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Public investment does not crowd out private supply of environmental goods on private land

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ABSTRACT

In landscapes where private land tenure is prevalent, public funds for ecological landscape restoration are sometimes spent subsidising the revegetation of cleared land, and the protection of remnant vegetation from livestock. However, the total area treated may be unclear because such projects are not always recorded, and landholders may undertake similar activities without subsidisation. In the absence of empirical data, in the state of Victoria, Australia, a reporting assumption has been employed that suggests that wholly privately funded sites match publicly subsidised sites on a hectare for hectare basis (a so-called "x2" assumption). Conversely, the "crowding out" theory of investment in public goods such as environmental benefits suggests that public investment may supplant private motivation. Using aerial photography we mapped the extent of revegetation, native vegetation fencing and restoration on 71 representative landholdings in rural south-eastern Australia. We interviewed each landholder and recorded the age and funding model of each site. Contrary to the local "x2" reporting assumption, about 75% of the total area of the 412 sites was from subsidised sites, and that proportion was far higher for the period after 1997. However, rather than displacing unsubsidised activity, our modelling showed that landholders who had recently been subsidised for a project were more likely to have subsequently completed unsubsidised work. This indicates that, at least in terms of medium-term economic impact, the large increase in public subsidies did not diminish privately funded activity, as might be expected according to the theory of crowding out.

1. Introduction

In parts of the world where natural habitats have been extensively cleared and fragmented, governments and conservation organisations seeking to reverse the decline in the extent and quality of those habitats must do so on private land (Saunders et al., 1993; Foreman, 2004; Chazdon, 2008; Thackway and Lesslie, 2008; Duncan and Dorrough, 2009; Zerger et al., 2009; Hanley et al., 2012). Increasing the cover of structurally complex and diverse native ecological communities will help protect biodiversity, and may provide more environmental benefits and ecological resilience than relatively homogeneous production landscapes (Tscharntke et al., 2005; Fischer and Lindenmayer, 2007). These benefits are a public good because, once created, they are not for the exclusive use or consumption of the owner, nor can they be attached to property rights (Hanley et al., 2012).

A range of investment types has been developed to encourage private landholders or lessees, communities or non-government and quasi-non-government organisations to participate in landscape scale restoration schemes. These include grants and subsidies, to revegetate or restore depleted vegetation types; or legal instruments to mitigate threatening processes associated with human settlements and use, or their recent withdrawal. These approaches have been discussed in the literature under such labels as Agri-environment schemes (Kleijn and Sutherland, 2003; Whittingham, 2007; De Snoo et al., 2012; Hanley et al., 2012), Payments for Environmental Services (PES) schemes (Farley and Costanza, 2010; Miteva et al., 2012), Conservation Easements

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(Fisher and Dills, 2012), and Natural Resource Management (NRM) Programs (Hajkowicz, 2009; Pannell and Roberts, 2010).

In recent decades, Government agencies have begun to adopt increasingly sophisticated approaches to conservation, collectively termed systematic conservation planning (Kukkala and Moilanen, 2012), with increasing attention on where and how public money should be spent restoring ecological systems. For example, Governments want to invest in securing the most depleted or endangered habitat types under long term arrangements (Stoneham et al., 2003; Natural England, 2010; Boykin et al., 2011; Fisher and Dills, 2012), in sites in more favourable contexts, and in 'biodiverse' plantings (DSE, 2006). However, private landholders are needed to volunteer their land, and act effectively as co-investing partners. These private landholders may have various motivations for participating in programs or undertaking specific actions (e.g., Bramston et al., 2011; Sheeder and Lynne, 2011), or anticipate net private benefits (Pannell, 2008), some of which may be at odds with Government objectives (e.g., Pannell et al., 2006; Cocklin et al., 2007; Pannell and Roberts, 2010; Sorice et al., 2013). The suite of market-like mechanisms for investing in ecological restoration, such as conservation auctions explicitly acknowledge, and seek to exploit, the private co-benefits that may exist for participants (Stoneham et al., 2003; Cocklin et al., 2007).

The degree to which Government's strategic objectives are realised cannot be ascertained unless spatial data on landholder intentions, site locations and management is available. Prior to some recent examples (Von Hase et al., 2010; Raymond and Brown, 2011; Fisher and Dills, 2012), spatial records of where public money was actually spent have typically been patchy at best (ANAO, 2008; Bernhardt et al., 2005; Brooks and Lake, 2007; Zerger et al., 2009; Fero et al., 2013). Moreover, in addition to co-investing with government, landholders may undertake similar activities independent of direct public investment (Smith, 2008; Harris-Adams et al., 2012), and the location and amount of these activities is unlikely to be officially recorded. As a result, the evaluation of policy aimed at affecting positive landscape change is complex, as it may comprise known or unknown quantities in known or unknown locations.

