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Optimal conservation investment for a biodiversity-rich agricultural landscape*

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This study develops a theoretical and empirical framework for optimal conservation planning using satellite land cover data and economic data from a farm survey. A case study is presented for a region within the South-west Australia Biodiversity Hotspot (*Nature* 403, 853). This Biodiversity Hotspot is a focus for conservation investment as it combines a relatively high level of biodiversity with severe threat to the biodiversity from agriculture. The conservation planning model developed determines the optimal set of bush fragments for conservation. This model can also be used to assess the trade-off between the budget and a vegetation species metric. Results from the case study show that, without an effective conservation scheme that at least fences fragments, significant plant biodiversity losses will occur in the North East Wheatbelt Regional Organisation of Councils region of the WA wheatbelt over a 10-year period. A perfect price discriminating auction scheme could reduce the costs of conservation by around 17 per cent relative to a fixed-payment scheme; however, a fixed payment on outcome (measured as change in the species metric) scheme represents a viable second-best alternative, to a conservation auction, where conservation spending is spatially targeted.

Key words: biodiversity, conservation planning, Western Australian wheatbelt.

1. Introduction

The conservation investment problem may be construed as one of allocating limited public funds between assets that change stochastically through time in response to management actions and the environment. The elements of a voluntary conservation investment scheme for bush fragments include a conservation contract that provides an incentive to conserve bush, a measure of conservation outcomes, that is a species (or biodiversity) metric and a monitoring scheme to either determine non-compliance or assess payment (Moxey *et al*. 1999). Conservation contracts can be classified as input-based, outcome-based or a mixture of the two. Input-based contracts specify conservation actions (for instance, reduce fertilizer, fence bush, retire land from

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agriculture or replant native plants) in return for a payment when actions are completed. Outcome-based contracts specify an environmental outcome (for instance, a measure of biodiversity or water quality) and base payments on the outcome. Most agri-environmental policies use input-based contracts, for instance, the United States Department of Agriculture Conservation Reserve Program (Reichelderfer and Boggess 1988), the Victoria Bushtender Scheme (Stoneham *et al.* 2003) and the UK Environmental Stewardship scheme (DEFRA 2010). A relatively small, but increasing, number of schemes pay landholders for environmental outcomes, see Zabel and Roe (2009) for a review.

The aim of this study is to develop a realistic conservation planning model that satisfies a number of characteristics: first, it is based on readily available data; second, it is consistent with ecological principles; third, it provides landholders with clear incentives for conservation; and fourth, it is not vulnerable to rent-seeking costs associated with asymmetric information in the form of hidden action and hidden information (Ozanne and White 2008). The model developed is a mixed contract that includes an input-based payment for fencing, that is observable, and an outcome-based payment on the change in a species metric.

The remainder of the study is organized as follows. The next section is a brief literature review. Section 3 sets up the theoretical model. Section 4 describes the North East Wheatbelt Regional Organisation of Councils (NEWROC) case study, Section 5 presents results, and Section 6 concludes.

2. Literature review

This literature review considers the key ecological and economic attributes of conservation investment: measuring environmental and biodiversity outcomes; principal-agent relationships; effort contracts on stochastic outcomes; and optimal investment in multiple assets.

In conservation ecology, there is a trend towards the greater use of mathematical decision theory. This trend is also linked to the development of systematic conservation planning (SCP) (Margules and Pressey 2000) and specialist optimization algorithms such as Marxan (Ball *et al.* 2009), capable of finding acceptable solutions to an integer programming problem termed the set covering or knapsack problem. Marxan finds an approximate least-cost conservation plan.

Conservation contracts based on the principal-agent model focus on incentives for participation under adverse selection (Wu and Babcock 1996; Moxey *et al.* 1999) and optimal monitoring to address moral hazard in the form of hidden actions (Ozanne and White 2008). Compared with planning models such as Marxan, these models include simple representations of ecosystems. However, planning models do not account for incentives and underestimate costs on private land. In particular, required transfer payments on private land tend to be increased by information asymmetries in the form of hidden action (moral hazard) and hidden information (adverse selection) typically relating to landholder compliance costs.

Here, a regulator contracts with farmers to conserve a bush fragment, through effort and fencing. Outcome-based conservation contracts, where the farmer is paid on the basis of a species (or biodiversity) metric, have much in common with labour contracts. In particular, the outcome variable, here the change in a species metric, in labour economics some measure of performance, is highly variable across fragments and through time. In the context of labour contracts, Zhao (2008) shows that, where it is not possible to monitor all actions, it is better to base rewards entirely on stochastic outcome measures. This describes the situation with conservation contracts: where a farmer's effort is impossible to observe, but it is possible to measure an increase in species protected through a surrogate species metric.

In terms of investing in multiple assets, Loch and Kavadias (2002) characterize the firm as allocating a fixed budget between competing projects:

> 'Optimal portfolios are difficult to define because of the combinatorial complexity of project combinations. However, at the aggregate level of the strategic allocation of resources across product lines, investment in a program is not an all-or-nothing decision, but can be adjusted, resulting in a higher or lower program benefit. (Loch and Kavadias 2002, p. 1227)'

They propose a marginal condition that can be applied to conservation investment where the regulator allocates a budget between fragments.

3. Ecological-economic model

The theoretical model represents two components: an ecological model and a farmer incentive model. These are combined as the regulator's problem where a biodiversity objective is maximized subject to the expected change in a species metric owing to fencing and conservation effort, the farmer's incentives to participate, and apply an optimal effort.

3.1. Ecological theory, the species-area relationship

In ecology, a species-area relationship or curve is a relationship between the area of a habitat and the number of species. Since MacArthur and Wilson (1967), it has been widely accepted as an ecological law. For instance, Keeley and Fotheringham (2003) review species-area relationships (SAR) for vascular plants in Western Australia and found it followed a semi-log function:

$$\tilde{S}^k(A^k) = \alpha^k + \beta^k \log A^k \tag{1}$$

where $\tilde{S}^k(A^k)$ is the maximum number of plant species (or species metric if the total number of plant species in a set of fragments is not known) from vegetation type k from total area, A^k. The parameters α^k and β^k are specific to the vegetation type and can be estimated from field sampling. Equation (1) is illustrated in Figure 1.

