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Auctions for multiple environmental outcomes, from desk to field in Victoria, Australia

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Abstract

This paper reports on an Australian pilot (*EcoTender*) of an auction for multiple environmental outcomes. The success of the auction is due to a linkage between the auction process and an innovative Catchment Modelling Framework (CMF) used to estimate *multiple* environmental outcomes including carbon, terrestrial biodiversity aquatic function and saline land area.

Auctions have been used in the past to distribute environmental funds. BushTender, a single dimension auction (one environmental outcome) demonstrated significant cost savings are achieved when compared to other grant based approaches (Stoneham *et al*. 2003). In general, auctions aim to provide private landholders with the incentive to truthfully reveal their cost of undertaking specified actions that produce environmental outcomes. If correctly applied auctions can help to overcome common problems involving *asymmetric information* – where landholders have information about the cost of undertaking an action but this information is hidden from the agency who is providing the funds. The agency needs both cost information from landholders and information about the environmental outcomes (*missing information*) provided by the proposed landuse change, to make decisions between environmental management options and allocate funds.

This is the first time a market-based policy has been fully integrated from *desk to field* with a biophysical modelling framework for the purchase of multiple outcomes. The CMF incorporates a suite of one-dimensional plant-based models that are explicitly linked to a fully distributed 3D-groundwater model. This framework solves the *missing information* problem of linking paddock scale landuse and management to catchment scale environmental outcomes. The framework also incorporates a number of biodiversity algorithms that estimate current and future eco-system benefits.

This framework provides the Victorian government with a replicable transparent evidence-based approach to the procurement of environment outcomes. The framework can be applied in any location if data are available for calibration and validation.

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Introduction

In open, decentralised economies markets are the primary institution through which individuals/firms engage in transactions that create value. Within the economic system, these individuals/firms search for transactions that maximise their private or collective well-being.

Markets have not, however, evolved in all areas of the economy. Even though markets are missing in some areas this does not diminish the need for individuals to engage in value creating transactions - it is just that markets have not taken root to facilitate these transactions. The usual approach in these situations has been for government to step in with command and control approaches. This has traditionally defined the scope of government activities and we observe a strong presence of government in sectors such as the environment, health, education etc.

Where markets for the environment are missing or inefficient economists argue that the welfare of society is reduced. Generally this is observed as a fall in income but in the case of the environment, if means that total well-being is diminished. If markets are missing for environmental goods and services, resources are likely to be over-allocated to exploitative activities, such as land clearing (where there are clear signals to investors), and under-allocated to conservation activities (eg. nature conservation). Understanding why markets have not evolved to deal with the environment is an important step in designing mechanisms that will efficiently allocate resources to the conservation of environmental goods and services.

Like other markets, we know that many individuals are willing to pay for an increase in environmental goods and services. We also know that there are many opportunities to invest in activities that supply these services. What then stops these transactions from taking place?

Bardsley *et al.* (2002) notes that ideas about why markets are missing or inefficient have changed over time. The authors note that Coase (1937) identified ‘transaction costs’ as the main obstacle to the existence of markets. Today this vague concept is better understood, and it is known that information problems lie at the root of most missing markets. Once this is understood, there is the possibility of addressing the problem directly through the use of modern technology and clever institutional design. The basic reason that asymmetric information destroys markets is that it is hazardous to do business with someone who has relevant but hidden information. In general terms, environmental problems bear similarity with the market for ‘lemons’ (Akerlof, 1970). Akerlof showed that the existence of asymmetric information (that is, where one party is informed about aspects of the economic problem and the other is not) can render some seemingly competitive markets inefficient. In the limit, this phenomenon can result in the non-existence of markets. The uninformed party, in many environmental problems, is liable to be exploited, and may be unwilling to participate. Because of this, the potential benefits of doing business (which may be very large) may not be realised.

The economic literature suggests that there are four broad ideas that explain why markets generally, and the environment specifically, might be missing: inappropriate property rights, externalities and public goods, asymmetric information and asset aggregation problems (Ausubel, 2001, Ortmann, 2003).

In the context of environmental problems this suggests that markets for some environmental goods and services might be created if relevant information is discovered and shared between buyers and sellers of these goods and services. A better understanding about the information problems that have been preventing markets from evolving for environmental goods has prompted both the research and development of the information required (linking landuse change with environmental outcomes) and the mechanisms needed to reveal other information (costs) needed for value creating transactions to occur.

Latacz-Lohmann and Van der Hamsvoort (1997) explain how one of these problems (information asymmetry) affect the functioning of markets for environmental goods and services associated with private land. They note that there is a "clear presence of information asymmetry in that farmers know better than the program administrator about how participation (in conservation actions) would affect their production plans and profit". It follows then that markets should be able to be created by addressing these information problems. By attending to a) mechanisms that reveal information from landholders (auction format and design) and b) disclosure of scientific information to inform purchasers about the quantum of services provided by bidders, Stoneham *et al.* (2003) show it was possible to create a market. They concluded, "The pilot auction (BushTender) has shown that it is possible to create at least the supply side of a market for nature conservation and in conjunction with a defined budget, prices can be discovered and resources allocated. Characterising nature conservation on private land as a problem of asymmetric information has improved our understanding of why this and related environmental markets are missing or ineffective and has introduced an alternative policy mechanism to those currently available." Further, this approach demonstrated that cost savings of up to 7 times are achievable when compared with previous grant based systems in the same area. BushTender focused on one environmental outcome, terrestrial biodiversity, for which the "*habitat hectare*" approach was applied along with other biodiversity-related information to help solve the missing information problem (Parkes *et al*. 2003).

This paper reports on the next advance in the application of market-based instruments to environmental problems associated with private land-use. It reports on a the information needed to conduct a pilot[4] multiple-outcome auction (EcoTender) where the purchaser is provided with information about the impact of land-use change from four environmental dimensions (carbon sequestration, aquatic function, dryland salinity impacts and terrestrial biodiversity). The Catchment Modelling Framework (CMF) was developed to estimate the environmental impacts of these multiple environmental outcomes and to spatially represent these to potential bidders and the purchaser (Victorian Government) of these services. This approach offers the prospect of improving the cost-effectiveness of the single dimension auction, possibly beyond that achieved by BushTender and reducing the costs of providing information about the impact of land-use change thereby reducing transaction costs.

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[4] The pilot is being conducted by the Victorian Department of Primary Industries and the Department of Sustainability and Environment.