In Victoria, Australia's most densely populated state, public policy to "reverse the long term decline in native vegetation extent and quality" began evolving around 1990 (East et al., 1996), as increasing grants for tree planting coincided with regulatory controls on clearing native vegetation from private land (Kyle and Duncan, 2012). Despite imperfect data, government agencies have a statutory responsibility to report on progress toward this objective within their jurisdiction (e.g., *Catchment and Land Protection Act* 1994 (State of Victoria)). Agencies have little choice but to resort to extrapolation from, and interpretation of, incomplete spatial data on vegetation management activities, together with assumptions about levels of unmapped activity, to estimate expected net spatial change or impact (Brunt and McLennan, 2006; DSE, 2008a). A key reporting assumption employed in Victoria in recent years is that for every hectare of publicly co-funded (i.e., government subsidised) activity, there is another hectare of privately funded (un-subsidised) activity elsewhere. This came to be known as the "x2" assumption (DSE, 2008a; GBCMA, 2008). Relevant empirical data for testing this assumption are few, and equivocal. Ambrosio et al. (2009) concluded that the assumption was justified, if not conservative, for landholders' conservation-oriented activity. Smith's (2008) survey of revegetation projects also found a substantial amount of privately motivated and funded work, although it decreased in proportion to publicly subsidised activity to almost entirely subsidised for the latter years.

There is a considerable body of theoretical and experimental literature from environmental economics and psychology concerning the relationship between intrinsic (private) and extrinsic motivation in the supply of public goods (Deci et al., 1999; Albers et al., 2008). This literature generally suggests a stable proportional relationship between co-funded and private activity is unlikely, because the public investment to increase the supply of restoration activity may 'crowd-out' private motivation to provide these services unassisted (Deci et al., 1999; Frey and Jegen, 2001; Bowles, 2008; Reeson and Tisdell, 2010). The possibility of crowding-out may be particularly apt in the context of this study. In 1997 the sale of a public utility led to the establishment of the Natural Heritage Trust; the largest-ever Australian investment in environmental programs, which was largely expended as small scale devolved grants to private landholders (Crowley, 2001; Hajkowicz, 2009).

In order to learn about how public and private interests had contributed to increases in native vegetation on private land – the public policy objective – we mapped extant relict/remnant native vegetation, naturally regenerating native vegetation and revegetation using native species on 71 representative landholdings in south-eastern Australia. We interviewed each landholder in order to characterise their socio-economic profile and enterprise type; and to determine the year, type, and resourcing model for each vegetation management activity carried out on their property. Our objectives were to establish who was undertaking these kinds of restorative works, particularly with respect to public and private funding; how much of different kinds of works landholders were undertaking; and how private investment may have changed over time in response to a substantial increase in public investment.

2. Material and methods

2.1. Case study areas

This study was located in three case study areas (Muckleford, Chiltern–Springhurst and Longwood Plains–Violet Town) in northern Victoria, Australia (a map is provided in Supplementary Material). These were broadly transitional, fragmented zones, which occurred between extremes of largely intact forest and relictual landscapes (*sensu* McIntyre and Hobbs, 1999) in the Goldfields, Victorian Riverina, and Northern Inland Slopes Bioregions, as defined by associations of landform, soils and vegetation (NLWRA, 2001). The socio-economic character of these areas has been broadly characterised as 'rural amenity' and 'rural transitional' (Barr et al., 2005). Formerly dominant farming practices such as livestock grazing are decreasing in area and intensity, whilst rural residential, peri-urban, wine and olive growing, and hobby farm uses are increasing, and pushing land values beyond their value for extensive grazing (Barr et al., 2005; Costello, 2007). Transitions from cropping and grazing to hobby farming and residential use via subdivisions are more common in the Muckleford and Chiltern–Springhurst case studies, whereas Longwood Plains–Violet Town retains a stronger focus on primary production.

Each of the case study areas has received considerable public investment aimed at vegetation protection and enhancement on privately owned land. Two main activities are used to increase native vegetation extent; the protection or enhancement of extant native vegetation; and the revegetation of formerly cleared land with indigenous species. Over recent decades these landscapes have also seen considerable spontaneous regeneration due to fewer producers, and a shift towards more intensive use of a smaller proportion of land area (Crosthwaite et al., 2008; Kyle and Duncan, 2012).

2.2. Selection of participants

We interviewed 71 landholders across the 3 case study areas with landholdings greater than 5 ha. We strove to include a broad range of rural landholders in our study across different parts of our study landscapes, and to minimise bias in our sample toward 'engaged' landholders. To that end, we stratified case study areas using a GIS derived "heat map" that represented the density of the cover of native tree revegetation and regeneration identified from aerial photograph interpretation (Kyle and Duncan, 2012). We calculated separate heat maps for revegetation and regeneration using the focal statistics tool in ArcGIS (Version 9.3, ESRI), which calculated a mean density value for each 10 × 10 m raster cell for circular neighbourhood of 3000 m. These grids were combined to produce four nominal classes of landscape cover: high–high, high–low, low–high and low–low revegetation and regeneration respectively. In each of our case study areas, regional contacts were then asked to help us locate landholders with property holdings in each cover class. Naturally, we were unable to completely avoid bias. Based on a new mapping comparison we were able to confirm that our sample of properties was representative of background levels of natural regeneration, but included some properties with considerably higher percentage cover of revegetation compared with background landscape levels (see Fig. 7, Kyle et al., 2012).

2.3. Mapping and interview

Immediately prior to visiting participating landholders' properties, external and internal boundaries and all zones of apparent woody native vegetation cover on their land were mapped from aerial photographs. On site, those vegetation features, plus the production management units (continuous areas of the property with uniform management), and details on the native vegetation units were recorded using an adaptation of the VegTrack database (Zerger et al., 2009). We distinguished vegetation zones that had undergone vegetation management works and been fenced out of the production units as "revegetation", "vegetation fencing" and "active vegetation restoration" (Table 1). Native vegetation regenerating naturally within production units but not fenced (unfenced regeneration) was recorded, but no data is presented here.