The species value of a bush fragment in a landscape is the number of unique species it contributes to total biodiversity or its marginal species. The marginal species $\tilde{S}_i^k$ for fragment i is the number of species lost by removing the fragment from the 'archipelago' or landscape of fragments (Moilanen 2007):

$$\tilde{S}_i^k = -\beta^k \log(1 - h_i^k) \tag{2}$$

where $h_i^k = A_i^k / A^k$ is the proportion of the total area A^k of vegetation type k covered by fragment i (of size A_i^k). The marginal species $\tilde{S}_i^k$ gives the maximum potential marginal species contributed to the landscape by a fragment of a given size, as defined by the SAR, regardless of the degradation within the fragment or across the archipelago. This is illustrated in Figure 1 as the reduction in species owing to removing fragment A_i^k from the archipelago.

The actual contribution of a fragment to a landscape's vegetation biodiversity depends upon its 'condition'. We define a vegetation condition metric as the proportion of species observed in a bush fragment compared with the expected number of species in an equivalent, but undegraded, fragment of the same vegetation type and area. The marginal species value for a fragment is adjusted by a vegetation condition metric μ_{it}^k at time t, with $0 \leq \mu_{it}^k \leq 1$ according to:

$$S_{it}^k = \mu_{it}^k \tilde{S}_i^k \tag{3}$$

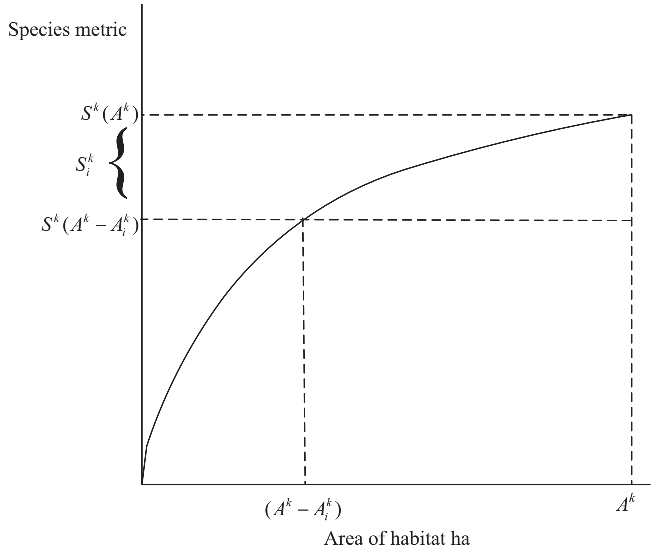

Figure 1 Species-area relationship.

The marginal species value S_{it}^{k} is relatively high for rare vegetation types in good condition on large parcels of land and low for common vegetation types in poor condition on small parcels of land. Using (3) as the basis for a plant species metric is problematic, given the non-linear and non-separable functional form of (1), when the total area is made up of a large number of fragments in different conditions. Equation (3) offers an approximation when the area of fragments is fixed and the condition μ_{it}^{k} is variable across fragments.

Thus, the following species metric is proposed. In aggregate, summing over vegetation types and fragments, the species metric at a point in time is:

$$\sum_{k}\sum_{i}\mu_{it}^{k}\tilde{S}_{i}^{k} \tag{4}$$

Over a planning horizon from $t = 0$ to $t = T$, the total change in the species metric is:

$$B = \sum_{k}\sum_{i} B_{ik} = \sum_{k}\sum_{i}(\mu_{iT}^{k} - \mu_{i0}^{k})\tilde{S}_{i}^{k} \tag{5}$$

The determination of μ_{iT}^{k}, the expected condition metric over planning horizon T, as a function of fencing and conservation effort is described in the next section. Equation (5) can be interpreted as a weighted sum of condition changes where fragments with a higher potential species are given a higher weight. The advantage of a marginal species weighting scheme is that it is increasing in the fragment area and the slope parameter, and therefore, it is consistent with the SAR; and when the total area of fragments considered under a conservation scheme is fixed, it is additive and thus can be used to form the objective function for a mathematical programming problem. From Figure 2, taken from (2) and the case study, the marginal species is a convex function of the area of the fragment. The weight $\tilde{S}_{i}^{k}$ increases most rapidly for

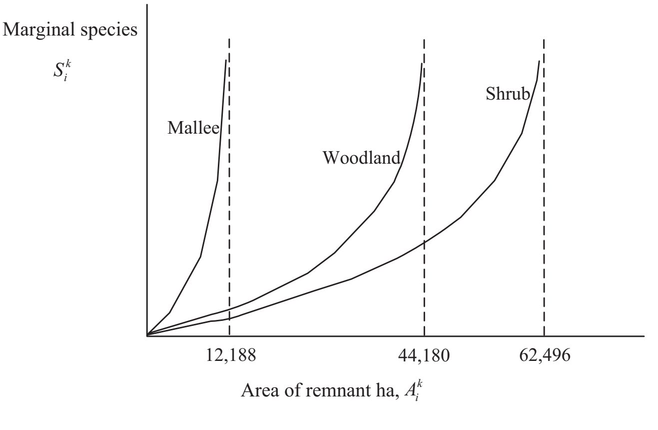

Figure 2 Marginal species of a fragment as a function of fragment area. The numbers given on the horizontal axis at the vertical asymptotes give the total area of fragments for the three vegetation types.

the vegetation type Mallee that has the smallest remaining area in the region and least rapidly for Shrub that has the largest.

3.2. Economic model

The motivation for considering an outcome-based component to a conservation scheme can be illustrated with the following example. An entirely input-based contract is akin to a contract that pays farmers a fee for actions linked to wheat production (ploughing, seeding and spraying), but no payment for the tonnes of wheat harvested. It is apparent that a farmer paid on this basis would pay little heed to their crops and the yields would be suboptimal. A similar argument applies to conservation schemes and possibly explains the low level of effectiveness of many conservation schemes.

