Contributed Paper

Carbon Payments and Low-Cost Conservation

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Abstract: *A price on carbon is expected to generate demand for carbon offset schemes. This demand could drive investment in tree-based monocultures that provide higher carbon yields than diverse plantings of native tree and shrub species, which sequester less carbon but provide greater variation in vegetation structure and composition. Economic instruments such as species conservation banking, the creation and trading of credits that represent biological-diversity values on private land, could close the financial gap between monocultures and more diverse plantings by providing payments to individuals who plant diverse species in locations that contribute to conservation and restoration goals. We studied a highly modified agricultural system in southern Australia that is typical of many temperate agriculture zones globally (i.e., has a high proportion of endangered species, high levels of habitat fragmentation, and presence of non-native species). We quantified the economic returns from agriculture and from carbon plantings (monoculture and mixed tree and shrubs) under six carbon-price scenarios. We also identified high-priority locations for restoration of cleared landscapes with mixed tree and shrub carbon plantings. Depending on the price of carbon, direct annual payments to landowners of AU\$7/ha/year to \$125/ha/year (US\$6–120/ha/year) may be sufficient to augment economic returns from a carbon market and encourage tree plantings that contribute more to the restoration of natural systems and endangered species habitats than monocultures. Thus, areas of high priority for conservation and restoration may be restored relatively cheaply in the presence of a carbon market. Overall, however, less carbon is sequestered by mixed native tree and shrub plantings.*

Keywords: agroecosystems, biodiversity and conservation banking, economic analysis, ecological restoration, emissions trading policy, incentives, reforestation

Pagos de Carbono y Conservación de Bajo Costo

Resumen: *Se espera que un precio por el carbono genere demanda por programas de compensación de carbono. Esta demanda podría dirigir la inversión en monocultivos basados en árboles que aportarían mayor producción de carbono que plantaciones diversificadas de especies nativas de árboles y arbustos, que secuestran menos carbono pero proporcionan mayor variación a la estructura y composición de la vegetación. Instrumentos económicos como la banca de conservación de especies, la creación y comercialización de créditos que representan valores de la diversidad biológica en terrenos privados, podrían cerrar la brecha financiera entre monocultivos y plantaciones más diversas proporcionando pagos a individuos que siembren especies diversas en sitios que contribuyan a metas de conservación y restauración. Estudiamos un sistema agrícola sumamente modificado en el sur de Australia que es típico de muchas zonas agrícolas templadas en el mundo (i.e., tiene una alta proporción de especies en peligro, altos niveles de fragmentación del hábitat y presencia de especies exóticas). Cuantificamos los beneficios económicos de la agricultura y de plantaciones de carbono (monocultivos y árboles y arbustos) bajo 6 escenarios de precio de carbono. También identificamos sitios de alta prioridad para la restauración de paisajes deforestados mediante plantaciones de carbono mixtas de árboles y arbustos. Dependiendo del precio del carbono, los pagos anuales directos a propietarios de AU\$7/ha/año a \$125/ha/año (US\$6–120/ha/año) pueden ser suficientes para aumentar los beneficios económicos de un mercado de carbono y promover plantaciones de árboles que contribuyen más que los monocultivos a la restauración de sistemas naturales y hábitats de especies en peligro. Por lo tanto, la*

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restauración de áreas de alta prioridad para la conservación y la restauración puede ser relativamente barata en la presencia de un mercado de carbono. Sin embargo, en general, las plantaciones mixtas de árboles y arbustos nativos secuestran menos carbono.

Palabras Clave: agroecosistemas, análisis económico, banca de biodiversidad y conservación, incentivos, políticas del comercio de emisiones, reforestación, restauración ecológica

Introduction

Carbon trading and its resultant market for carbon offsets are expected to drive investment in the sequestration of carbon through tree plantings (i.e., carbon plantings). Most carbon-planting investment has been in monocultures of trees that offer a rapid return on investment but have relatively little compositional and structural diversity (Bekessy & Wintle 2008; Munro et al. 2009). There are additional benefits available should carbon plantings comprise native species of diverse composition and age that are planted strategically to meet conservation and restoration objectives (hereafter ecological carbon plantings) (Bekessy & Wintle 2008; Dwyer et al. 2009; Bekessy et al. 2010). Ecological carbon plantings may increase availability of resources and refugia for native animals, but they often yield less carbon and are more expensive to establish than monocultures and therefore are less profitable. Economic incentives are required to motivate landowners to invest in ecological carbon plantings. For example, Bekessy and Wintle (2008) propose financial incentives for tree plantings that restore natural systems and endangered species' habitats in addition to sequestering carbon. Financial incentives would need to be large enough to motivate investment in locations where ecological restoration is a high priority. We aimed to quantify the trade-off between carbon sequestration and the provision of structural and compositional diversity and to estimate the payments needed to achieve restoration and conservation goals.

The volume of carbon sequestered in new carbon plantings increases rapidly until the vegetation reaches maturity (Silver et al. 2000) and is a function of the identity of the tree species planted, management actions, and the lithology, climate, and topography of the site (Landsberg & Waring 1997; Roxburgh et al. 2006). The carbon sequestered at the site above an estimated baseline is the commodity that can be traded in a market. As a general rule, faster-growing species in sites with warmer and wetter climates and more fertile soils sequester carbon faster than species in other types of sites (Silver et al. 2000).

