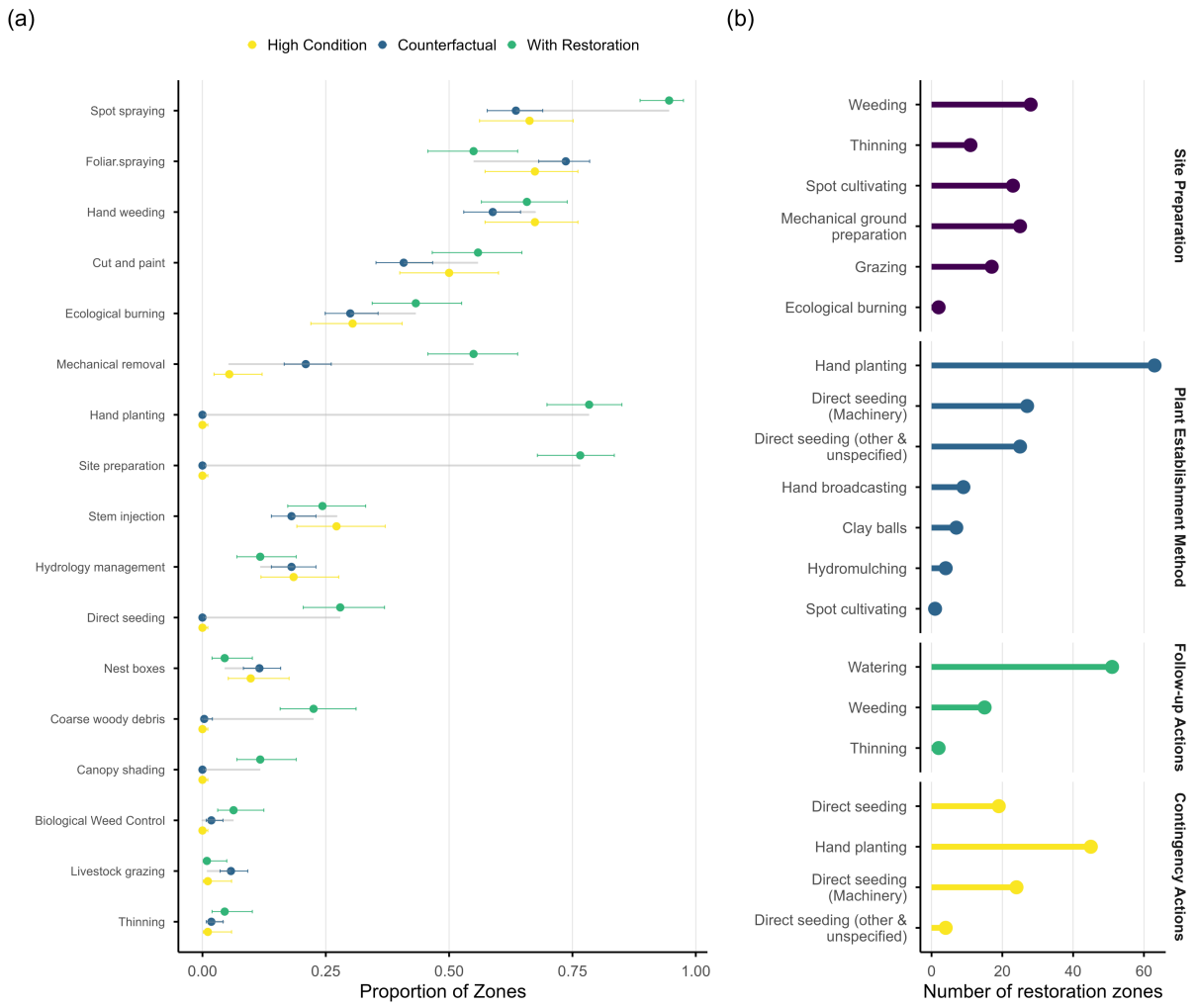


Figure 3

Analysis of management actions on BSAs. (A) Number of times individual management actions are used in counterfactual zones, high condition zones and zones with restoration. High condition (yellow) zones were categorised as having a VI>70. Counterfactual zones were derived from matching (Blue). Zones with restoration (green) were allocated active restoration actions and gains. (B) Number of times a restoration management action is recorded in each zone decomposed by sequential phases: site preparation, plant establishment, plant materials, follow-up management and contingency activities.

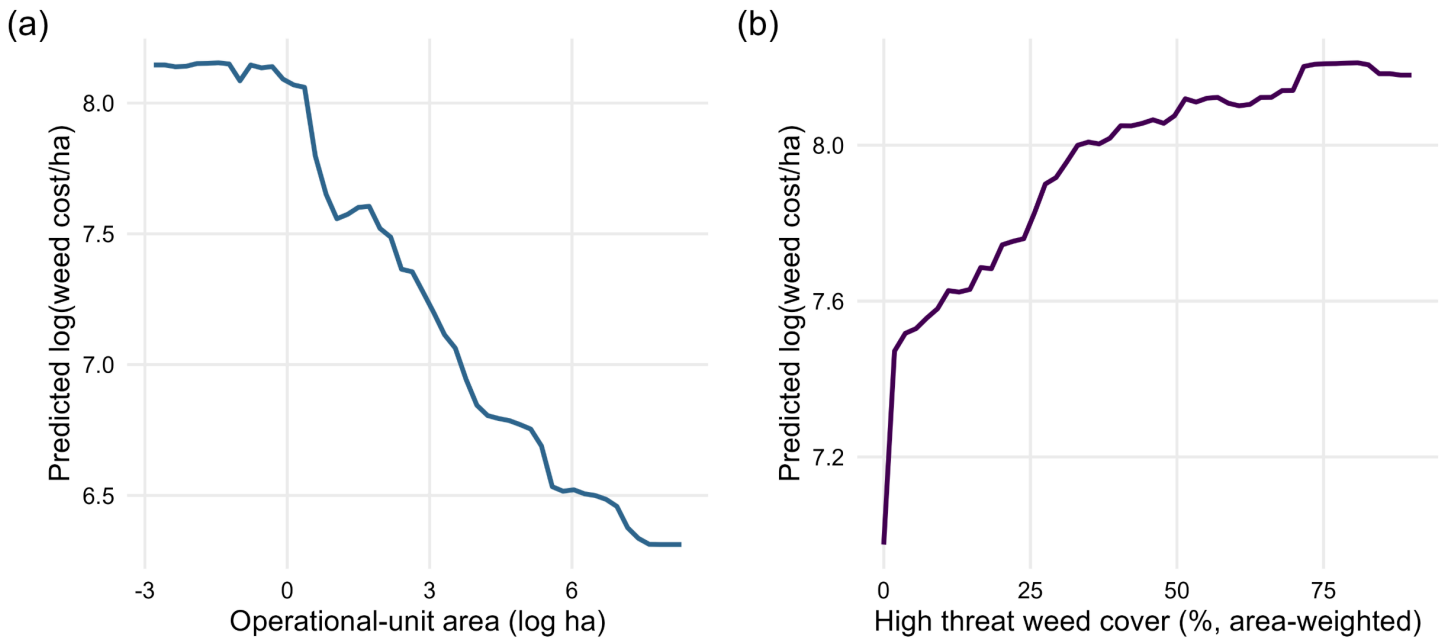


Figure 4

Partial dependence of predicted log weed cost per hectare on the two continuous predictors: (A) operational-unit area and (B) high-threat weed cover. Most weed-cost variation is structured at the property-level; within-sample variable importance is shown in Figure S11.