The main purpose of our interviews was to verify details about each mapped zone. First however, we gathered basic profile data on the age, length of time as owner and degree of reliance on the farm for income. Participants were asked to identify the production management units within their property (grazing, cropping, etc). Then, for each mapped vegetation zone, participants were asked to nominate the year the work was completed, and the funding source as "Private" (if the project was wholly privately resourced), or "Publicly co-funded" (if public funds were received). We asked them to nominate the relevant agency or environmental funding program as a crosschecking procedure. All personal data were collected and are maintained under the provisions of the *Information Privacy Act 2000* (State of Victoria).

We note that all of the public co-investment sites also involve private investment, coarsely estimated from our data to be 40–80% of material, capital and labour costs. However, they were treated as distinct because, in attracting public money, there will have been public-good criteria to be satisfied as well as the personal motivation of landholders.

Table 1
Definition of native vegetation works carried out on private lands. These represent mutually exclusive categories in the mapped data.

Type of work	Description
Vegetation fencing	Zones of native tree cover that have been fenced off but were not actively managed beyond that. Often the primary motivation for the landholder was to keep stock out of unproductive or dangerous land. Grazing may occur when feed was low elsewhere.
Active Vegetation Restoration	Zones of native vegetation cover that have been fenced-off and were actively managed to promote natural regeneration. Pest plant and animal control, strategic grazing and/or sub-zones of revegetation or supplementary planting were typically applied.
Revegetation	Zones of reconstructed native vegetation artificially established via planted seedlings or direct sowing of seeds. The sites were typically prepared with tillage and weed spraying.

2.4. Data analysis

In our presentation of summary data of the landholder sample and the total amount of native vegetation sites on their properties we used the landholder classification developed by Race et al. (2012) as an analytical filter. Their classification was developed following qualitative interviews with a representative sub-sample of 30 landholders from the present study. Race et al. (2012) distinguished between Full Time Farmers (FTF), Part-time Farming Landholders (PTFL), and Lifestyle Landholders (LL) on the basis of self-identification, size of property holding, and source of income. Such classifications, which are generated to delimit decision-making tendencies, for example; industry type and scale; value-orientation; and attitude to NRM issues, may help NRM practitioners to better understand their audience and therefore better target incentives and other programs (Vanclay et al., 1998).

As a means of assessing the strategic significance of publicly co-funded versus wholly privately funded sites we assessed the location of sites according to Bioregional Conservation Significance of Ecological Vegetation Classes, a 1:100 000 map for Victoria (DSE, 2008b). This dataset combines two spatial conservation planning tools, the Bioregion classification and a model of pre-1750 native vegetation type. Together, these surfaces represent relative levels of regional depletion of vegetation, compared with presumed pre-European settlement extent.

2.5. Modelling the effect of subsidy on landholders' private revegetation activity

Finally, to complement the analysis based on total area of revegetation across the 71 properties, we also modelled the impact of subsidies on the propensity for *individual* landholders to undertake privately funded revegetation. This analysis was limited to revegetation, as the other types of activity were too few to model. We modelled the probability that privately funded revegetation occurs on a given property in a given year between 1989 and 2008, as a function of the occurrence of publicly co-funded revegetation at the same property in the preceding five years. This timeframe allowed for 10 years of data either side of the NHT public funding increase. We also ran the models using subsidy received during the previous 10 years as a predictor, and the results were very similar.

The aim of this model was to test the assumption that increased public-funding for revegetation may 'crowd-out' wholly privately funded revegetation projects. Specifically, we asked whether previous public co-funding was negatively correlated with the propensity of landholders to undertake wholly privately funded revegetation at the level of the individual property owner. We controlled for property size; because larger properties tended to have more "sites", although they accounted for proportionally less of the property holding. Also, although we had no direct economic data for our participants, we wanted to include a predictor related to the likelihood that farmers would have surplus profit to spend. We chose to use data on the indexed wool price from the previous year (sourced from the Australian Government Department of Agriculture, Fisheries and Forestry website), because it is traditionally one of the most important export commodities in rural Australia (Productivity Commission,

2005) and has been implicated in patterns of native vegetation management in our study area (Duncan et al., 2010).

We implemented the model in a Bayesian framework using the open-source software packages R (R Development Core Team, 2010), and JAGS (Plummer, 2003). The model fit was a hierarchical logistic regression (Gelman and Hill, 2007), where the probability of a landholder undertaking an i^{th} privately funded revegetation work, $Pr(Y_i = 1)$, at the j^{th} property during the k^{th} year, is modelled as a linear function with a logit link.

Not all property by year combinations exist in the dataset as some landholders took on their properties after 1989, and therefore the total number of potential i^{th} projects, is less than the number of years times the number of properties.