The auction approach explicitly recognises the heterogenous nature of landholders opportunity costs to undertake alternative land management. The auction allows landholders to determine the payment they require in order to undertake the agreed management. The environmental outcomes also vary between landholders for the same management. Past modelling approaches have adopted large homogenous land areas assuming the environmental outcomes within the area are the same for all landholders. The CMF models management at the scale it occurs (farm/paddock), explicitly accounting for the heterogenous nature of the environmental outcomes. If heterogeneity exists it is possible to get more environmental outcomes for a given environmental budget.

A number of studies have suggested that conservation programs have been inefficient because they have focused on on-site information rather than environmental outcomes. For example Ribaudo (1986) argues that conservation programs have focused on on-site physical criteria, such as soil erosion, rather than the benefit to the environment – the environmental outcome. Wu and Boggess (1999) show that in the presence of environmental thresholds allocating conservation funds based on on-site physical criteria could result in little environmental benefit. In both cases there was very limited empirical scientific data to support their findings. Ribaudo relied upon qualitative empirical analysis of one environmental benefit (erosion and water quality) to demonstrate his argument. Wu and Boggess used theoretical models to demonstrate their point but highlighted the need for empirical models to inform investment decisions.

There is a growing recognition that environmental outcomes are correlated – benefits are jointly produced by the same action. For instance, revegetation may jointly produce carbon, improvements to water quality and wildlife benefits. Wu and Bogess (1999) refer to this as an ecosystem-based approach that recognises the interaction between alternative environmental benefits. They show that an efficient fund allocation must account for both physical production relationships between environmental outcomes and the value of those outcomes. Wu and Skelton-Groth (2002) developed an empirical model to demonstrate the extent of fund misallocation when jointly produced environmental benefits are ignored.

The catchment modelling framework presented in this paper focuses on providing the missing information linking environmental outcomes with actions on private land. The framework provides empirical estimates of correlations between environmental outcomes and explicitly links on-site landuse changes with off-site environmental outcomes. The framework has been designed to explicitly model and report the joint production of environmental outcomes which links effectively with policy to efficiently allocate conservation funds.

This paper discusses and illustrates a new empirical framework for the assessment of multiple environmental benefits followed by a discussion of metrics used to estimate environmental outcomes. Results and discussion are presented in the final sections followed by recommendations for further research and conclusions.

Review of past modelling frameworks

In order to address the missing information issues, a review of contemporary catchment scale models was undertaken to identify a potential framework/s capable of assessing the site specific and off-site environmental outcomes arising from alternative land management.

The framework needs to operate at the appropriate resolution to link farm scale landuse change to offsite catchment scale impacts. Further, the model needs to report transparent measures of environmental outcomes. For instance, a change in aquatic health may contain measures of erosion to stream and litres of water to stream, were both impact on aquatic health in different ways. The final aquatic outcome may, for example, be the product or addition of these measures. In order to determine preferences for environmental outcomes the framework needs to be transparent in the manner in which measures are combined.

In the past, physically based one-dimensional simulation models have been used to evaluate the production and environmental aspects of farming systems, including the amount of deep drainage lost lost below the plant root zone (Coram and Beverly, 2003). The amount of excess water available (defined as rainfall less soil evaporation and plant water use) includes deep drainage and surface runoff which should be partitioned into recharge to the deeper groundwater and lateral flow to stream. This partitioning is important because the vertically dominated recharge pathway results in very different environmental outcomes to the laterally dominated flow pathway.

Past studies using 1D farming systems-models have assumed deep drainage contributes only to, and is analogous to, groundwater recharge. For instance, the Liverpool Plains study (Paydar et al., 1999, Ringrose-Voase and Cresswell, 2000), identified large anomalies between recharge estimates based predominantly on deep drainage predictions derived using one-dimensional models compared with those derived based on groundwater hydrograph responses. These anomalies are directly attributable to the lack of partitioning and the lack of accounting for lateral flow processes.

The SHE model (Danish Hydraulic Institute, 1991) attempted to account for partitioning by explicitly linking farming systems models with groundwater models. However this model operates on a regular grid (representing both surface and groundwater) and adopts a generalised vegetation algorithm (Kristensen and Jensen, 1975). The grid approach limits the models ability to describe spatially varying land units which may exist at a finer scale than the regular grid cell and consequently forces the user to homogenise each grid cell to only one landuse. Additionally, the same vegetation algorithm is used to describe each landuse with different parameter sets. This limits the models ability to simulate phenological plant responses, which is important when predicting grazing/livestock interactions and pasture competition. Alternatively, the USDA soil and water assessment tool (SWAT) also uses a generalised vegetation algorithm to simulate landuse. However, SWAT does not preserve spatial resolution and does not explicitly model distributed groundwater dynamics, but rather adopts a lumped parameter approach (Neitsch *et al*., 2001).

In contrast to the physically-based catchment models described above, generalised approaches based on average annual relationships between evapo-transpiration demand and rainfall have been developed (Holmes and Sinclair, 1986, Zhang *et al*., 1999). Recent studies have adopted these empirical relationships to assess the impact of landuse change on mean annual runoff for grassland and forest catchments (Vertessy and Bessard, 1999, Zhang *et al*., 2001, Zhang *et al*., 2002). These models have limited temporal and spatial resolution to assess the impact of landscape intervention at the paddock/farm scale. Further, they are not explicitly linked to a distributed groundwater model, which is essential to estimate the groundwater discharge and off-site watertable impacts.

The Catchment Modelling Framework (CMF) was developed because none of the above approaches provided farming systems models that operated at the catchment scale and are explicitly linked to groundwater (Beverly *et al.*, 2003). Further, they did not provide transparent estimates of environmental outcomes nor the ability to combine biophysical information into environmental outcomes in a systematic manner.

The Catchment Modelling Framework

The CMF is an enhancement of the Catchment Analysis Tool (DPI, 2004) which incorporates a suite of one-dimensional farming systems models into a catchment modelling framework with modification to account for lateral flow/recharge partitioning. The CMF consists of an interface and a simulation environment. The interface is used to assemble time-series and spatial data sets for use by simulation models, visualisation and interpretation of data, and the interrogation of simulation outputs. The interface was designed to assist in both the pre- and post-processing of spatial and temporal data sets.