The economic returns of carbon plantings are highly variable and depend primarily on carbon yield and price and opportunity costs (Newell & Stavins 2000; Richards & Stokes 2004; Torres et al. 2010). In this context, opportunity cost is usually expressed as the profit from agricultural production. The spatial variation in carbon yield and costs, including establishment, maintenance, transaction, and opportunity costs, means that the net economic returns of carbon plantings are also likely to vary spatially.

Not all ecological carbon plantings will provide the same conservation and restoration benefits (Polasky et al. 2008; Thomson et al. 2009; Lindenmayer et al. 2010). Ecological carbon plantings located so as to reduce vegetation fragmentation, expand and link remnant vegetation patches, and restore ecosystem function will be of greatest benefit to conservation and restoration planning goals (Crossman & Bryan 2006; Benayas et al. 2009; Munro et al. 2009). Complex vegetation structure often provides resources for bird nesting, perching, and shelter (McElhinny et al. 2006) and sites for seedling establishment (White et al. 2004).

We modeled the spatial distribution of traditional agricultural land uses, monoculture carbon plantings, and ecological carbon plantings in the agricultural areas of South Australia. We quantified the economic returns from agriculture and from carbon plantings (monoculture and ecological) under six carbon-price scenarios. We also identified high-priority locations for restoration of cleared landscapes with ecological carbon plantings, assuming the plantings are designed in such a way to comply with the definition of ecological restoration (Society for Ecological Restoration International Science and Policy Working Group 2004). We measured the benefit of ecological carbon planting as the contribution plantings made to targets for expansion and connection of patches of remnant vegetation ensuring 30% of all biophysical surrogates for species richness and diversity (environmental variables, sensu Margules & Pressey 2000) are contained in remnant and replanted vegetation. Finally, we quantified the cost of payments required to locate ecological carbon plantings in areas of high conservation and restoration priority.

Methods

Study Area

Our 15.8 million-ha study area (Fig. 1a) was in the agricultural districts of the state of South Australia. The area has 10.6 million ha of land under agricultural production, much of which was cleared of native vegetation over 50 years ago. The remaining 5.2 million ha is primarily remnant native vegetation (Fig. 1a). Most of the

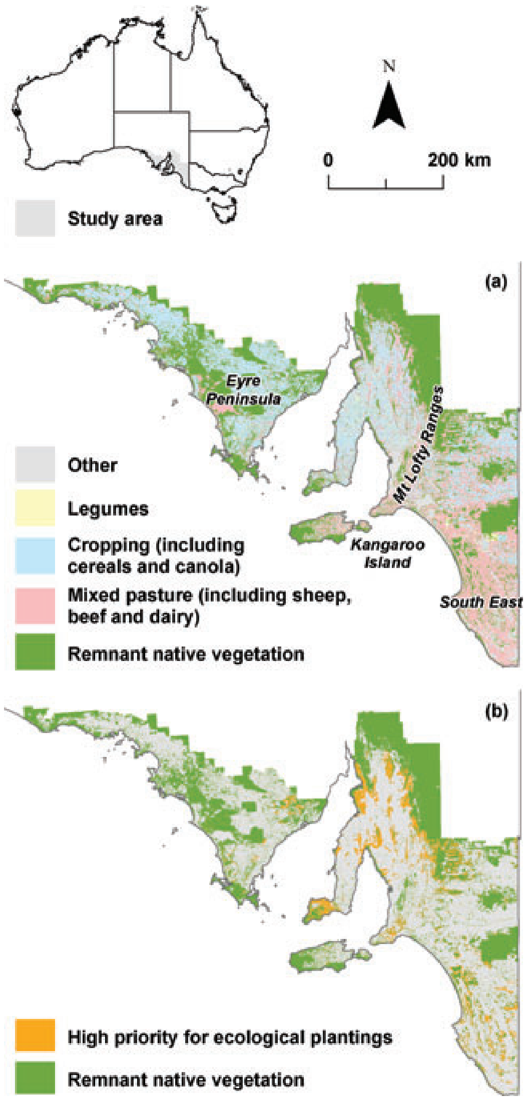

Figure 1. (a) Location of the study area and the dominant dryland agricultural land uses modeled in this study and (b) locations in the study area identified as priorities for ecological restoration.

remnant vegetation is in 20 large (>10,000 ha) contiguous fragments in the northern and western parts of the study area, where rainfall is low. Kangaroo Island, Mt. Lofty Ranges, southern Eyre Peninsula, and the South East are characterized by higher rainfall and economically profitable agricultural lands within which are about 50,000 small fragments (mean = 24 ha) of remnant vegetation. The Mt. Lofty Ranges and Kangaroo Island are rich in endemic and native species (Australian Government 2010*a*). *Acacia* spp. and *Eucalyptus* spp. are the dominant tree species in the remnant vegetation, and major native vegetation types are open forests, woodlands, and open woodlands.

The dominant agricultural uses were annual crops (e.g., wheat, barley, and canola) (5 million ha; 47.2% of cleared area), annual legumes (e.g., field peas, Faber beans, lupins, and lentils) (0.3 million ha; 2.8%), and grazing of sheep and cows (4 million ha; 37.7%) (Department of Water Land and Biodiversity Conservation 2008). The climate is largely Mediterranean with average annual rainfall ranging from 250 mm in the northern parts of the study area to over 1000 mm in the southern Mt. Lofty Ranges and lower South East.