The fitted model was of the form

$$\begin{aligned} Pr(Y_i = 1) &= \text{logit}^{-1}\left(\mu + \beta_1 F_i + \gamma_{j[i]} + \delta_{k[i]}\right), \quad \text{for i} = 1,\ldots,1075 \\ \gamma_j &= \beta_2 S_j + \epsilon_j, \quad \text{for j} = 1,\ldots,71 \\ \delta_k &= \beta_3 W_k + \eta_k \quad \text{for k} = 1,\ldots,20 \\ \epsilon_j &\sim N(0, \sigma_\epsilon) \\ \eta_k &\sim N(0, \sigma_\eta) \end{aligned}$$

where Y_i is a vector of response data denoting whether a private-funded revegetation work occurs. The probability a work occurs, $Y_i = 1$, is then the inverse logit of a linear equation that includes an intercept term, μ, plus an effect of past public co-funding, $\beta_1 {}^* F_i$. Here F_i is a vector of binary data denoting whether a publicly co-funded project occurred at the properties in the preceding five years and β_1 is the coefficient indicating the magnitude and direction of the effect. The linear equation at the highest level of the model also included terms accounting for property-level ($\gamma_{j[i]}$) and inter-annual ($\delta_{k[i]}$) variation, with the square bracket notation in the indices used to indicate that *j* and *k* are partially nested within *i*. The sub-models for property-level and inter-annual variation have two components each. The property-level sub-model includes an effect of property size, $\beta_2 S_j$, where S_j is a vector of farm sizes in hectares, and β_2 a regression coefficient plus unexplained extra variation, ε_j. And similarly the inter-annual sub-model includes an effect of indexed wool price in the preceding year (as an indicator of economic capacity), $\beta_3 W_k$, with W_k a vector of prices in dollars and β_3 another regression coefficient plus unexplained inter-annual, η_k. The property-level and inter-annual error terms, ε_j and η_k were modelled with normal distributions centred on zero and with estimated standard deviations, σ_ε and σ_η.

Following the recommendation of Gelman et al. (2008), a Cauchy distribution with a scale parameter of 25 and mean of 0 was used as a weakly informative prior for the intercept term, μ, whereas weakly informative Cauchys with scale parameters of 10 and mean 0 where used for the regression coefficients β_{1-3}. The standard deviations of the property-level and inter-annual error terms had weakly informative half-Cauchy priors with mean of 0 and scale parameters of 25 (following Gelman, 2006).

The model fit was run using 3 chains with each chain run for 10 000 iterations with the first half of each chain discarded as burnin. Convergence was assessed with visual inspection of trace plots and on the condition that the potential scale reduction factor was <1.1 and effective number of posterior samples >100 for each model parameter.

All covariate data was centred on zero and scaled by multiplying by two standard deviations. Scaling by two standard deviations made the covariates equivalent to an evenly distributed binary variable and meant the magnitude of each regression coefficient was directly comparable to the others (Gelman, 2008).

To determine the relative importance of the model covariates and unexplained property-level and inter-annual variation we undertook a variance components analysis using the method outlined in Hector et al. (2011; see also Gelman, 2005). We compared the magnitudes of the five variance components, effects of previous co-funding, property size and wool price, and unexplained property-level and inter-annual variation, on the standard deviation scale. To calculate the components explained by covariates we multiplied the absolute effect-size by the standard deviation of the input variables (in each case this was ~0.5 due to the scaling of the covariate data). To calculate equivalent components for the unexplained variation we used the standard deviations of the parameters ε_j and η_k, the finite population standard deviations, rather than use the parameter estimates for σ_η and σ_ε, which are the super-population standard deviations.

3. Results

The 71 landholders had undertaken projects at a combined total of 401 sites totalling 3264 ha between 1961 and 2009. Seventy-two percent of the sites (289) were publicly co-funded. Works ranged in

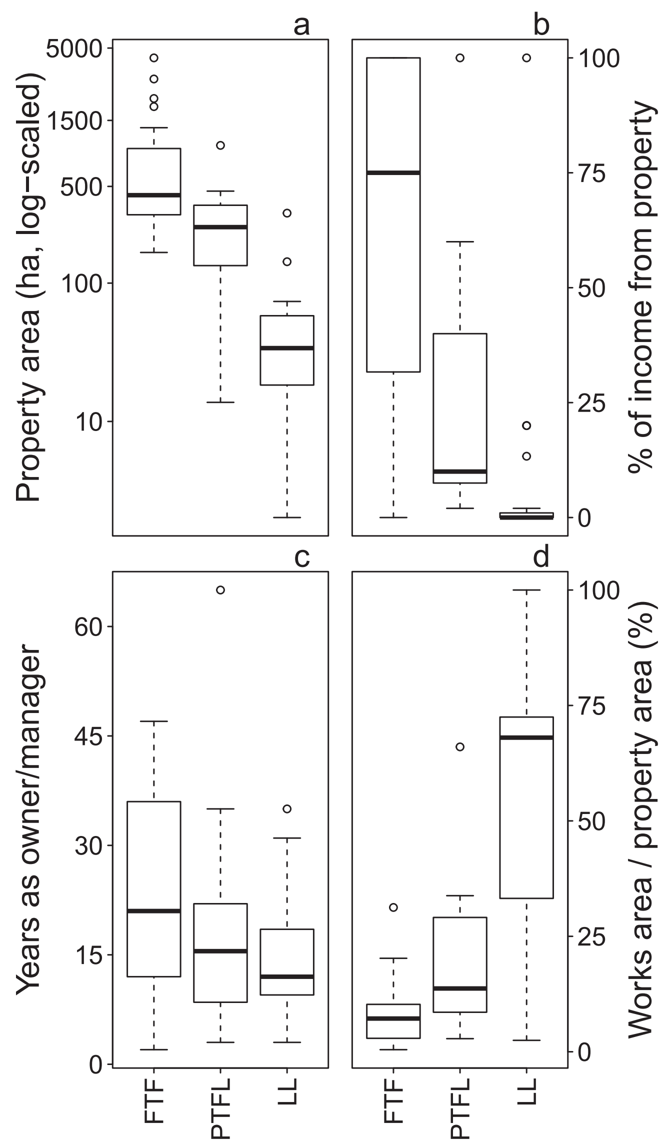

Fig. 1. Summary personal and property attributes of the sample of 71 landholdings, classified according to Race et al.'s (2012) framework of Full Time Farmers (FTF, $n = 31$), Part-time Farming Landholders (PTFL, $n = 21$) and Lifestyle Landholders (LL, $n = 19$).