The interface is also used to apply rule-based methods to analyse landscape features. For instance, remnant native vegetation maps showing current coverage are used to assess the spatial significance of alternative revegetation options. Generally, this type of analysis is rule based (ie. patch size and shape, connectivity of remnant patches, distance from sources of refuge such as river corridors or sources of replenishment such as large patches of native vegetation,). In most cases the rules are developed based on current understanding of the spatial needs of relevant species and coded into the interface for application in different catchments. The interface was developed using MATLAB[5] and can be distributed as an executable to non-technical users and stakeholders.

The simulation environment is an assemblage of one-dimensional farming systems models capable of simulating pasture, crop, trees and a fully distributed 3-dimensional groundwater model[6]. The CMF is simulates daily soil/water/plant interactions, overland water flow processes, soil loss, carbon sequestration and water contribution to stream flow from both lateral flow (overland flow and interflow) and groundwater discharge (base flow to stream). The agronomic models can be applied to any combination of soil type, climate, topography and land practice. Using the interface, outputs from these simulations can be compiled for visualisation, interpretation and interrogation.

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[5] Information about MATLAB etc

[6] The groundwater model relies on third party software (MODFLOW). The simulation environment has been designed to produce scripts that automates the process of employing the third party software.

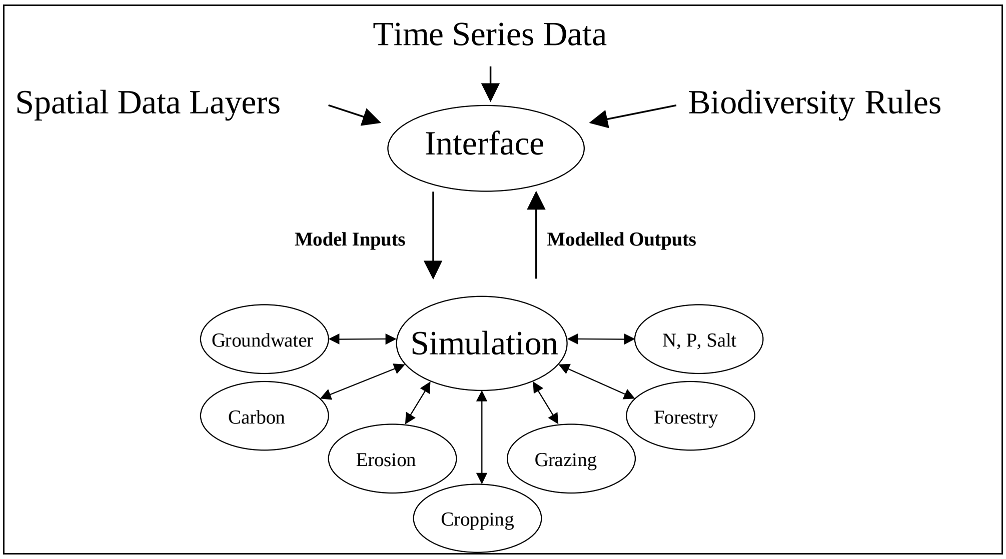

Figure 1: Models incorporated into the Catchment Modelling Framework

The CMF develops both a surface element network and a groundwater mesh based on unique combinations of spatial data layers. Typically the spatial data necessary to derive the surface element network includes soil, topography, landuse and climate. The groundwater model requires spatial data pertaining to aquifer stratigraphy such as the elevations of the top and basement of each aquifer, spatially varying aquifer properties and river/drainage cadastral information. Additional data includes time-series records of stream flow, groundwater hydrograph, groundwater pumping, and irrigation. Importantly the surface element network is typically at a finer resolution than the underlying groundwater mesh.

Model outputs

Outputs from the model can be characterised based on scale as either specific to the management scale (paddock/farm) or the sub-catchment to catchment scale (Table 1). As simulations predict soil/water/plant interactions on a daily basis, a comprehensive range of time-series outputs is available for each surface element. These include:

- complete water/soil balance (soil moisture, soil evaporation, transpiration, deep drainage, runoff, erosion),
- vegetation dynamics (crop/plantation yield, forest stem diameter, forest density, carbon accumulation).

At the sub-catchment to catchment scale outputs include:

- stream dynamics (water quantity and salt loads);
- groundwater dynamics (depth to watertable, aquifer interactions, groundwater discharge to land surface and stream).

Table 1: Model outputs

Land management scale	Catchment scale
Surface runoff	Groundwater discharge
Deep drainage	Stream salinity
Soil moisture content	End-of-valley response
Evaporation-transpiration	Groundwater pumping impacts
Later flow (runoff and sub-surface)	Zone of saturation
Sediment loss	
Biomass	
Harvest yield	
Carbon	
Salt	

The following section outlines how the CMF is used for the development and application of environmental outcomes adopted in the pilot study.

Environmental outcomes: converting model outputs

There is a growing recognition that multiple environmental outcomes are produced jointly by the one action. For instance, revegetation may jointly produce carbon, improvements to water quality and wildlife benefits. Wu and Bogess (1999) refer to this as an ecosystem-based approach that recognises the interaction between alternative environmental outcomes. Therefore, decisions about preferred investment for environmental benefit need to therefore reflect the inherently different functional characteristics of different ecosystems. Such investment decisions are often further complicated by the need to compare a range of actions across broad landscapes and different ecosystem types that may produce varying amounts of different outcomes of dissimilar value.

Wu and Bogess (1999) indicate that an efficient fund allocation must account for both physical production relationships between environmental outcomes and the value of those outcomes. Building on this concept, the EcoTender pilot uses an information framework that defines each environmental "outcome" in terms of 'service' or the change in the level of function resulting from the landholder actions and "significance" the value of that change.

To estimate the change in level of function, it is necessary to have a standard reference point against which change is measured. Adapting the policy approach applied in Victoria for assessing conservation status of biodiversity assets (NRE 2002), it was decided to use pre-1750 as the "natural benchmark" against which current ecosystem function and change in function arising from landholder management actions in the catchment can be assessed. Under such an approach, the pre-1750 landscape is modelled using the assumed pre-European settlement vegetation types to provide an understanding of native vegetation cover both current and prior to clearing. The current and pre-1750 modelled landscapes can then be used to measure changes in landscape function resulting from landholder interventions based on a progression towards 1750. In this context, the pre-1750 "function" is not a target but simply a reference point for measuring change. The pre-1750 benchmark approach is also used to estimate the change in native vegetation quality or extent resulting from landholder actions (see below).