The expected vegetation condition metric term can be generalized as:

$$\mu_{iT}^{k} = f_{i}^{k}(T, \mu_{i0}^{k}, x_{i}^{k}, e_{i}^{k}, a_{i}) \tag{6}$$

where the condition metric is a function of contract duration T, the initial condition μ_{i0}^{k}, annual conservation effort e_{i}^{k}, a binary variable $a_i \in 0, 1$ indicating fencing $a_i = 1$ or unfenced $a_i = 0$ and a vector of environmental covariates x_{i}^{k}.[1] These covariates include fragment shape complexity, connectivity to other fragments, management history and degradation processes such as salinization and fire. In what follows, the environmental covariates x_{i}^{k} are dropped to simplify the notation, and an empirical analysis showed that none of these were significant in determining the vegetation condition metric after including the initial condition.

Define the expected change in condition metric as:

$$\eta_{i}^{k}(T, \mu_{i0}^{k}, e_{i}^{k}, a_{i}^{k}) = f_{i}^{k}(T, \mu_{i0}^{k}, e_{i}^{k}, a_{i}) - \mu_{i0}^{k} \tag{7}$$

It should be noted that non-participation is possible and would be indicated by setting inputs equal to zero, $f_{i}^{k}(T, \mu_{i0}^{k}, 0, 0)$.

The expected biodiversity change from conservation is related to actions, initial conditions and contract duration by rewriting (5) as:

$$B = \sum_{k} \sum_{i} (\mu_{iT}^{k} - \mu_{i0}^{k}) \tilde{S}_{i}^{k} = \sum_{k} \sum_{i} \left(\eta_{i}^{k}(T, \mu_{i0}^{k}, e_{i}^{k}, a_{i}) \tilde{S}_{i}^{k} \right) \tag{8}$$

where B is the total expected change in species metric summed across i fragments and k vegetation types. Note that the term $\tilde{S}_{i}^{k}$ is independent of conservation effort and is a constant weight for each fragment given by (2).

[1] A bush fragement may include more than one vegetation type, but perimeter fencing encloses the whole fragment.

3.3. Farmer incentives

In this subsection, we introduce farmer incentives for participation and conservation effort on the basis of perfect information relating to the opportunity cost of effort and the cost of fencing. The implications of assuming perfect information are explored in the empirical section.

To simplify the notation, effort e_i is assumed constant in each year and the present-value of effort cost is given by:

$$\sum_{t=1}^{t=T} \delta^t w_i e_i = \tau(T) w_i e_i \tag{9}$$

where δ^t is the discount factor and $\tau\ (T) = (\delta^{T+1} - \delta)/(\delta - 1)$ the sum of a geometric progression. The landholder maximizes expected income from participating in a conservation scheme over a single contract period:[2]

$$m_i(\pi_i^0, \pi_i^T, e_i, a_i) = a_i(\pi_i^0 - w_i^0) + \left[\delta^T \pi_i^T \eta_i(T, \mu_{i0}, e_i, a_i)\tilde{S}_i - \tau(T) w_i e_i\right] \tag{10}$$

where $m_i(.)$ is profit from the conservation scheme, π_i^0 is an initial payment, and π_i^T a payment or charge based on the change in the species metric over the contract period. The ability of the regulator to levy a charge for a decline in the condition of the bush fragment depends upon the allocation of property rights and liability for failure to achieve conservation targets. In the empirical model, we assume that penalties are possible. The term w_i^0 is the cost for fencing, and w_i is the shadow wage rate. The regulator sets compensation rates so that the firm has an incentive to apply an optimal effort and fence off an optimal area. The farmer's effort is consistent with an incentive constraint (Laffont and Tirole 1993, 195):

$$\tilde{e}_i(\pi_i^T) = \arg\max E[m_i(\pi_i^0, \pi_i^T, e_i, a_i)] \tag{11}$$

This constraint ensures that the incentives for effort and fencing are consistent with the transfer payments. Assuming differentiability of (11), an internal solution is found where:

$$\delta^T \pi_i^T \tilde{S}_i \frac{\partial \eta_i}{\partial e_i} = \tau(T) w_i \tag{12}$$

That is the marginal expected payment equals the marginal cost. Even if the opportunity cost of effort is equal for all fragments, the transfer payment may vary between fragments owing to the variations in the marginal productivity of effort in conservation that is the term $\tilde{S}_i(\partial\eta_i/\partial e_i)$ in (12). These arise owing to different vegetation types and different areas. Rearranging (12) as:

[2] The vegetation type subscript k is omitted for clarity in this section.

$$\pi_i^T = \frac{\tau(T)w_i}{\delta^T \tilde{S}_i(\partial\eta_i/\partial e_i)} \tag{13}$$

indicates that the minimum transfer payment is positively related to the opportunity cost of effort, but inversely related to the marginal productivity of effort in conservation.

If the regulator uses a fixed transfer payment, $\bar{\pi}^T$, for effort, then the equality in (14) would be replaced by the weak inequality:

$$\bar{\pi}^T \leq \frac{\tau(T)w_i}{\delta^T \tilde{S}_i(\partial\eta_i/\partial e_i)} \tag{14}$$

This allows for non-participation where the cost per unit of effort exceeds the expected transfer payment.

3.4. The regulator's problem

The regulator aims to maximize the sum of species gains, measured by the species metric, by recruiting bush fragments to a conservation scheme given a fixed budget. This is a cost-effectiveness analysis in that costs are given, so a cost-effective conservation plan is one that gives the largest increase in the species metric for a given budget cost. The fate of all bush fragments is accounted for both those participating and those not. The regulator determines the initial transfer payment and outcome payment. Two forms of 'take-it-or-leave-it' conservation contracts are analysed: (i) a first-best contract, based on observable effort, where the payment on outcome is perfectly discriminated between fragments and (ii) a fixed-payment contract where effort is unobservable and improvements in outcomes are paid at a fixed rate for all participating fragments. Other assumptions include: for both contracts, fencing is always observable; the opportunity cost of effort per day is known with certainty by the farmer and in the first-best solution, by the regulator. In a fixed-payment scheme, the regulator pays out on improvement in the species metric regardless of effort: in other words, a landholder may be paid for random positive fluctuations in bush condition, whilst not engaging in any effort. Initially, it is assumed that the cost of fencing per km and the cost of effort per day are the same across all remnants, and the total cost of both per fragment will vary as a result of variations in the perimeter for fencing and the area for effort.