size from a 0.1 ha revegetation site, to a 164 ha vegetation fencing project (median 2.9 ha). The interview sample included 31 Full Time Farmers, 21 Part-time Farming Landholders and 19 Lifestyle Landholders, according to the classification of Race et al. (2012). The dominant land use on the properties were sheep grazing (28%), cattle grazing (25%) and mixed farming (livestock plus other enterprise/s, e.g., cropping, 25%), with 21% used for residential purposes only. Most residential properties belonged to Lifestyle Landholders who used their property for a residence or non-farming place of work only, although a few had some sheep, cattle or small plantations and crops (e.g., fruit trees, vines, hardwoods).

Property holdings varied in size from around 5 ha for residential blocks to farms over 3000 ha (Fig. 1a). The median property size for Full Time Farmers was nearly 500 ha, higher than the other cohorts. On-farm income as a percentage of household income ranged from 0 to 30% for Lifestyle Landholders (median = 0) and from 0 to 100% (median = 75) in the case of Full Time Farmers (Fig. 1b). Most surveyed lifestyle and part time farming landholders had been owners or managers of the land in question for 10–22 years, somewhat longer for the Full Time farming cohort (Fig. 1c). The proportion of property area that consisted of active native vegetation management restoration varied from 0 to 100% (Fig. 1d). Although the larger holdings of Full Time Farmers had proportionately less land under native vegetation works, the median value for the Full Time Farmer group was still around 10%.

The 289 publicly co-funded sites in our sample accounted for 2600 ha, compared with 112 sites of privately funded works accounting for over 600 ha (Fig. 2). Thus, over the entire dataset there was a ratio of more than 4 ha of co-funded sites to each 1 ha private. The bulk of the area of publicly co-funded and privately funded sites was comprised of remnant fencing sites, whereas in both cases revegetation sites were far more numerous. This reflects the fact that fenced-off remnant sites on average were considerably larger than revegetation sites (Fig. 3).

Publicly co-funded revegetation sites were significantly larger on average than privately funded revegetation sites (Welch two sample t-test on log scaled data, df = 126.5, $P < 0.0001$, 95% Confidence Interval on true difference in means 2.5–5 ha; observed medians 2.7 ha compared with 0.73 ha; see Figure in Supplementary Material). Co-funded and privately funded active remnant restoration (both medians 5.6 ha) and remnant fencing sites (co-funded median 5.7 ha, private 4.8 ha) were similar in size regardless of the funding model, although in each case there were more smaller privately funded sites and the largest few sites tended to be publicly co-funded. A generalised linear model of log site area (ha) as a function of site type × funding source, and year, confirmed that revegetation sites are smaller than remnant restoration and fencing sites; that private revegetation sites were smaller than co-funded sites; and that the pattern did not change over time (years). Therefore, co-funded revegetation sites were always bigger than privately funded sites but the gap did not increase over time.

All sites established on these landholdings from 1961 to 1974 were privately funded and privately funded totals exceeded co-funded totals for almost half of the subsequent 20 years. A substantial increase in publicly co-funded hectares per year was evident in the years following the launch of the Australian government's NHT_{I} (1998–2002) and NHT_{II} (2002–2008) funding initiatives (Fig. 3). Although the contribution of private sites was proportionally dwarfed over that period, these data indicate no decline in private hectares per year. The amount of privately funded work was more variable year to year but increased to a relatively stable average of approximately 20 ha/yr over the 71 landholdings by the early 1990s.

This pattern held true when the data were broken down into individual landholdings. Our hierarchical model showed that those who had received funding for a revegetation project within the last 5 years were more likely to undertake an unfunded revegetation activity than those who had not (model ROC = 0.7988). The variance components analysis, shown in Fig. 4, demonstrates that the explanatory power of the effect of previous co-funding is as important, if not more so, than the size of the holding, or wool prices from the previous year, a plausible surrogate of economic