Agricultural Production and Economics

Estimating Yields

We used the Agricultural Production systems SIMulator (APSIM) (Keating et al. 2003) to model annual production of wheat and legumes (field peas). The APSIM model simulates crop production at point locations as a function of soil, climate, inputs, and management regimes, such as fertilizer and pesticide application rates and timing.

We stratified the study area into 80 zones with similar soils and climates on the basis of soil data (Department of Water Land and Biodiversity Conservation 2007) and BIOCLIM (Nix 1986) climate surfaces. We matched soil and climate characteristics of each zone to soil and climate characterizations for South Australia that are archived in the APSIM database. We calculated APSIM yield estimates for wheat and field peas for each of the 80 soil-climate zones to produce continuous surfaces of predicted yields of wheat and field peas as a function of long-term average climate (1889–2008). We performed all spatial analyses on 1 ha grids in ArcGIS (version 9.3) (ESRI 2009).

We extracted estimates of stocking rates for beef cattle and sheep and total area grazed by livestock from the 2006 Agricultural Census data (Australian Bureau of Statistics 2006). The agricultural census collects farm-resolution agricultural production and income statistics for major commodities and aggregates these at the coarser resolution of local government administrative districts.

Quantifying Profit

We based our calculations of agricultural profit on Bryan et al. (2009), who calculated profit at full equity (i.e., economic return to land, capital, and management, exclusive of financial debt). We calculated an annual profit at full equity (PFE_c) layer for each commodity (c) in the set of agricultural commodities (C), where C is wheat, field peas, beef cattle, or sheep.

$$\mathrm{Revenue}_c = (P1_c * Q1_c * \mathrm{TRN}_c) + (P2_c * Q2_c * Q1_c), \tag{1}$$

$$\text{variable costs}_c = (\mathrm{QC}_c * Q1_c) + \mathrm{AC}_c\,, \tag{2}$$

$$\text{fixed costs}_c = (\mathrm{FOC}_c + \mathrm{FDC}_c + \mathrm{FLC}_c), \quad \text{and} \tag{3}$$

$$\text{PFE}_c = \text{revenue}_c - (\text{variable costs}_c + \text{fixed costs}_c), \quad (4)$$

where P is price, Q is yield, TRN is proportion of the herd sold, QC is quantity-dependent variable costs, AC is area-dependent variable costs, FOC is fixed operating costs, FDC is fixed depreciation costs, and FLC is fixed labor costs (Table 1). We sourced the variable and fixed costs (Table 1) from a gross margin handbook (Rural Solutions 2008) and average commodity prices for the period 2002–2007 (Table 1), before the commodity boom of mid-2008, from a commodity market publication (Australia Bureau of Agricultural and Resource Economics 2010). We used the APSIM outputs for crop yields (Table 1). All currencies here and in following analyses are in Australian dollars (AU\$1 = US\$0.95, AU\$1 = € 0.70 in October 2010).

SPATIALLY ALLOCATING PROFIT

Annual rotations of crops and livestock grazing are not captured by one-time classifications of land use and vary as a function of soils and climate. We accounted for agricultural rotations in our modeled long-term economic profit of agriculture by calculating the areal proportion of each crop and type of grazing livestock within a 10 km (100 cell) radius of each 1 ha grid cell. We assumed the areal proportion reflects the frequency of each land use (Bryan et al. 2011). Outputs were 4 layers of frequency, 1 for each land use (w_c): $\sum_{c \in C} w_c = 1$. We calculated a single layer of economic returns from agriculture as the long-term expected annual profit:

$$\text{PFE} = \sum_{c \in C} w_c \text{PFE}_c. \quad (5)$$

Carbon Sequestration and Economics

ESTIMATING SEQUESTRATION RATES

We used the tree-stand growth model 3-PG (physiological principles predicting growth) (Landsberg & Waring 1997) to simulate annual carbon sequestration under permanent carbon plantings in the part of the study area cleared for agriculture. The 3-PG model calculates total above- and below-ground biomass of a stand after accounting for soil water deficit, atmospheric vapor pressure deficits, and stand age.

The 3-PG model requires data on average monthly climate (temperature, solar radiation, rainfall, number of frost days), site factors (latitude, soil texture, maximum available water in the soil, and soil fertility), initial densities of trees, and management practices such as thinning and fertilizing. Empirical data on tree species' growth parameters, including canopy structure; partitioning of biomass between leaf, wood, and roots; wood density; litter fall; and root turnover rates, are included in 3-PG as species parameter files. The model runs on a monthly time step and outputs include diameter at breast height, stem volume, leaf area index, and biomass of leaf, wood, and roots. The 3-PG model was originally parameterized for a generic species, but species-specific parameters have since been calibrated for many commercially valuable trees (Paul et al. 2007) and most recently for mixed species used in permanent ecological restoration plantings (Polglase et al. 2008).