The regulator maximizes (8) subject to the incentive constraint (11), an individual rationality constraint from (10):

$$m_i(b_i, \pi_i, e_i, a_i) \geq 0 \quad \forall i \tag{15}$$

and an expected budget constraint

$$\sum_i \left(a_i\pi_i^0 + \delta^T\pi_i^T\eta_i(T, \mu_{i0}, e_i, a_i)\tilde{S}_i\right) \le g \tag{16}$$

where g is the budget. The variables $\{\pi_i^0, \pi_i^T\}$ define the policy applied to each fragment. In the case of non-recruitment, these would be zero.

Loch and Kavadias (2002) propose marginal conditions that provide a rule-of-thumb for optimal effort for a bush fragment. Assuming an internal solution with $a_i = 1$ and $e_i > 0$, the condition for spatial efficiency is:

$$\frac{\tilde{S}_j D_e(\eta_j)\vec{e}'_j(\pi_j^T)}{\tilde{S}_h D_e(\eta)\vec{e}'_h(\pi_h^T)} = \frac{\tilde{S}_j\eta_j + \pi_j^T\tilde{S}_j D_e(\eta_j)\vec{e}'_j(\pi_j^T)}{\tilde{S}_h\eta_h + \pi_h^T\tilde{S}_h D_e(\eta_h)\vec{e}'_h(\pi_h^T)} \quad \forall j, h \in N \tag{17}$$

where $D_e(\eta_i^k) = \partial\eta_i/\partial e_i$. This states that the marginal gain in biodiversity for each dollar spent on effort should be equal in each fragment included in the conservation scheme.

Conservation planning is based on the expected change in condition. The *actual* change in condition and the payment made to the producer depends upon short-term climatic fluctuations, although, in practice, a regulator may decide to delay assessments in years of extreme weather conditions such as drought.

3.5. Markovian representation of condition change

Over large landscapes, it is not possible to assess biodiversity condition in the field on a continuous scale. The vegetation condition values are therefore divided into L discrete states. Define a Markov transition probability matrix $P_i(e_{ij}, a_{ij})$ as a function of conservation effort and fencing. The $1 \times \mathbf{N}$ vector p_i^0 of absolute probabilities indicates the initial condition; for instance, $p_i^0 = (1, 0, 0, 0, 0)$ indicates that the initial condition for fragment i is in state one with certainty.[3] After T periods, the absolute probability is:

$$p_i(e_i, a_i, T) = p_i^0 P_i(e_i, a_i)^T \tag{18}$$

Note that $P_i(e_i, a_i)^T$ indicates that the matrix is raised to the power of T and the expected change in the species metric from (7) becomes:

$$\eta_i(T, \mu_{i0}, e_i, a_i)\tilde{S}_i = \mu' p_i(e_i, a_i, T) - \mu_{i0} \tag{19}$$

where μ is the $1 \times \mathbf{N}$ vector of fixed discrete biodiversity conditions across all N fragments, for instance in the case study $\mu = (0.1, 0.3, 0.5, 0.7, 0.9)$.

[3] If there is uncertainty about the exact condition of a fragment, it is possible to give the initial condition as an expectation, thus $p_i^0 = (0.5, 0.5, 0, 0, 0)$ indicates that there is a 50 per cent chance that the fragment is in state 1 and a 50 per cent chance that it is in state 2 (White 2005).

A further summary measure of the ecological change is the absolute probabilities after a long (∞) time period p_i (e_i, a_i, ∞). This summarizes the expected long-term equilibrium. Finally, Markov chains for intermediate levels of conservation effort $0 < e_i < e_i^{\max}$ are given as weighted averages of the 'fencing only' and 'conservation' Markov chains.

4. NEWROC case study

Applying the regulator's model requires a link between actions and vegetation condition change, an estimate of the species metric and an estimate of the costs of fencing and effort. These steps are applied to the NEWROC study area, a region within the South-west Australia Biodiversity Hotspot.

4.1. Study area

The study area is the North East Wheatbelt Regional Organisation of Councils (NEWROC) in Western Australia. The area is more than 250 km north-east of Perth, comprising approximately 17,000 km^2 of crop land within the intensive land use zone and approximately 7000 km^2 in the extensive land use zone. Average rainfall decreases north-eastward from 340 to 290 mm, with approximately 65 per cent of rain falling within the 5-month growing season (May–September). The NEWROC region is part of the Avon River Basin, in which over 50 per cent of the 4000 vascular plants indigenous to the Basin are known to be endemic (Keighery and Lyons 2001; Gibson *et al.* 2004). This diversity is linked to the age of the landscape (some landforms are estimated to be 2–3 billion years old) and long periods of geographical isolation supported by climatic and tectonic stability (Chen *et al.* 2003). A survey of the wheatbelt region as a whole estimated 2609 plant species (taxa) (Gibson *et al.* 2004), while (Beard *et al.* 2000) estimate 5710 for the entire Southwest Botanical Province.

Biodiversity within the NEWROC region faces significant regional threats from landscape fragmentation and increasing salinization (e.g. Myers *et al.* 2000; Sattler and Creighton 2002; Prober and Smith 2009). Land clearing has been extensive, with only 12 per cent of indigenous vegetation remaining within the intensive land use zone (ILZ) of the NEWROC region. The remaining vegetation occurs as isolated fragments, inhibiting the ability of species to recruit from neighbouring areas after disturbance events such as altered fire regimes or disease. Currently, 6 per cent of the ILZ is classified as salinity affected, and 28 per cent is considered at risk of salinization by 2050 (NLWRA 2001). Combined with other threats such as grazing and biotic invasions, the high endemism (where many species are unique to a local region only) means that there is a high expected rate of species loss. The degree of degradation of vegetation fragments across this region has not been systematically surveyed; however, the degradation is known to be extensive (Avon Catchment Council 2005).