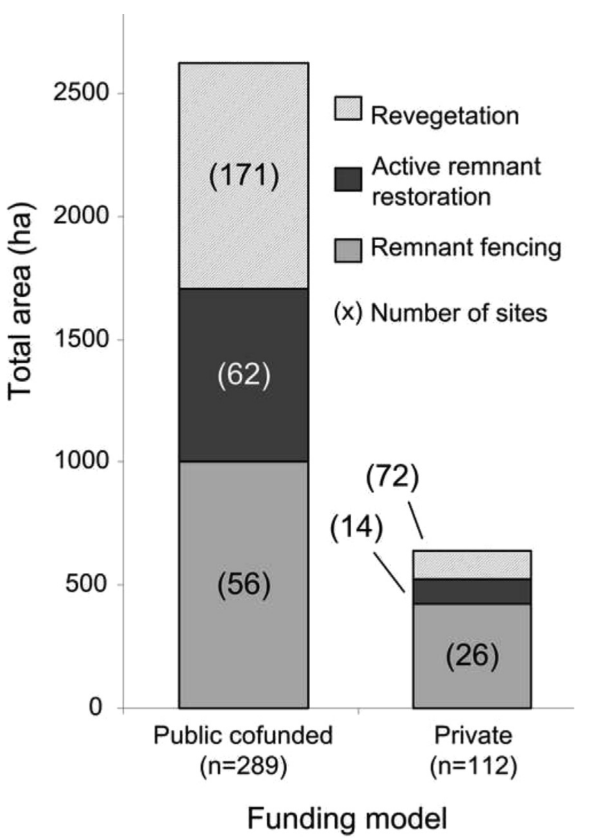

Fig. 2. Summary of publicly co-funded and privately-funded revegetation, remnant fencing and active remnant restoration sites for 71 landholdings.

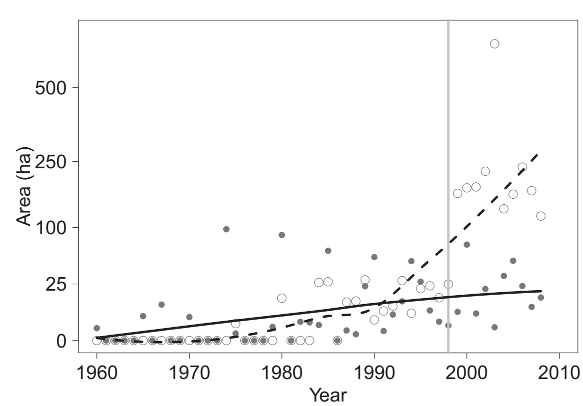

Fig. 3. Total area (ha) of privately funded (filled symbols, solid spline) and publicly co-funded (open symbols, dashed spline) per year, combining revegetation and native vegetation sites. The Y-axis is the natural log of area with back-transformed labels. The commencement of the NHT funding program in 1998 is marked with a vertical reference line.

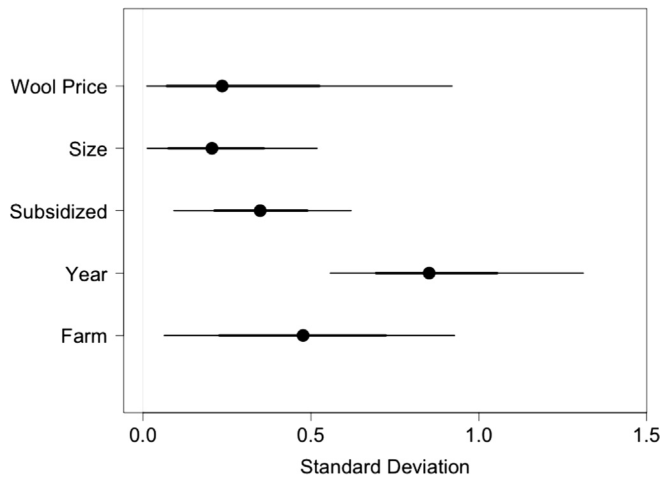

Fig. 4. Variance components from the model of the probability of privately funded revegetation. Components are shown on the standard deviation scale. Black circles indicate the medians of the posterior distributions with thick lines spanning the inner 68% credible interval and the thin line spanning the 95% credible interval.

position. The model estimates ($\beta_{1-3,}$ Table 2) demonstrate that the effect of previous co-funding is positive. In contrast, the estimated effects of property size and wool prices are only weakly positive and possibly zero. In a given year, the average landholder who has recently received subsidy for a project is still less than 10% likely to undertake a privately funded activity (Fig. 5). Indeed, even in concert, these predictors of the probability of privately funded revegetation do not explain the majority of variation either inter-annually or at the property level (Fig. 4).

As well as being often larger, we found that publicly co-funded sites tended to have been established in higher Bioregional Conservation Status areas than private funded sites (Fig. 6). There was little difference between co-funded or private sites in the lower BCS categories, where the prevalence of BCS categories associated with works was roughly proportional to their property level prevalence. However, on average, publicly co-funded sites in the Vulnerable class were established in proportion to their prevalence whereas privately funded sites were 20% below proportional. While both publicly co-funded and privately funded works occurred in land classed as Endangered at lower than proportional rates, the proportion of publicly co-funded sites was often considerably higher than privately funded sites.

4. Discussion

Our survey quantified the amount of native vegetation fencing, restoration and revegetation on 71 landholdings, undertaken over the preceding 50 years, and the underlying funding model for those sites. By comparing publicly co-funded sites (cost-share with landholders) against wholly privately funded sites we addressed several key assumptions relating to the theory and practice of public investment in the natural environment.

Table 2
Standardised regression coefficients and standard errors of the probability of privately funded revegetation model.