Actions

Landholder actions in the pilot are limited to indigenous revegetation and improved remnant native vegetation management. Revegetation requires the establishment of indigenous species in formerly cleared areas to achieve required target based on the modelled pre-1750 vegetation types for the site. Remnant native vegetation management involves landholder commitments that improve the vegetation quality of the site as assessed in comparison to a ‘benchmark’ that represents the average characteristics of a mature and apparently long-undisturbed state for the *same* vegetation type (Parkes *et al*. 2003, DSE 2004).

Indigenous Revegetation

Revegetation is limited to Ecological Vegetation Classes[7] (Table 2 below shows examples from the total 38 used) based on the pre-1750 vegetation maps of the region (Woodgate *et al*. 1996, Parkes *et al*. 2003, DSE 2004). This requires that landholders agree to minimum standards including type, species and target densities (based on an EVC benchmark), site preparation and follow-up management.

Table 2: Examples of EVC groupings applied by the model.

Bioregion	**Description**	**Target Trees (plants /ha)**	**Target Large Shrub (plants/ ha)**	**Target Medium Shrub (plants/ ha)**	**Target Small Shrub (plants /ha)**	**Target Large Tufted Graminoid (plants/ ha)**	**Target Total (woody plants/ ha)**
Goldfields	Heathy Dry Forest	100	50	1000	1500	500	2650
Goldfields	Heathy Woodland	50	0	1200	2000	0	3250
Goldfields	Floodplain Riparian Woodland	50	50	200	100	500	400
Goldfields	Box Ironbark Forest	100	0	1000	500	0	1600
Goldfields	Grassy Woodland	50	0	600	500	500	1150
Wimmera	Ridged Plains mallee	50	0	200	1000	500	1250
Wimmera	Semi-arid Woodland	50	0	600	2000	0	2650
Wimmera	Lignum Wetland	0	0	800	0	0	800

Where:

Trees = overstorey species (usually > 10m tall)
Large shrubs = sub-canopy species > 5m tall
Medium shrubs = shrubs 1-5m tall
Small shrubs = shrubs 0.2-1m tall
Large tufted Graminoid = non-woody grass-like plants > 1m tall

To evaluate the change in each outcome the catchment model was calibrated to pre-1750 EVC vegetation cover and extent and simulations were undertaken for 44 years based on 1957-2000 historical climate data. Each of the EVC types (Table 2) was characterised on the basis of varying root depth, root densities and over and understorey canopy dynamics.

Remnant Native Vegetation Management

Remnant native vegetation is defined as established vegetation of a type (EVC) relevant to that which existed in 1750, prior to settlement and clearing. The aim of remnant native vegetation management is to improve the condition of the vegetation

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[7] Ecological Vegetation Classes – EVCs are the level at which native vegetation has been mapped across Victoria. In general, EVCs are defined by a combination of floristics, life form, position in the landscape and an inferred fidelity to particular environmental attributes

through landholder commitments that maintain and/or improve the quality of indigenous vegetation on the site. This may include foregoing entitled uses such as firewood harvesting and grazing (fencing) or active management beyond current obligations under legislation such as weed control, pest animal control and supplementary planting of understorey species.

Outcomes

An output is the direct result of an action as estimated using the CMF. For instance, the action of replacing pasture with indigenous trees results in a measurable output such as a reduction in recharge at the site. In the context of this project we are interested in the outcome that would result from the restoration and maintenance of remnant vegetation including a reduction in recharge – thus the importance of connectivity within the landscape. For example will the fall in recharge contribute to reducing the amount of saturated land or reduce the amount of saline water entering a stream as base-flow.

The outcome used to assess the bids is limited by available scientific information. For instance, a reduction in recharge can be described in the following steps.

1) Fall in recharge
2) Fall in saline discharge to stream from groundwater
3) Reduced impact on riverine flora and fauna
4) Followed by an assessment of the significance of the flora and fauna within the context of local and regional stream networks. The final outcome could be an aggregate of the service provided to both riverine flora and fauna and adjusted for river significance.

Currently there is very limited data available to complete steps 3 and 4. In order to score an outcome it is usually assumed that there is a positive linear relationship between steps 2 and 3 and the measure used at step 2 is an accurate proxy for 3.

Estimating the outcome is a more appropriate measure of the impacts of land use/management intervention. For instance there may be two sites located within a catchment where recharge was estimated to be reduced by 40mm due to revegetation. However, when the outcome is calculated as the change in groundwater contribution to the stream, the same recharge results in a 10mm and 25mm change in base-flow outcome. There may be a number of reasons for this including, location in the catchment with respect to the stream, soil type, groundwater characteristic and slope. By measuring the outcome this pilot focused on improving the quality/quantity of landscape elements thereby meeting environmental objectives. Reducing the threat – recharge – does not necessarily lead to an improvement in both the outcomes and the environmental asset/s of concern.

Stock and flows

The outcomes that result from land use change or management actions need to be assessed as either a change in stock or a change in flow. For instance, a reduction in recharge may result in less saturated land affected by rising groundwater when the water table has reached a new equilibrium. The change in saturated land at equilibrium is the benefit of intervention – a reduction in the stock of saturated land. Alternatively the change in saturated land could be viewed as a flow of benefits through time. As the water table approaches equilibrium there is less and less saturated land until equilibrium is reached. On reaching equilibrium there is a constant flow of benefits – the change in saturated land equivalent to the change in stock measure of saturated land.

If all actions resulted in a permanent and instantaneous change, it is possible to compare benefits based on changes in stock. However, if the form of intervention results in a time dependant outcome they may be more accurately compared based on the flows.

Figure 2 below shows the outcome resulting from two actions with respect to time. Action A is revegetation with native species and action B is revegetation with commercial forestry with harvesting at regular intervals. Action A provides increasing benefits up to T1 reaching a maximum of A_{max}, and remaining at A_{max}. Action B provides increasing benefits up to B_{max} (where $A_{max} = B_{max}$) but then declines following harvest and rises back up to B_{max}. The decline in benefits from action B arises when the trees are harvested. Typically this type of benefit flow is observed for groundwater discharge and carbon accumulation.