4.2. Spatial data

The historical archive of calibrated Landsat TM imagery (Wallace *et al.* 2004; Furby 2007) provides a panel data measure of μ_{it}^{k}. The Band 5 of each Landsat TM image (25 m resolution) was clipped to the polygon defining each bush fragment, for each of 12 image capture dates between 1988 and 2007. A set of textural metrics were then applied to the images clipped to the bushland boundary (Baker and Cai 1992) to predict biodiversity condition as elicited from the ecological experts. Imagery relating to outliers were viewed manually and assessed to determine the source of any anomaly, for instance cloud cover, fire or clearing. This measure of vegetation condition metric was applied to 465 viable vegetation fragments, whose 'core area' was greater than 40 ha after allowing for a 30 m buffer strip. Focal species research in Western Australia suggested 40 ha as a notional minimal size for effective habitat (Lambeck 1997; Brooker 2002), while the 30 m buffer zone is based on minimum corridor width recommendations by (Frost *et al.* 1999). Focal species are a species whose ecological needs are similar to a range of other species; therefore, it serves to indicate suitable habitat and management for a range of species The core area is then defined by removing a 30 m buffer zone from the fragment boundary (to compensate for geospatial error and edge effects).

The vegetation condition metric μ_{it}^{k} can also be assessed directly in the field through biodiversity inventories and compared with benchmark states or expected species richness (e.g. as given by a species-area curve). In this study, condition estimates were elicited from biodiversity experts using site photographs. Photographs for Auction for Landscape Recovery (Gole *et al.* 2005) were acquired for 37 vegetation fragments in 2004 and a further 40 in 2007.

The Landsat data give information on the bush condition of 465 bush fragments of 40 ha or more in twelve images from 1988 to 2007. Functional principal components analysis (FPCA) (Ramsay and Silverman 2006) was used to determine the similarity between biodiversity condition trends of individual fragments, represented as smoothed beta-spline functions over the 20-year time horizon. The fragments were assigned to different classes through hierarchical clustering by Ward's method (Ward 1963; Blashfield 1976). Six response classes were chosen as fewer classes omitted the range of possible response types, and more only duplicated the existing response types to reduce within class sample size.

Each panel in Figure 3 represents all fragments classed into each response class using the FPCA approach. Response Class 1 is predominantly nature reserves and other public lands (70 per cent) managed by the Department of Environment and Conservation, but includes some private land, and Class 2 is undegraded bush in gradual decline, similarly Class 3. Class 4 is bush improving in condition possibly because of fencing and the discontinued use of the bush for grazing. Class 5 is bush fragments in rapid decline. Class 6 is degraded bush. It is notable from Figure 3 that, for all the response classes,

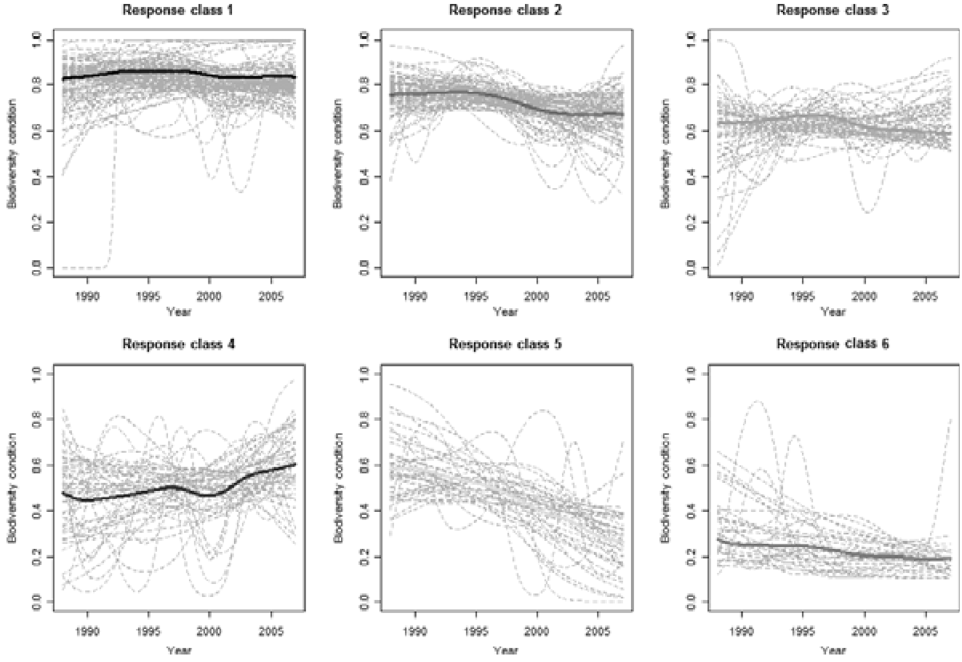

Figure 3 Vegetation condition metric response classes. The bold line in each case gives the expected response and dashed lines the actual condition change of individual bush fragments within each class.

there is a high degree of variability in condition both across fragments and through time within a response class.

4.3. Estimating Markov transition probabilities

State and transition models have been applied to a number of ecological systems (White 2005), including the wheatbelt ecosystems of Western Australia (Yates and Hobbs 1997). The Vegetation Assets, States and Transitions (VAST) is a standard framework for reporting the condition of native vegetation in Australia (Thackway and Lesslie 2006). The VAST framework 'classifies vegetation by degree of human modification as a series of states, from intact native vegetation through to total removal'. (op cit, p53). It is applicable across regions and across vegetation types. Here, we link the VAST framework to the Landsat data and using the FPCA response classes as a sampling frame to estimate Markov transition probabilities for each response class. In themselves, Markov transition matrices for vegetation condition states provide information about the underlying rate of vegetation change (White 2005).