Coefficient	Covariate	Standardised effect size	Standard error
β_1	Previous co-funding (5 yrs)	0.619	0.292
β_2	Farm size (ha)	0.179	0.341
β_3	Wool price previous yr	−0.037	0.689

4.1. *Times-two is too much, but private activity may not be crowded-out*

First, the assumption employed by regional and state government in Victoria (Brunt and McLennan, 2006; DSE, 2008a), that each publicly co-funded hectare of native vegetation restoration or revegetation is matched by a privately funded hectare, was refuted by these data. Estimates of total area of restoration activity based on the "x2" assumption will substantially overestimate the number of hectares of private activity occurring each year. The provenance of these data must be borne in mind. Our study landscapes are broadly characterised as rural-amenity, or in transition (Barr et al., 2005), with ever more lifestyle landholders and fewer full time farmers (Race et al., 2012). Wholly private investment in revegetation and restoration activities will vary locally (beyond our sample landholdings), but perhaps even more beyond our case study; both in gross area terms, and in proportion to publicly subsidised activity. Further work to explore how these patterns vary with the landholder and dominant industry characteristics will be welcome, but in the meantime, continued uncritical use of the "x2" assumption is dubious. Whilst our findings disagree strongly with the specific application of the "x2" assumption, as we discuss later, they do lend qualified support to the notion that private landholders have matched the Government in its "biggest ever environmental investment".

Particularly since the late 1990s, wholly privately funded works were proportionally dwarfed by the increase in public funding for private land conservation through major federal investment programs like Natural Heritage Trust $_{\mathrm{I\ \&\ II}}$, and the National Action Plan for Salinity and Water Quality. Smith's (2008) analysis of revegetation activity in a Western Australian wheatbelt case study also showed a decline in the proportion of wholly privately funded work through the early- to mid-1990s. A body of theory, experimental and field data from diverse contexts and participants suggest that where an institution introduces monetary reward or inducement for a public good, it may crowd-out private motivation to supply the good voluntarily through triggering and rewarding market-like behaviour (Deci et al., 1999; Frey and Jegen, 2001; Reeson and Tisdell, 2010). With this in mind, it seems implausible that voluntary private investment in a restoration and revegetation activities would ever have been expected to keep pace with an unprecedented increase in the scale of public investment. Race et al. (2012), reporting on semi-structured interviews with a subsample of our landholder cohort, found that 25 of 31 respondents needed financial help from government to undertake more restoration works, even on a small scale. This observation lends support to the idea that some expectation had been created by increased availability of public money. However, for a number of reasons, our data do not conform to what might be expected under a simple crowding-out scenario.

We have demonstrated that whilst the publicly subsidised hectares-per-year surged, private landholders as a group continued to establish new sites of their own accord at modestly increased rate. Individually, they were more likely to undertake unfunded work if they had been the recipient of a subsidy for a similar activity within the previous five years. Furthermore, given that each co-funded site relied on private co-contribution in the vicinity of dollar for dollar, the total contribution of private capital, including co-contributions, increased at the same rate as the injection of public money in NHT. Does this reflect a remarkable achievement of leveraging public good from private investment, a win–win for society *and* landholders, or perhaps that the public money has been captured for considerable net private benefit (*sensu* Pannell, 2008)?

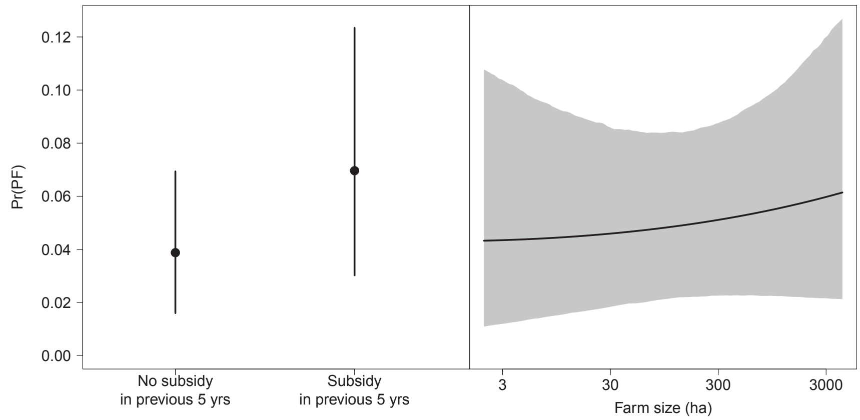

Fig. 5. Comparison of the effect of co-funding in the previous five years with the effect of farm size on the probability of a privately funded revegetation at average property in an average year.

4.2. Is ecological restoration best viewed as a public or private good?

Deci et al.'s (1999) meta-analysis of the crowding out literature highlighted that positive feedback can increase the voluntary supply of public goods. It is possible therefore that the strong positive narrative about landscape restoration from community and government in recent decades, combined with commitment of significant funds, may have increased the non-monetary reward for stewardship behaviour (e.g., Bramston et al., 2011) to counter any crowding-out effect. However, because material improvements to individual's landholdings may result from these activities (new fences, improved aesthetic qualities), participants may also anticipate and participate for net personal benefit. Pannell's (2008) private–public benefit framework illustrated how different projects could be situated in benefit space anywhere from predominantly private benefit to predominantly public. Presumably, the same variation could be true of individual participants or subscribers within a given project (see Dettmann et al., 2000). Some production-oriented respondents in our study offered that they participated in remnant vegetation protection schemes because it was a way to get a new fence, which would make livestock management easier. In that instance, the motivation of government as purchaser and landholder as participant are complementary, but discreet. Revegetation and restoration activities are complex joint-products (*sensu* Hanley et al., 2007); comprised of some mixture of private goods (e.g., farm business, land value) and non-rival and non-exclusive public goods (e.g., beauty, contribution to species persistence, landscape resilience).