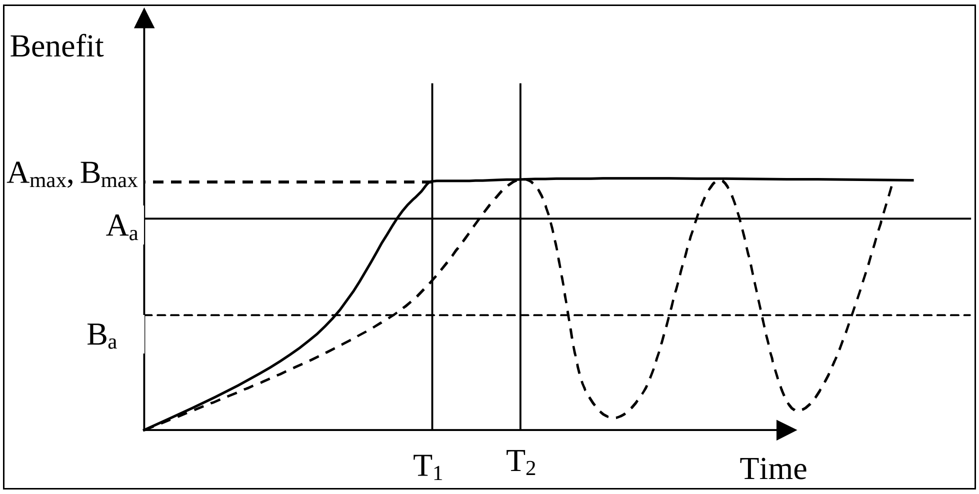

Figure 2. Benefit flows and time

For action A equilibrium was reached at T1 and for action B a temporary equilibrium was reached at T2. If the actions were compared as stocks at T2 they would be evaluated as having equal benefit, A_{max}. However, this approach does not account for the variability of the benefits provided through time by action B after time period T2.

Instead, if the actions are compared as the average benefit at a point in time greater than T2 the benefits measures would be A_a and B_a resulting in A_a ranked as providing greater benefits than B_a. Instead of using an average, the flow benefits could be discounted to reflect present value. Further research is required to determine the appropriate approach and time periods.

It is assumed that actions in the pilot are permanent in so far that both revegetation and remnant management will be ongoing. Further, the nature of the actions results in a continuous flow of benefits up to a maximum similar to example A – revegetation. Therefore when applying the CMF to determine the change in outcome a steady state solution was adopted to estimate the long term equilibrium condition under the altered vegetation/management regime.

Steady state approach to estimating outcomes

The steady state solution derived using the CMF model represents the long-term equilibrium condition within the pilot region arising from locally modified vegetation/management regimes. This condition exists when the water table has reached a point were the fluctuations in inflow and outflow remain relatively constant through time. When applying the CMF the steady state can be achieved in two ways. Firstly the CMF can be run for a long time horizon whilst observing the variation in GW flows and level. When this variation between successive periods reaches a user-defined threshold (based on a minimum variance between current and last period or past average), both the inflows/outflows and groundwater levels are reported as representing steady state conditions. Issues with this approach include computational time, climatic variations between years and the lag time between water entering and leaving the groundwater storages. The level of variance a user is willing to accept determines the steady state solution. Further, if there is a prolonged period of either high or low rainfall the system may be exhibiting steady state properties (low variance) however it is a product of the rainfall. This is particularly noticeable for extended low rainfall periods when groundwater inflows are very low or next to zero and outflows are constant for long periods. This would exhibit itself as a local solution rather than a global solution.

Alternatively, the surface (unsaturated zone) can be run over a long time horizon and all recharge and surface flow information can be stored for later input to the groundwater model. Climate variation between periods stills exists however the groundwater lag is isolated. Instead the long run average of recharge and surface flows in used as the input to the groundwater model and the steady state solution is then run until the groundwater has reached equilibrium. This approach overcomes the local solution issue and requires much less computational time. This approach was adopted in this pilot.

Saline land

Saline or saturated land is commonly defined as the area where depth-to-watertable is less than 2 meters. The groundwater height was estimated using the CMF model and the area of land classified as saturated or impacted by waterlogging was defined as those regions where surface elevation (based on a digital elevation model) less groundwater height was within 2 metres. The service score is the change in saturated land area (ha). The metric for change in saline land is the sum of the change in land area within 0.1, 0.5, 0.8, 1.0, 1.5, 2.0 m of the groundwater. The steady state approach is used to estimate the area of land.

The significance can be determined by the importance of that land within the catchment context. For example under current conditions there may be 525 ha classified as saturated. Following the implementation of the action, the amount of saturated land is reduced to 515 ha – the service score is 10ha. The significance of the 10ha is determined based on current use. For instance the 10ha may include cropping, roads, buildings and wetlands. However, in order to determine the overall significance preferences need to be explicitly expressed for each land type. Preference information was not available in a form that could be applied systematically in the pilot. Rather preferences for the pilot have been expressed as an equal weighting for each land type, reducing the outcome score for saline land to change in area alone. That is, the final metric for saline land is the fall in hectares of land within 2 metres of the water table.

Aquatic Function

Aquatic function is particularly challenging because it needs to take into account groundwater (GW) flows to stream, surface water (SW) flows to stream and the quality of both. SW and GW steady-state contributions to stream were calculated for both pre-1750 EVC coverage and current land use. The SW volumes were based on both the surface and sub-surface lateral flow contributions to stream. The GW contribution to stream includes groundwater loss to stream and groundwater discharge volumes to surface, and in turn to stream.

To assess the impact on in-stream biodiversity it was necessary to consider the relative volume and quality of SW and GW streamflow contributions. However currently there is very little science available to provide repeatable and transparent interpretations of the impacts on flora and fauna due to various flow regimes and varying ratios of SW and GW streamflow contributions. Therefore the following approach was adopted and is an adaptation or extension of the steady state principle used for the saturated land area assessment. It is recognised that this approach has been developed in the absence of clear scientific relationships between surface water flows to stream and pollutants and their relative impact on riverine flora and fauna.

Within the pilot catchment the groundwater is saline and it was assumed that a fall in saline emissions to stream may provide a benefit to the flora and fauna. Similarly, a fall in surface water arriving at stream was assumed to reduce the amount of nitrogen, phosphorous and sediment, which benefits riverine for flora and fauna. Further a change in flow timing and magnitude towards pre-1750 conditions was assumed beneficial to riverine health. .