We simulated four carbon-planting systems described in Polglase et al. (2008) for which the plants in the systems would grow in our study area. All species were native to areas of Australia with climate similar to that in the study area. We simulated the annual growth of three trees typically grown in monoculture (*Eucalyptus globulus*, native to Tasmania, constrained to precipitation $\geq$ 550 mm/year; *Eucalyptus camaldulensis*, native to the study area, constrained to 350–549 mm/year; *Eucalyptus kochii*, native to Western Australia, constrained to <350 mm/year). For the simulations of ecological carbon plantings we used a set of trees and shrubs representative of those planted for ecological restoration in temperate southern Australia (species list in England et al. 2006). We assumed the ecological carbon plantings were planted and managed in such a way as to comply with the definition of ecological restoration (Society for Ecological Restoration International Science and Policy Working Group 2004).

England et al. (2006) calibrated the ecological carbon-planting parameters with field measurements and samples from 53 sites in southern Australia formerly used for agriculture that have been planted with seedlings of native species, directly seeded with native species, or have regenerated naturally. Species were primarily a mixture of native eucalypt trees and acacia shrubs that were present before agricultural clearance of the land. England et al. (2006) sampled an additional 16 sites to independently validate 3-PG predictions of biomass accumulation in the ecological carbon plantings. The age of revegetation and regeneration sites was 5–25 years (England et al. 2006). England et al. (2006) and Polglase et al. (2008) provide further details on calibration methods and the age and species composition of regeneration and revegetation sites.

We used an initial planting density of 1000 stems/ha, a density typical of plantation forestry (Polglase et al. 2008) and ecological restoration (Department of Sustainability and Environment 2006) in temperate parts of southern Australia. We modeled a 41-year time horizon (2009–2050). We assumed each tree system modeled reached maturity by 2050. Outputs for each system were annual dry biomass flux for each year from 2010 to 2050. We calculated carbon sequestration per year (Q_f), expressed as carbon dioxide equivalent terms ($CO_2{}^{-e}$), by using the equation $\frac{44}{12}(0.5 * B_f)$ to convert

Table 1. Profit function parameters and associated values used to determine economic returns of the dominant agricultural commodities produced in South Australia.

Parameter	Description[a]	Value of wheat	Value of field peas	Value of beef cattle	Value of sheep
P1	price of primary product (\$/t or \$/DSE)	216	274	97	25–35
P2[b]	price of secondary product (\$/kg)	na	na	na	3.19–4.27
Q1	yield of primary product (t/ha or DSE/ha)	0–3.62	0–4.84	0–9.70	0–9.70
Q2	yield of secondary product (kg/DSE)	na	na	na	5
TRN	proportion of herd sold ($0 \leq$ TRN ≤ 1 for livestock)	na	na	0–0.46	0–0.96
QC	quantity-dependent variable costs (\$/t or \$/DSE)	16–32	16–32	2–3	3–7
AC	area-dependent variable costs (\$/ha)	122–311	153–197	6–13	1–13
FOC	fixed operating costs (\$/ha)	22–32	22–32	5–21	1–46
FDC	fixed depreciation costs (\$/ha)	10–29	10–29	1–7	1–15
FLC	fixed labor costs (\$/ha)	22–36	22–36	1–20	1–25

[a] *Monetary units are Australian dollar. Abbreviations: DSE, dry sheep equivalents; na, not applicable.*
[b] *Wool production.*

dry biomass weights B_f for each system f in the set of tree systems F (where F is ecological carbon plantings, *Eucalyptus globulus*, *Eucalyptus camaldulensis*, *Eucalyptus kochii*). We used 0.5 as an approximation of the proportion of carbon in biomass, which ranges from 47–52% (West 2009). We calculated average annual rates of sequestration by dividing total $CO_2{}^{-e}$ sequestered at year 2050 by 41.

ECONOMIC RETURNS FROM CARBON SEQUESTRATION

We calculated the net present value (NPV_{fp}) of permanent monoculture and ecological carbon plantings to 2050 across the study area for each tree system (f) and each carbon price (p) in the set of carbon prices (P), where P is \$10/t $CO_2{}^{-e}$, \$15/t $CO_2{}^{-e}$, \$20/t $CO_2{}^{-e}$, \$25/t $CO_2{}^{-e}$, \$30/t $CO_2{}^{-e}$ or \$45/t $CO_2{}^{-e}$.

$$\mathrm{NPV}_{fp} = \mathrm{PVB}_{fp} - \mathrm{PVC}_f, \quad (6)$$

where PVB_{fp} is the present value of the benefits and PVC_f is the present value of the costs for each f. We selected carbon prices that reflect a range of prices realistically expected in a carbon market (Lawson et al. 2008). In October 2010, $CO_2{}^{-e}$ was trading at \$21 on the European Union carbon market, the largest carbon market by volume traded (European Energy Exchange 2010). We calculated PVB_{fp} as

$$\mathrm{PVB}_{fp} = \sum_{t=0}^{T} \frac{P_p Q_{ft}}{(1+r)^t}, \quad (7)$$

where r is the annual discount rate of 7% and T is the time horizon of 41 years. We calculated PVC_f as

$$\mathrm{PVC}_f = EC_f + \sum_{t=0}^{T} \frac{MC + \mathrm{PFE}}{(1+r)^t}, \quad (8)$$

where PFE is the opportunity cost of carbon plantings, EC_f is an initial establishment cost, and MC is the annual maintenance and transaction cost. Both EC_f and MC are uniform over the study area. We used \$25/ha for MC (Polglase et al. 2008). We used \$2000/ha for the ecological carbon plantings establishment cost and \$1250/ha for each of the monoculture carbon plantings establishment costs. The establishment cost we used for the monoculture plantings is comparable to that used in other studies of monoculture carbon plantings (Polglase et al. 2008), and the establishment cost of ecological carbon plantings is typical of a recent project conducted by the South Australian State Government. We converted NPV_{fp} to equal annual equivalent terms EAE_{fp}:

$$\mathrm{EAE}_{fp} = \mathrm{NPV}_{fp} \frac{r(1+r)^t}{(1+r)^t - 1}. \quad (9)$$

Carbon plantings are economically viable where they are more profitable than existing agriculture (i.e., $\mathrm{EAE}_{fp} > 0$). We assumed carbon plantings are not harvested.