In addition to the theoretical explanations for why co-funded and privately funded continued in parallel, there are practical factors relevant to our example that could be important. In the past decade for example, eligibility criteria have changed and schemes are generally more competitive. In Victoria, public funds are generally no longer available for revegetation projects less than 20 m wide. These sites are expected to have a lower probability of sustained biodiversity value, so wholly private funding might be the only means of completing such a project. Similarly, in competitive bidding processes, if there were sufficient supply of willing landholders then some otherwise eligible projects may miss out where the conservation benefit is calculated to be lower (e.g., Schilizzi and Latacz-Lohmann, 2007).

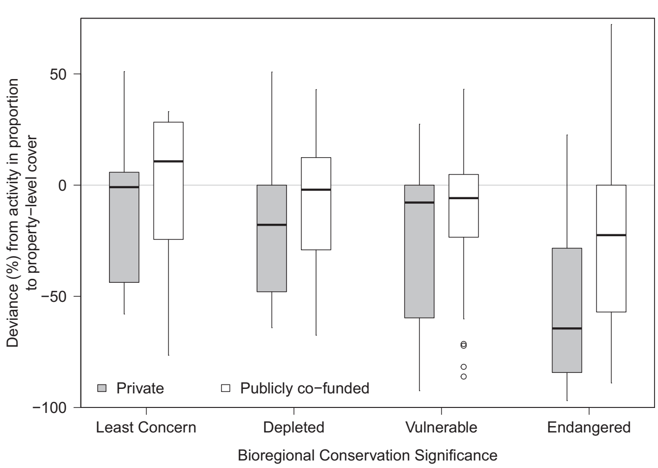

Fig. 6. A comparison of the deviance in private (filled) and publicly co-funded (open) native vegetation restoration activity in each BCS category, from proportional to the pre-1750 prevalence of that BCS category (LC = Least Concern, D = Depleted, V=Vulnerable, and E = Endangered). Deviance is given by the % of property originally occupied by each BCS category subtracted from the % of total restoration works on the property in each category. For example, if 50% of a property area was originally in the Endangered category, and 50% of the area of restoration and revegetation sites had also occurred in the Endangered category, the deviance would be 0.

4.3. Public investment leads to larger sites and higher conservation significance

If public investment was merely paying landholders to do what they were going to do already, there should be little distinction between the attributes of co-funded and privately funded sites. By contrast, our data provided two examples of public funding achieving outcomes distinct from those that landholders might otherwise fund themselves. First, public co-funding can entice landholders to release sites in areas of higher conservation significance areas that otherwise are not volunteered. Von Hase et al. (2010) recently found evidence for the alignment of subsidised conservation activity on private land with strategic priorities for the Cape Region of South Africa, but our data illuminate the contrast between subsidised and unsubsidised activities. Given that degree of depletion of native vegetation communities (negatively related to conservation significance) tends to be positively related to land productivity, it makes sense that landholders would be less inclined to voluntarily remove such areas from productive use (Raymond and Brown, 2011). Fisher and Dills (2012) also showed that higher priority areas tend to cost more to protect in their analysis of a portfolio of conservation easements (legal covenants) and fee simple acquisitions (land purchases) of The Nature Conservancy (www.nature.org). Fee simple purchases, 2–3 times more expensive to establish than easements, were located within strategic priority areas in 86% of cases compared with 64% of easements. In our study, publicly co-funded sites also tended to be larger in area, and this pattern was more pronounced the more cost-intensive the activity. Publicly co-funded revegetation sites were more than four-times larger on average than privately funded sites.

4.4. Limitations as a test of crowding out

These data provided important, empirical insights into the relationship between private and public funding in ecological restoration, in the context of a quantum increase in public investment, but it is by no means a definitive test of the crowding out hypothesis. First, our data are circumstantial; we have no counterfactual evidence about patterns of privately funded activity that might have occurred in the absence of a major increase in public investment. Secondly, although we were able to construct a socio-economic profile of our landholder sample, it remained unclear to what extent our three sampling areas more represent replicates or contrasts with respect to the response variable, which is about what landholders do. More broadly, our quantitative analysis lacked direct insight into landholders' motivations; therefore it may not predict the response to future patterns of public funding. For example, what might happen if public investment in ecological restoration returned to pre-1997 levels? One fear regarding the major intervention of institutions and market instruments into areas of voluntary activity for public environmental goods is that it may disrupt the social fabric and networks of people who organise to undertake such activities. It may also catalyse a lasting switch toward more selfish, market-oriented behaviour, and less pro-social behaviour (Reeson and Tisdell, 2010). If that is the case it suggests that the current levels of public investment may need to be maintained indefinitely, or even increase.

5. Conclusions

Our study has provided critical insight into the effect of government investment both on the environment itself, and on private investment in the environment. We showed how the patterns of public and private investment have changed in our case study areas since the earliest recorded sites in 1960, and how objectives for undertaking such works have also changed. Insights such as these could become routinely available to natural resource managers and investment planners if the spatial recording of publicly co-funded and privately funded works, along with appropriate socio-economic data, were adopted. Given that land-use patterns, landholder communities and motivations are far from static, it will be important to refine, repeat, update studies such as this. If not, we will remain vulnerable to unreliable and untested assumptions about what, where and why people are engaging in vegetation management as a basis for planning, investment and reporting of impact.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2014.01.041.

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