The modelled pre-1750 landscape assumes that in-stream biodiversity in the pilot catchments were adapted to the prevailing conditions at that time as determined by the contributions from ground water and surface water. That the greatest change in these elements under current practice is due to surface water contribution indicates that a reduction in SW contribution to stream is considered of greater importance than that of GW. Further to this, SW contributions to stream have altered the timing of peak and low flow periods and the temperature of the water – both of which contribute to the viability of in-stream biodiversity.

Currently within the CMF it is possible to examine the temporal aspects for changes in water volume with and between years however nutrients are not reported. As a proxy for nutrients changes in erosion arriving at stream are reported and combined with the changes in water. As such, the final metric used in the pilot for aquatic function is the product of water quantity (sum of both SW and GW mm/annum) by erosion (t/ha).

Terrestrial biodiversity

Remnant native vegetation management

Habitat service - There are a number of actions that landholders can take to maintain or improve the condition or extent of habitat on private land. These include foregoing entitled uses such as firewood harvesting and grazing; active management of threats beyond current obligations such as control of weeds and pest animals or supplementary planting of species-deficient areas. The value of these actions can be expressed as a Habitat Services Score (HSS) where HSS_i represents the change in quality and quantity of habitat at a Site “i”. The Habitat Services Score (HSS) measures the amount of terrestrial biodiversity improvement offered by the various landholder management commitments.

Biodiversity significance - Landscapes that have been modified for agricultural purposes will not necessarily retain a representative mix of habitat types and will generally contain biodiversity assets at varying levels of depletion and naturalness. One way of expressing the conservation value of different sites is with a Biodiversity Significance Score (BSS) where BSS_i represents the biodiversity value of '*Site i*'.

The BSS rates each site according to its conservation value. The BSS depends on the type and quality of native vegetation on the site and its relative conservation status[8], the plants and animals that may use the site as habitat, and the position of the site in the broader landscape and its contribution to maintaining or improving the regional native vegetation context for a range of important mobile fauna species.

Conservation status is determined using concepts of rarity and degree of threat (NRE 2002). Vegetation quality uses the 'habitat hectares' approach of Parkes et al. 2003, which assesses the vegetation according to a number of site-based attributes (e.g. tree cover, understorey diversity and cover, weediness, amount of regeneration, amount of organic material etc.) and a number of local landscape attributes (size of patch and amount and configuration of surrounding native vegetation).

Each of the site-based attributes is assessed and scored against a benchmark that represents the average characteristics of a mature and apparently long-undisturbed state for the same vegetation type (Parkes *et al*. 2003). The landscape context (LC) score for each site is determined using a mathematical algorithm that provides a measure of the current amount and relative distribution of native vegetation within the vicinity of the site (Ferwerda 2003). The landscape context algorithm is based on the general principles that large, round patches (high area : perimeter ratios) provide the best opportunity for ecological processes to be maintained; and remnants that are surrounded by other remnants or connected to larger remnants by 'links' or 'stepping stones' provide better habitat opportunities than isolated remnants.

The landscape context (LC) layer is combined with some additional spatial rules to derive the Biodiversity Landscape Preference (BLP) layer. The LC layer is weighted to reflect those parts of the landscape where both the requirement for restoration and "function" of native vegetation restoration activities are optimised. These are typically areas located between the most intact landscapes where the functionality of restoration is greatest but where the requirement for restoration is least, and the most fragmented landscapes where the requirement for restoration is greatest but the functionality of restoration is least. The weighted LC layer is combined with rules relating to patch size and shape, connectivity of remnant patches, distance from sources of refuge such as river corridors or sources of replenishment such as large patches of native vegetation to derive the BLP. These rules have been derived based on current understanding of the future spatial needs of key mobile fauna species.

The BLP layer is effectively an assessment of the future spatial considerations of restoration. It provides a relative preference for different parts of the landscape as a measure of their potential role in restoring broader landscape function.

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[8] Using Ecological Vegetation Classes that have been assigned a bioregional conservation status such as endangered, vulnerable, depleted or rare.

The BSS uses information held in corporate (government) databases, LC and BLP maps and site-based information to verify what is on the site. The metric used in the pilot is the product of HSS and BSS.

Revegetation

In principal the scoring of revegetation is the same as remnant native vegetation management. The service score is determined by a combination of size of the site and its impact on the amount and configuration of native vegetation in the local landscape and the estimated change in vegetation condition of the site. The former is a measure of the change in landscape context (LC) resulting from the revegetation while the latter applies a fixed score to revegetation that meets a minimum required standard based on the EVC benchmark.

The significance score uses the same approach as remnant native vegetation management except that the role of the site as habitat for plants and animals is not assessed. The metric used in the pilot is the product of HSS and BSS.

Carbon

The carbon outcome is calculated for each site by estimating the change in accumulated carbon (t/ha) between the current condition and the established EVC at maturity. Both the benchmark and current condition account for different spatial vegetative cover, canopy and root development for each vegetative class.

The is no significance measure for carbon because it is a diffuse pollutant. However, the location significance of the revegetation is captured in the significance scoring of terrestrial biodiversity. The metric used in the pilot is the tonnes of carbon sequestered at each site.

The following table summarises the outcomes used in the pilot.

Attribute	Change in level of service	Desirable change	Significance
Terrestrial Biodiversity	Δ habitat score (habitat maintained or improved per ha)	Increase	Biodiversity conservation significance , threatened species conservation status, habitat quality, landscape preference
Aquatic function	Δ water “quality” (tonnes of soil / ha arriving at stream) Δ water quantity (mm of water / ha arriving at stream)	Decrease	(not in pilot)
Saline land area	Δ saline land (ha with groundwater < 2m)	Decrease	can discriminate - but equal weighting in pilot
Carbon sequestration	Δ carbon sequestered (tonnes / ha)	Increase	n/a

Table 3. Summary of outcomes, service and significance

The terrestrial biodiversity, aquatic function and saline land outcomes are summed for each site to produce the Environmental Benefits Index (EBI). Carbon is dealt with as a market good and landholders are paid separately for each unit produced. The selection of bids is based only on the EBI and the cost of the bid, farmers adjust their bid given the knowledge they will receive carbon payments if their bid is accepted.