Identifying Priority Locations for Ecological Restoration

We used a systematic landscape restoration model to identify an efficient set of sites for ecological restoration in locations cleared for agriculture (Crossman & Bryan 2006; Bryan & Crossman 2008). The linear programming model identifies the most efficient set of sites that achieve an a priori objective given a set of constraints.

We delineated the study area into 133,524 spatial units (mean area = 69 ha). The spatial units were spatially discrete combinations of pre-European vegetation communities (12), climate zones (18), soil types (18), and bioregions (9) (Table 2). We indicated whether the area was selected for restoration as a binary variable. The variable was a vector $\mathbf{X}$ with dimension N and elements x_i that can take the value of either 1 (unit i is selected for restoration) or 0 (unit i is not selected) for $i = 1, \ldots N$. The objective is to select units (i.e., sites) for restoration that are adjacent to remnant vegetation and thus reduce fragmentation, given the constraint that at least 30% of the area of each major pre-European vegetation community, climate zone, soil type, and bioregion is covered by

Table 2. Description of parameters used to identify priority locations for ecological restoration.

Parameter	*Description*
Fragmentation index	percent remnant vegetation cover in 5-km radius from every location in study area; linearly rescaled 1 (least fragmentation—high priority) to 5 (greatest fragmentation—low priority)
Dispersal index	Euclidean distance (*D*) from remnant vegetation rescaled with a negative exponential transformation; locations closest to remnant vegetation have exponentially greater importance on the basis of dispersal ecology theory (Willson 1993) and are scored 1 (closest to remnants) to 5 (furthest from remnants); calculated as $e^{-0.001D}$
Pre-European communities	estimated extent of major vegetation types prior to clearance for agricultural development; mapped with historical sources and interpolated remnant vegetation and landscape types (Bickford & Mackey 2004)
IBRA[a] subregions	subregions from the mapped Interim Biogeographic Regionalization of Australia (Australian Government 2010b)
Soil types	dominant soil types within the South Australian Soil Landscape spatial database (Department of Water Land and Biodiversity Conservation 2007)
Climate zones	areas of relatively homogenous climate (precipitation, temperature, and solar radiation); derived with methods of Crossman and Bryan (2006)

[a] *Interim Biogeographic Regionalization of Australia.*

at least 1000 ha of vegetation (restored and remnant). Mathematically we minimized

$$\sum_{i=1}^{N} \frac{x_i}{D_i F_i} \tag{10}$$

$$\text{subject to} \sum_{i=1}^{N} x_i \geq \max(0, 0.3A_i - V_i,\ \min(1000, A_i) - V_i) \quad \text{for } i = 1, \ldots N, \tag{11}$$

where D_i and F_i are the mean values of dispersal and fragmentation indices (Table 2) within each spatial unit i, V_i is the area of remnant vegetation, and A_i is the total area of each spatial unit i. For the dispersal index (Table 2), we applied dispersal theory (Willson 1993) to assess the likelihood that each cleared location will receive a propagule from neighboring remnant vegetation. We used the fragmentation index (Table 2) to assign a score to each cleared location for the proportion of remnant vegetation-type cover within a 5 km radius. We adopted a 30% threshold of vegetation cover because, although somewhat arbitrary, it is consistent with one recommended proportion for conservation of birds and mammals (Andren 1994). We used the general algebraic modeling system (GAMS) to build the linear programming model and solved it with CPLEX (GAMS Development Corporation 2009).

Cost of Conservation and Restoration

The aim of an economic instrument such as a conservation bank is to compensate land owners for providing a public service; in this case switching from the more profitable monoculture carbon plantings to the less profitable ecological carbon plantings. We estimated the cost of conservation and restoration payments (BC_p) as the difference in NPV between the monoculture and ecological carbon plantings in locations identified as a priority for ecological restoration that are profitable for monocultures at all carbon prices. We converted costs to equal annual equivalent terms with Eq. 9. We summed restoration payments in priority locations for all carbon prices to calculate the total amount necessary to compensate land owners for growing the potentially less profitable ecological carbon plantings.

Results

Much of the study area was profitable for agriculture under long-term average climate and recent commodity prices (Fig. 2a). Some locations returned up to $500/ha/year (Fig. 2a). A small number of areas were not profitable for agriculture given long-term average climate.