Estimating annual Markov transition matrices requires an adjustment for periods between Landsat captures. Intervals between Landsat image capture dates were either 1 year (2004–2007) or 2 years (1988–2004). For each of the six biodiversity response classes, a transition matrix was estimated over each interval between image captures by counting transitions between each pair-wise combination of 'states' (1-5). Biodiversity condition scores at each time were assigned to one of the five states, based on 0.2 intervals (i.e. state 1:$\mu \in [0, 0.2)$; state 2: $\mu \in [0.2, 0.4)$; and so on). Of the eleven paired capture dates, transitions for eight were observed over a 2-year interval, and three over a 1-year interval, producing two separate matrices of transition counts.

We adopted the methods of Craig and Sendi (2002) and Takada *et al.* (2010) to estimate a single 1-year interval transition matrix over the 1988–2007 archive period, from data where intervals between capture periods were unequal. A bootstrap procedure was tested for the homogeneity of the transition matrix over the archive period (i.e. detecting where there are statistically significant changes in the dynamics of the system over time), for each of the biodiversity condition classes. The transition matrices for each response class were found to be homogeneous over time (significance level $\alpha = 0.05$).

A number of assumptions were made to link response classes to the survey data as a means of attributing costs to actions and thus response classes. Three response classes were selected where Class 1 represents intensive conservation that includes fencing plus a high level of effort in terms of weed elimination and revegetation, Class 3 represents fencing, but no conservation effort, and Class 6 represents no conservation. It would be possible to extend the analysis to all six response classes if more information became available on bush management.

The uncertainty concerning management relates primarily to the unobserved effort. We know approximately how much labour is used on nature reserves (B. Beecham, pers. comm., 2007). It is also possible to identify land where sheep have been excluded in most seasons through either on-ground photographs or aerial photographs (tracks and grazing pads). Given the uncertainty associated with the effort, the optimal conservation plan was subjected to a sensitivity analysis to assess the robustness of the fragment selection to under or over estimates of the labour input. The Markov transition probabilities are given in Table 1. The state transition literature in ecology (Yates and Hobbs 1997) discusses the importance of thresholds. The approach here is to estimate the Markov chains and identify where thresholds occur between biodiversity condition states. The degree of 'communication' between states is given by the probability of getting from each state to every other and indicates the strength of threshold effects if these 'escape' probabilities are low. This is the case with the Markov chains for response Classes 1 and 2. In the case of response Class 6, states 1–4 do not communicate with state 5. The Markov matrices account for both long-term ecological change and short-term fluctuations owing to weather and management.

Table 1 Markov transition matrices $P_i\,(e_{ij}, a_{ij})$

Condition μ	[0, 0.2)	[0.2, 0.4)	[0.4, 0.6)	[0.6, 0.8)	[0.8, 1.0]	$p_i(e_i, a_i, \infty)$
			Response class 1			
[0, 0.2)	0.6475	0.2861	0.0663	0.0001	0.0000	0.0023
[0.2, 0.4)	0.0120	0.6755	0.2333	0.0792	0.0000	0.0361
[0.4, 0.6)	0.0014	0.0389	0.8363	0.1234	0.0000	0.1836
[0.6, 0.8)	0.0006	0.0182	0.0997	0.7916	0.0899	0.2153
[0.8, 1.0]	0.0000	0.0000	0.0000	0.0344	0.9656	0.5627
			Response class 3			
[0, 0.2)	0.5253	0.4747	0.0000	0.0000	0.0000	0.0225
[0.2, 0.4)	0.0499	0.8448	0.1053	0.0000	0.0000	0.2138
[0.4, 0.6)	0.0000	0.0782	0.8328	0.0774	0.0116	0.2879
[0.6, 0.8)	0.0000	0.0000	0.0586	0.9208	0.0206	0.4338
[0.8, 1.0]	0.0000	0.0000	0.0048	0.2867	0.7085	0.0421
			Response class 6			
[0, 0.2)	0.8380	0.0525	0.0095	0.0000	0.0000	0.8622
[0.2, 0.4)	0.6303	0.3404	0.0293	0.0000	0.0000	0.0848
[0.4, 0.6)	0.0000	0.2477	0.6893	0.0630	0.0000	0.0431
[0.6, 0.8)	0.0000	0.0000	0.2737	0.7263	0.0000	0.0099
[0.8, 1.0]	0.0038	0.0002	0.0003	0.4211	0.5746	0.0000

4.4. Survey data to estimate conservation costs

The survey was conducted on 60 farms in the NEWROC. Face-to-face interviews took place in two periods November to December 2007 and February to March 2008. Landholders were drawn by a random stratified sample from the shires of Koorda, Mukinbudin, Mt Marshall, Nungarin, Trayning, Westonia and Wyalkatchem in proportion to the number of agricultural holdings in each shire.

The mean farm size was 6593 ha, with a minimum of 650 ha and a maximum of 30,000 ha. The farms were predominantly owned with 88 per cent of land area owned by the respondents, 11 per cent leased and 1 per cent share cropped. Cereal crops, lamb and wool production were the main farm enterprises. Around 87 per cent of respondents produced cereal crops and 66 per had a sheep enterprise. On average, 58 per cent of total farm area was cropping, 26 per cent grazing, 14 per cent native bushland and 2 per cent oil mallee production.

Of the 66 per cent of respondents engaged in sheep production, a mean number of 1471 ewes were recorded, with a minimum of 18 ewes and a maximum of 6000 ewes. The mean sheep stocking rate per forage hectare was 1.57 DSE (dry sheep equivalents) with a minimum of 0.2 DSE and a maximum of 4.5 DSE. A mean lambing rate of 84 per cent was recorded, with a minimum of 50 per cent and a maximum of 130 per cent.