Validation and implementation

The pilot was run in two sub-catchments in Victoria, namely the Avon Richardson[9] (371,000ha) and Cornella (47,000ha) (Figure 3). Catchment selection was based on data availability, the areal extent of any proposed land use change, the type of management considered by land managers and a requirement that the focus catchment be a priority region as identified by the appropriate state authorities. The landscape also needed to be topographically and climatically variable and the catchment also needed to be unregulated (not controlled by in-stream structures and no diversions for other uses such as irrigation) and monitored so as to provide continuous streamflow and water quality data to underpin model calibration and validation. Additionally, catchment selection was based on the presence and quality of time-series groundwater observation data, which is used to conceptualise and validate the groundwater dynamics.

The current landuse in the Avon Richardson comprises 52% cropping, 37% grazing, 6% trees and the remaining 5% constituting urban infrastructure and water bodies. Annual rainfall ranges from 350 to 765 mm/year. In contrast the current landuse in the Cornella catchment comprises 53% cropping, 26% grazing, 20% trees and the remaining 1% constituting urban infrastructure and water bodies. Annual rainfall ranges from 450 to 670 mm/year

Figure 3. Pilot areas

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[9] The following discussion is limited to the Avon Richardson but the same process occurred for the other catchment.

For each spatial vegetation coverage, discrete land units across the catchment were defined based on soil, slope, climate, landuse, land management and elevation. Each land unit varied in size ranging between several hectares to tens of hectares and was connected to an underlying groundwater model. Assigned to each land unit was a biophysical farming system model simulating daily soil/water/plant interactions .

The calibration procedure adopted a split sample test with non-overlapping calibration and verification periods. The calibration strategy applied to pre-scenario conditions between 1957 and 1995 whereas model verification was assessed on data measured between 1996 and 2000 inclusive.

Calibration of the framework was based on matching measured catchment yield and salt export, stream dynamics, selected groundwater hydrograph responses, depth-to-watertable information and mapped groundwater discharge areas. Stream flow analysis techniques were applied to measured stream gauge data to derive quickflow[10] (overland, sub-surface and groundwater surface discharge) and groundwater baseflow (groundwater flows into streams) estimates. The calibration criterion compared these quickflow and baseflow time-series data sets with predicted volumes to calculate goodness of fit based on 44 years of historical climate data.

In the case of the Avon-Richardson catchment, the simulated area of groundwater discharge was 16,200 ha which was in agreement with the mapped 15,500 ha. Groundwater mean annual baseflow was simulated to be in the order 250-300 ML/year, which was also in agreement with gauged stream flow data. The validation process of the CMF has produced results consistent with measured stream flow and recharge estimates (Beverly *et al.*, 2003, Paydar and Gallant, 2003, Tuteja *et al.*, 2003, 2004).

Field validation

In order to undertake field validation the CMF was used to assess outcomes in terms of saline land area, aquatic function, soil loss (erosion) and terrestrial biodiversity under both current and pre-1750 landuse. The pre-1750 condition was based on Ecological Vegetation Class (EVC) description of vegetation cover (Parkes and Newell, 2003, see Table 2 above).

The CMF systematically simulated the impact of changing landuse to pre-1750 on 25ha parcels of land across the entire catchment whilst assigning current landuse to all other land units[11]. The resultant predictions were assembled as spatial maps and were used to identify "*hot spots*" within the catchment where a 25ha change in landuse/management had the greatest impact on (a) groundwater discharge volume to stream, (b) groundwater discharge volume to surface, (c) change in depth to watertable and (d) surface flow volumes to stream.

Based on the predicted changes field validation was undertaken aimed at assessing the robustness, resolution and appropriateness of the simulation predictions derived using the CMF modelling approach. For instance three *hot spots* were identified as high

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[10] Quickflow is the some of overland flows and near surface flows following immediately after a rainfall event.

[11] There are approximately 14975, 25ha parcels in the Avon-Rchardson.

impact locations. In these cases they were situated in the low lying area of the catchment and showed a high impact on off-site saline land. The sites were visited to determine if they were high groundwater areas and if revegetation with indigenous species would result in a significant fall in saline land. The first site was under cropping and tile drainage systems had been installed indicating that the area was subject to water logging. Water logging is synonymous with high discharge (either through infiltration excess or groundwater) thus indicating the site would be a good place to revegetate and reduce saline land, supporting the modelled results. The other two sites were currently used for grazing but there was remnant vegetation remaining. The remnant vegetation was swamp species also indicating the area was subject to water collection and water logging, again supporting the modelled results.

A number of other sites were visited to compare modelled (based on aerial/satellite imagery) landuse with observed landuse. As expected there were a number of anomalies between the broad-scale spatial data sets and field observations of landuse. In order to ensure the modelled results of landuse change are representative, field officers were provided with the ability to validate and modify modelled landuse whilst undertaking a site visit.

Pilot implementation

The following table outlines the steps taken to implement pilot. The pilot was advertised through regional radio and newspaper media calling for expressions of interest from interested landholders.

Table 4 . Implementation Steps

1. Expressions of interest – landholders located in project areas register an expression of interest through their EcoTender field officer.
2. Site Assessments – the EcoTender field officer arranges a site visit with each registered landholder. The field officer assesses the site and advises the landholder on the significance of the site from a range of environmental perspectives, and identifies potential native vegetation management and revegetation options for consideration by the landholder.
3. Development of Management Plans – landholders identify the actions they are prepared to undertake and the field officer prepares a five-year management plan as the basis for a bid.
4. Submission of bids –landholders submit a sealed bid that nominates the amount of payment being sought by them to undertake the agreed management plan.
5. Bid Assessment – all bids are assessed objectively on the basis of:

- the estimated change in the on and off-site environmental outcomes (the amount of change in environmental outcome);
- the value of the assets affected by these changes (significance);
- dollar cost (price determined by the landholder).

Funds are then be allocated on the basis of ‘best-value for money’.

6. Management agreements – successful bidders are able to sign final agreements based on the previously agreed management plan (from 3 above).
7. Reporting and Payments – periodic payments and reporting occur as specified in the agreement.

Implementation step 2 required field officer training and the development of an interface to estimate the environmental outcomes. Given the spatial nature of the pilot a system was devised whereby field officers entered GPS data into a hand held device (IPAQ), which was later down-loaded for use in the CMF.

For each site field officers used the IPAQ to collect and store GPS coordinates, record current landuse or EVC, and record detailed information about the current condition of the EVC (tree density, logs present, weeds, pests, etc). This was followed by a discussion with the landholders about actions (management plans) that could be undertaken to provide environmental outcomes. Field officers would indicate what type of actions were best suited and the minimum standard required. The field officer would then down-load the information into the CMF for validation of the associated data (current landuse or EVC) and then calculate the environmental outcomes of the proposed landuse change.