Monoculture carbon plantings (Fig. 2b) sequestered an average of 2.9 t $CO_2{}^{-e}$/ha/year (SD 2.5) more than the ecological carbon plantings, which sequestered an average of 13.4 t $CO_2{}^{-e}$/ha/year (SD 4.4) (Fig. 2c) across the entire study area. In locations with $\geq$ 550 mm rain/year monoculture carbon plantings sequestered an average of 21.8 t $CO_2{}^{-e}$/ha/year (SD 2.7). Up to 31 t $CO_2{}^{-e}$/ha/year could be sequestered in the wettest locations, which received up to 1000 mm rain /year. Monoculture carbon plantings in areas with rainfall $<$ 350 mm/year in the north and east of the study area sequestered an average of 13.8 t $CO_2{}^{-e}$/ha/year (SD 1.8). Ecological carbon plantings sequestered an average of 19.2 t $CO_2{}^{-e}$/ha/year (SD 2.5) and 8.5 t $CO_2{}^{-e}$/ha/year (SD 2.0) in the same high and low rainfall areas, respectively

Because monoculture plantings sequestered more carbon than ecological carbon plantings (Fig. 2d), they were also more profitable (Fig. 2e). For example, in the $20/t $CO_2{}^{-e}$ carbon-price scenario, 80% of the study area was

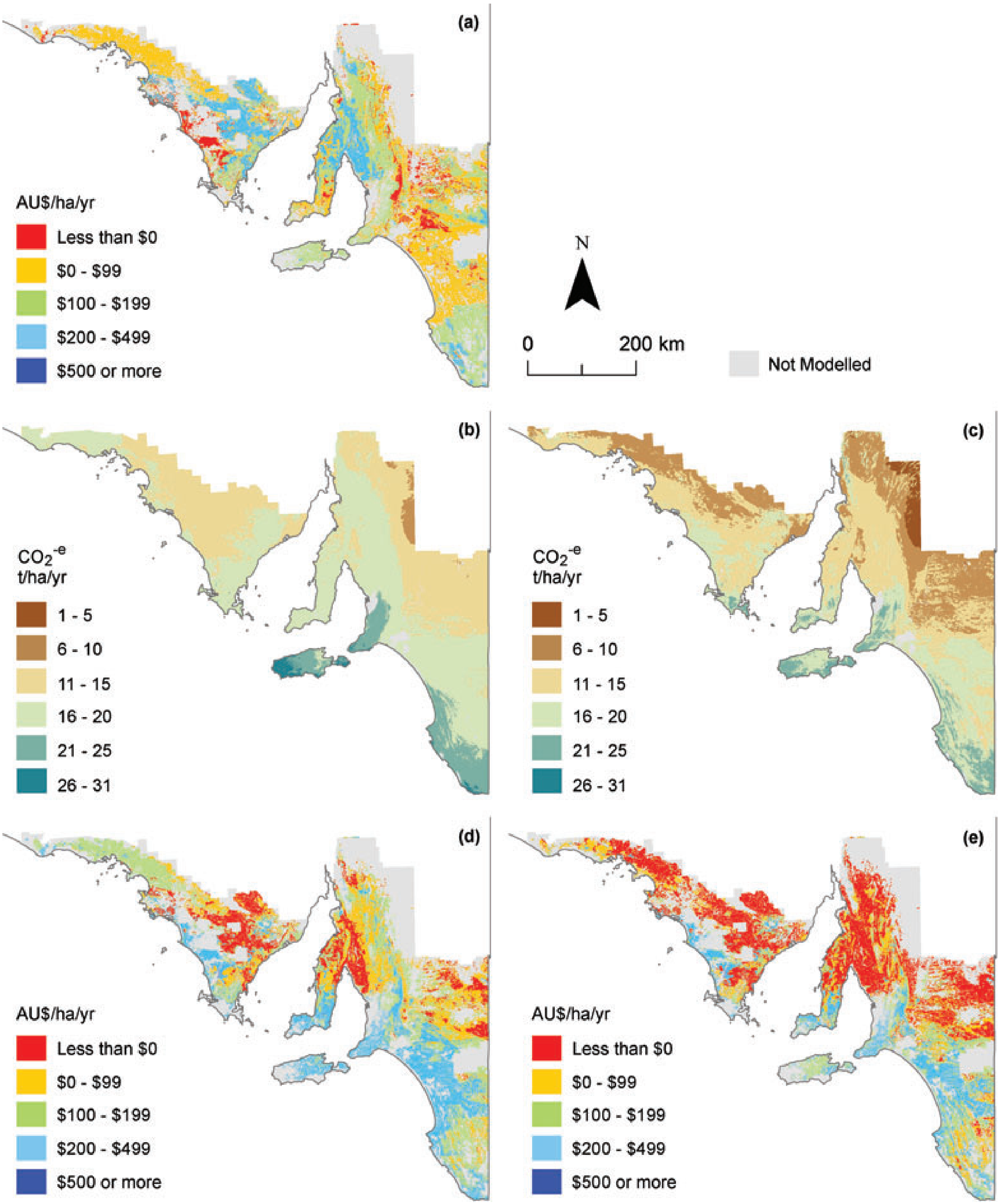

Figure 2. (a) Annual profit (PFE) of dryland agriculture in the study area, (b) annual carbon sequestered with monoculture carbon plantings, (c) annual carbon sequestered with ecological carbon plantings; (d) annual profit of monoculture carbon plantings at AU\$20/t $CO_2{}^{-e}$, and (e) annual profit of ecological carbon plantings at AU\$20/t $CO_2{}^{-e}$.