Respondents on average applied 20.36 days work per year to nature conservation. Major limiting factors to involvement in nature conservation were

Table 2 Variables for the conservation planning model

	Mean	Standard deviation	Minimum	Maximum
Fencing cost $ per km	2094.84	329.27	1600.00	2600.00
Wage rate $ per hour conservation work	32.76	26.93	10.00	150.00
Person days of conservation work per annum per farm	20.36	28.08	0	180.00
Days labour per bush ha per annum	0.0708	0.0362	0	0.8571

Note: maximum labour days per bush ha per annum is included in the model as one standard deviation above the mean that is 0.107.

lack of time, funds and the availability of farm labour. An average of 38 per cent of bush fragments were fenced. Table 2 gives the parameters for the conservation planning model.

4.5. Species-area relationship

Estimates for the parameters of species-area curves in the WA Wheatbelt for vascular plants only are found in Keeley and Fotheringham (2003). Based on these values, the estimates for the parameter β^k for the archipelago of fragments in (1) for the NEWROC region are 0.227, 0.299 and 0.273, respectively, for woodland, mallee and shrub. A more accurate species-area curve for the NEWROC would require species lists for each bush fragment derived from a comprehensive field survey.

5. Results

These results are derived from the regulator's conservation planning model using the CONOPT (Drud 1992) non-linear programming algorithm in GAMS. The binary fencing variable $a_i \in 0, 1$ is approximated by making the variable non-negative and less than one. The fact a_i is only constrained by the budget means that this variable takes non-binary values $0 < a_i < 1$ in a small number of cases. This can be addressed by interpreting intermediate values as partial inclusion.

Two regional planning schemes are compared: the first-best scheme assumes perfect information on effort and fencing, the regulator only has to pay for *actual* effort not changes in vegetation condition owing to fencing (or chance). A 'second-best' fixed-payment scheme is based on observed fencing, but unobservable effort; therefore, the regulator pays out on condition change *even* when the effort is zero. The results are extended by a sensitivity analysis on the maximum conservation effort, a Monte Carlo assessment of the effects of variability in the opportunity cost of labour, and the likely implications of adverse selection and the budget effects of the contract duration increasing.

5.1. Baseline results

Table 3 and Figure 4 give the first-best and fixed-payment solutions. Results are compared on the basis of cost-effectiveness measured by the species change achieved for a given budget. With no conservation, the expected change over 10 years in the species metric falls by 76.69 per cent. With all fragments conserved at a cost of \$40 million, it increases by 20.75 per cent. Maintaining the current species metric costs around \$15 million. At lower budget levels, conservation gains are largely from fencing fragments and not from effort. As the budget rises effort increases and, at the maximum budget, outcome-based payments, π_i^T, account for 74 per cent of the total budget. Table A1 provides additional statistics.

The expected cost-effectiveness gains between the first-best and fixed payment are minimal at low budget levels as fencing is the most effective inter-

Table 3 Results for the first-best contract

Budget \$ (000's)	Per cent change in Species metric	Total effort input days	\$ Total initial payment (000's)	\$ Total final payment (000's)	No. of fragments conserved
0	−79.69	0	0	0	0
5000	−22.43	130	4707	293	160
10,000	−5.39	766	8284	1716	353
15,000	3.10	2215	10,040	4960	449
20,000	8.39	4403	10,135	9865	454
25,000	13.31	6600	10,216	14,784	458
30,000	16.80	8778	10,328	19,672	463
35,000	18.89	11,011	10,326	24,674	463
40,000	20.72	13,232	10,351	29,649	464
40,187	20.75	13,301	10,363	29,804	465

Note: Results are reported for every \$5 million increment in budget, the last budget amount indicates the cost of maximum effort and all fragments fenced.

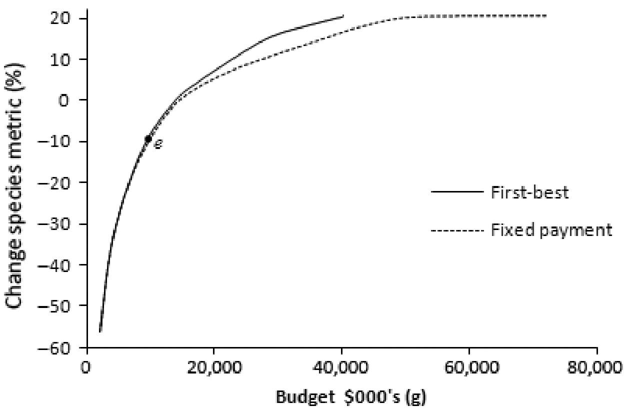

Figure 4 Total species changes for different schemes and budget levels over 10 years. Point e gives the species change when the whole area is fenced. At low budgets, the priority is only to fence rather than to fence and apply additional effort. Thus, the fence only line approximately coincides with the first-best line.

vention, and it is observable in both cases. At higher budget levels, the gain in cost-effectiveness increases (Figure 4).

5.2. Sensitivity analysis

A sensitivity analysis is applied to parameters that are difficult to observe directly. The first-best contract was rerun with the maximum labour requirement increased and decreased by 50 per cent. These changes have a significant effect at high budget levels, but a minimal effect at low budget levels when fencing alone is the preferred action, see Figure 5.

The actual opportunity cost of effort varies significantly from farm to farm. This potential cost of asymmetric information was analysed by a Monte Carlo simulation where the wage rate is drawn for each farm randomly from a normal distribution truncated at zero with a standard deviation of \$26.93. A thousand independent wage draws were made for each fragment, and the optimal conservation planning model runs with species change fixed at 0.03 and the budget minimized. Assuming perfect information about farmer's opportunity costs of effort, the first-best contract budget is \$17.3 million. Applying a fixed price contract scheme assuming average opportunity costs required a budget of \$20.3 million, an increase of 17 per cent over the first-best. This result approximately indicates the maximum cost saving that could be achieved by a price discriminating conservation auction.