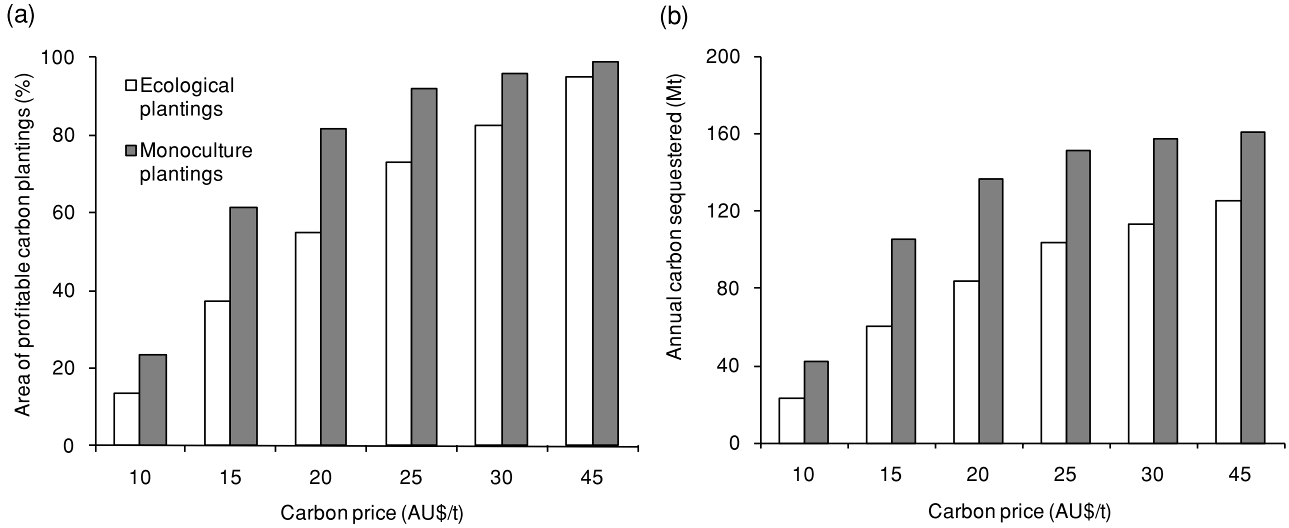

Figure 3. (a) Percentage of the study area in which carbon plantings are more profitable than existing agriculture under different carbon-price scenarios (AU$) and (b) annual carbon sequestered by carbon plantings if all profitable areas are planted.

more profitable if monocultures were planted than if existing agricultural uses continued, potentially sequestering 5.6 Gt of $CO_2{}^{-e}$ to 2050. The proportion dropped to 55% for ecological plantings, and 3.4 Gt of $CO_2{}^{-e}$ was sequestered by 2050. At $20/t $CO_2{}^{-e}$ and where it was profitable to establish carbon plantings, monoculture and ecological carbon plantings were more profitable than existing agriculture by $149/ha (SD 97) and $114/ha (SD 127), respectively (Figs. 2d & 2e). The profitability of carbon plantings (Fig. 3a), and hence the total area and $CO_2{}^{-e}$ sequestered, increased as carbon price increased (Fig. 3b) when it was assumed carbon plantings were established where they would be profitable.

An area of 1.1 million ha was identified as high priority for ecological restoration (Fig. 1b). In these locations profitability of monoculture plantings and ecological plantings differed (Fig. 4). At a low carbon price of $10/t $CO_2{}^{-e}$, on average an annual payment of $7/ha (SD 3) to the landowner would be needed to make ecological plantings competitive with monoculture plantings. The average annual payment increased to $50/ha (SD 28) at a medium carbon price of $20/t $CO_2{}^{-e}$ and increased to $125/ha (SD 88) at a high carbon price of $45/t $CO_2{}^{-e}$ (Fig. 4). In relative terms, the ecological restoration annual payment as a proportion of monoculture profit declined as carbon price increased. At $10/t $CO_2{}^{-e}$ the annual payment equaled 50% of the profit from monocultures, whereas at $20/t $CO_2{}^{-e}$ and $45/t $CO_2{}^{-e}$ the annual payment equaled 32% and 20% of the profit from monocultures, respectively.

The total payment needed to compensate landowners to convert priority locations for ecological restoration that are profitable for monocultures to ecological carbon plantings was $170 million at $10/t $CO_2{}^{-e}$, and the payment increased to $1.8 billion at $45/t $CO_2{}^{-e}$ (Fig. 5). The potential carbon sequestered up to 2050 with monoculture carbon plantings in the profitable locations was 269 Mt at $10/t $CO_2{}^{-e}$ and 780 Mt at $45/t $CO_2{}^{-e}$. At a lower total sequestration by 2050 with ecological carbon plantings, 232 Mt was sequestered at $10/t $CO_2{}^{-e}$ and 634 Mt was sequestered at $45/t $CO_2{}^{-e}$. As carbon price increased, the cost of ecological plantings increased and the amount of carbon sequestered decreased (Fig. 5). Nevertheless, the proportion of priority locations for ecological restoration converted to ecological carbon plantings increased from 30% to 96% as carbon price increased from $10/t to $45/t $CO_2{}^{-e}$.

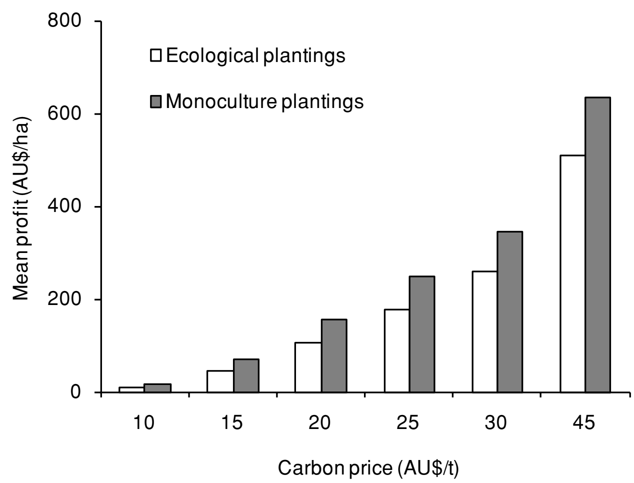

Figure 4. Mean profit (AU$) from ecological and monoculture carbon plantings within priority locations for ecological restoration.

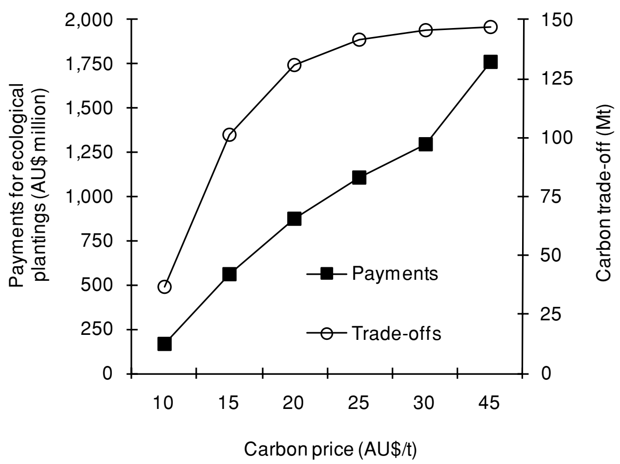

Figure 5. Total payments (AU$) to landowners to convert priority locations for ecological restoration that are profitable for monocultures to ecological carbon plantings and the associated carbon trade-offs (i.e., the difference between amount of carbon sequestered with monoculture carbon plantings and ecological carbon plantings).

Discussion

There is a large opportunity cost of converting agricultural land to native vegetation for conservation and restoration. It is also expensive to plant structurally and compositionally diverse native species because of the large effort required to collect and germinate seed from diverse sources and plant in a strategic manner (Pannell et al. 2006; House et al. 2008). Unlike in Europe and the United States, there is little direct financial compensation available to Australian farmers for providing public environmental goods and services (House et al. 2008). Our model shows that a market for carbon offsets could provide sufficiently high compensation for converting existing land uses to tree plantings for carbon sequestration. An additional, although relatively small, incentive would be needed for landowners to establish lower-yielding ecological carbon plantings in locations that are a priority for ecological restoration.

Well-designed economic instruments may provide an incentive for landowners to sequester carbon and to revegetate substantial tracts in temperate agricultural systems. Establishment of species conservation and biodiversity banks and credit trading is one economic instrument (Fox & Nino-Murcia 2005; Bekessy & Wintle 2008) through which ecological carbon plantings earn credits equal in value to the difference in economic returns between monoculture and ecological carbon plantings (Fig. 5). If this instrument is implemented, the landowner would trade the carbon sequestered in the carbon market. The ecological benefits of ecological carbon plantings would be sold separately to a biodiversity or conservation bank at a flat rate or could be auctioned through a competitive bidding process such as a tender with potentially greater efficiencies to the bank (Connor et al. 2008). Auctions of biodiversity and conservation credits are a more cost-effective way of allocating resources to conservation on private land than a fixed-priced scheme (Stoneham et al. 2003; Connor et al. 2008). Alternatively, regulations could be implemented that mandate the use of ecological carbon plantings in locations that are a high priority for conservation and ecological restoration, and such regulations could be combined with compensation in the form of simple annuity payments (Shogren et al. 2003), performance-based payments (Ferraro & Kiss 2002), or payments for ecosystem services (Jack et al. 2008). The size of these payments would need to be sufficient to compensate for the lower-yielding and less profitable ecological carbon plantings (Fig. 4).

International instruments such as the Kyoto Protocol's Clean Development Mechanisms (CDM) and the proposed Reduced Emissions from Deforestation and Forest Degradation (REDD) create incentives that may negatively affect biological diversity and reduce ecosystem complexity (Putz & Redford 2009; Stickler et al. 2009). Instruments aimed at maximizing tree-based carbon sequestration may encourage replacement of old-growth forest and native nonforest cover with fast-growing monocultures (Putz & Redford 2009). These potentially unwanted outcomes could be avoided by compensating landowners for lost income when they maintain old-growth forest.

There may be other substantial risks associated with widespread carbon plantings, whether monoculture or ecological. Increased tree cover in agricultural landscapes reduces runoff and groundwater recharge which then reduces water availability for human consumption (Zhang et al. 2001; Farley et al. 2005; Jackson et al. 2005). Additionally, widespread replacement of productive agricultural crops with trees can affect food security (Tilman et al. 2009). It will be important to account for these trade-offs when designing policy that encourages carbon plantings.

Our model shows that ecological restoration may be possible to effect relatively cheaply across wide tracts of agriculture. Nevertheless, we believe that it will require economic instruments that provide incentives for landowners to earn income from carbon offsets.

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