Ideally conservation contracts should be for a relatively long period. Figure 6 shows the increase in budget for the first-best and the range of species metric change for three contract durations 10 (the benchmark), 20 and 50 years. After 50 years, the Markov transition probabilities are approaching a steady state. These results show that most of the species metric changes have occurred over 10 years and it is therefore possible to deliver conservation schemes with a sequence of short contracts. A 50-year contract

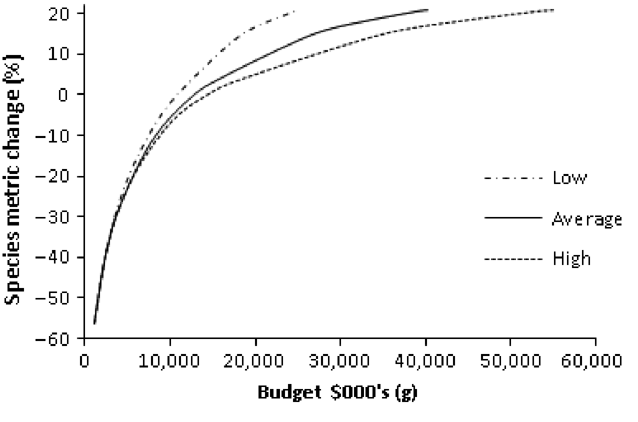

Figure 5 Sensitivity analysis of maximum effort required for conservation. Low represents 0.0535 person days per ha of bush per year; average is 0.107; and high is 0.1605.

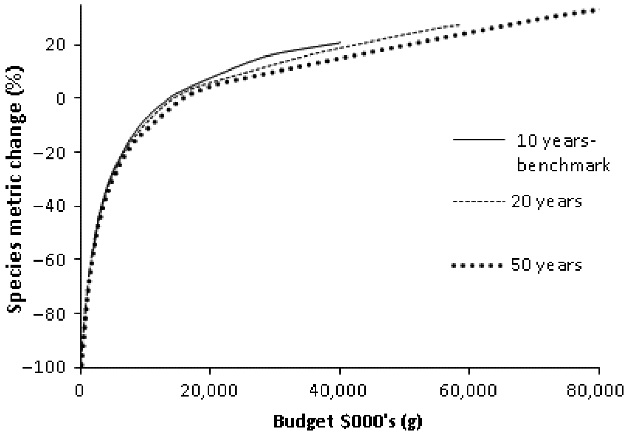

Figure 6 Sensitivity analysis for different contract durations.

may be difficult to implement as anything other than a series of shorter contracts, given typical business and policy planning horizons.

6. Conclusion

As biodiversity is increasingly scarce, prioritizing conservation has become a key economic problem for society. It is a problem that requires a combined economic and ecological approach that accounts for the spatial complexity and variability inherent in ecosystems and the need to provide incentives to private landholders in the presence of asymmetric information.

This integrated economic and ecological analysis of conservation planning has brought conservation into the fold of conventional production economics where inputs determine the condition of vegetation and the appreciation or depreciation of biodiversity assets through time. It has defined the regulator's objective function and the landholder's incentive constraints so that feasible optimal policies can be determined.

There are four key results from the empirical conservation planning model developed here: first, without a conservation scheme, significant biodiversity losses will occur in the NEWROC; second, the first priority is to fence bush fragments; third, as the budget increases, then a combined fencing and outcome-based contract increase the cost-effectiveness of a conservation scheme; and fourth, the additional costs of a fixed transfer payment scheme over the first-best indicate a potential cost saving from a price discriminating conservation auction of around 17 per cent.

The broader implications of this type of approach to conservation policy in agricultural landscapes are as follows. First, that conservation planning should be based on spatial data relating to links between actions and biodiversity outcomes. There is scope for further research on this relationship especially by relating farm records on bush management through time to observed changes in spatial data including aerial photographs and satellite data. Second, the duration of contracts should be commensurate with the rate of ecological change otherwise short-term noise, as shown in the NEWROC data, may mask long-term changes and contracts may be terminated prematurely. Third, outcome payments could be based on self-reporting to avoid administrative costs, with random spot checks.

With biodiversity loss accelerating (Millennium Ecosystem Assessment. 2005), society should consider innovative approaches to the problem of biodiversity conservation. An outcome-based (or mixed input- and output-based) contract would appear to have a number of advantages. In particular, they largely avoid the moral hazard problem inherent in input-based contracts but, more fundamentally, outcome-based contracts provide a strong incentive for landholders to understand what determines bush condition and how to reduce the risks of failure to achieve conservation outcomes. In the same way that farmers are adept at managing crop and livestock production under challenging and changeable environmental conditions, they would also learn to manage bush remnants. This study is an initial step in addressing some of the issues related to outcome-based contracts, in particular, outcome measurement, contract design and the optimal selection of conservation contracts with an outcome-based component.

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Appendix

Table A1 Summary statistics.

Variable	Mean	Standard deviation	Min	Max	Total
Area of fragments hectares	267	785	40	10,072	124,386
Perimeter (km)	11.16	16.74	2.80	182.64	5191
Fencing cost \$000's[2]	22.33	33.49	5.60	365.28	10,383.38
Effort cost \$000's[3]	64.09	188.05	9.59	2413.19	29,803.52
Proportions					
Woodland	0.386	0.413	0.000	1.000	–
Shrub	0.433	0.432	0.000	1.000	–
Mallee	0.153	0.341	0.000	1.000	–
Other	0.028	0.117	0.000	0.981	–
Area					
Woodland	132.68	342.47	0.08	4282.74	44,180.80
Shrub	221.62	544.31	0.60	8402.65	62,496.06
Mallee	135.42	88.15	2.64	1032.89	12,187.65
Other	96.84	72.48	0.05	1029.95	5520.08
β_k (species-area curve)					
Woodland	0.227				
Mallee	0.299				
Shrub	0.277				
Initial condition (μ_i)	0.595	0.261	0.002	1.000	
Vegetation species metric[1]					
Conservation	0.164	0.825	−3.343	9.520	–
Fence	−0.066	0.804	−7.024	5.833	–
No conservation	−0.630	2.005	−25.423	1.442	–

Notes: All variables had 465 observations. Only fragments over 40 ha considered.

1. Change in the vegetation species metric is measured after 10 years given as an average for all vegetation types.
2. Total fencing costs are the cost to fence all fragments.
3. Effort costs are the total cost of maximum effort across all fragments.