Results

The following results are limited to pre-implementation, as field application commenced in early June 2005 and final assessments are not currently available. Therefore, the results are limited to the assessment 25ha parcels used in the validation exercise which underpin the pilot implementation. It should be noted that the magnitude of the changes are not linear in so far that, the sum of the impacts arising from landuse change on any two parcels may result in a greater change than the addition of the impacts derived from each parcel. That is to say outcomes are area and spatially dependent. The distribution and magnitude of results reported here would be different if smaller or larger parcels of land were simulated. The results for the key components are presented below.

Aquatic function

For aquatic function the SW and GW contributions to stream were calculated under steady state for both pre-1750 coverage and current land use across the entire catchment. The SW is based on both the surface and sub-surface lateral flow contributions to stream. The GW contribution to stream includes groundwater loss to stream and groundwater discharge volumes to surface, and in turn to stream. Modelled results indicate that changing landuse to pre-1750 condition across the entire catchment would result in a reduction in recharge of 89,000 ML/year and a 35,000 ML/year reduction in lateral flow to stream relative to current condition.

Table 5. SW and GW contribution to stream, EVC pre-1750 and current landuse

Catchment coverage	Surface water contribution to stream (GL/year)	Ground water contribution to stream (GL/year)	Mean annual total stream flow (GL/year)
EVC pre-1750	19 (91%)	2 (9%)	21
Current landuse	54 (67%)	27 (33%)	81
Percent change (pre-1750)	+184%	+1250%	286%

Table 5 shows the SW and GW contributions to stream under current and pre-1750 conditions. Notably there has been a significant increase in both flow regimes relative to pre-1750 conditions. Surface water flows increased by 184% due to tree clearing and the introduction of pasture and annual cropping enterprises. There has been a correspondingly very large increase in groundwater flows to stream (1250% increase). Therefore any reduction in surface water and groundwater contributions to stream, a movement towards pre-1750, is considered desirable (based on the benchmark approach discussed above).

If scientific evidence were available to describe the relative impact of each type of water it may be desirable to weight them, however for this pilot no weighting’s were attached. For each of the 25ha parcels modelled the sum of the change in GW and SW ranged between 0 and 30 ML with an average response of 4.1ML assuming steady state conditions. Changes in erosion to stream varied between 0 and 0.36 tonnes per ha/annum. Based on these results the final aquatic outcome varied between 0 and 42 averaging 4.2.

Change in saline land area

Changing landuse to pre-1750 condition on discrete 25 ha units was predicted to reduce the area of saline land by 8.5ha on average with the maximum impact being approximately 125 ha depending upon landscape position and groundwater characteristics. Figure 4 below shows a map of the variation in impact. This type of map is also used by field officers for communication with farmers to indicate the relative importance of their site (farm) within the landscape.

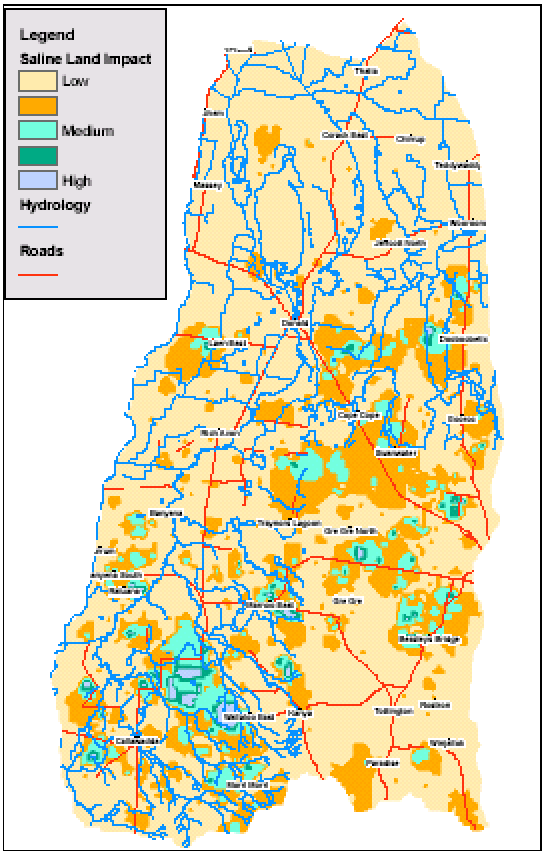

Figure 4. Change in saline land impact

The groundwater impacts were then compared to the changes in recharge for each site to see wether recharge is an accurate proxy to make decisions about investment in the landscape. Of the 14,975 cells modelled, the correlation was 27% between a change in recharge and a change in saline land area.

The impacts of a change in recharge vary across the catchment as a function of the underlying groundwater characteristics and groundwater flow directions and gradients. For example, simulated results of the impact on groundwater discharge volumes to stream arising from the reintroduction of remnant vegetation show that the greatest impacts are associated with the south-easterly region of the Avon-Richardson catchment. This is not surprising as the groundwater flow direction is in a north-westerly direction such that reafforestation in areas located in the north-westerly portion of the catchment have little impact on groundwater discharge volumes to the dominant river system located in the centre of the catchment. That is, a unit change in recharge (arising from reafforestation) in the north-westerly zones of the catchment have very different impacts relative to a unit change in recharge in the south-easterly regions of the catchment. As a general rule a unit change in recharge will have a different impact on saline land depending on where it occurs in the catchment. This result suggests recharge is not a suitable proxy for investment when considering the off-site impacts (saline land area) of landuse change.

Terrestrial biodiversity

A priori, it is not possible to report the biodiversity outcomes because the habitat service score requires a site visit to determine the current condition of the site and to assess particular biodiversity assets (e.g. habitat for rare or threatened species). However, components of the biodiversity significance score, biodiversity landscape preference (BLP) and landscape context (LC) can be examined because they are modelled using existing information on native vegetation extent and configuration. BLP ranged between 0 and 90 and LC ranged between 0 and 23.

Change in carbon

Carbon sequestration ranged from 0 to 34 kg/m^2 averaging 13 kg/m^2. The total amount of carbon sequestered is driven primarily by the EVC replacing current practice. Figure 5 below is a map of the change in sequestered carbon (t/ha) arising from replacement of current landuse with pre-1750 vegetation for the Avon Richardson sub-catchment.

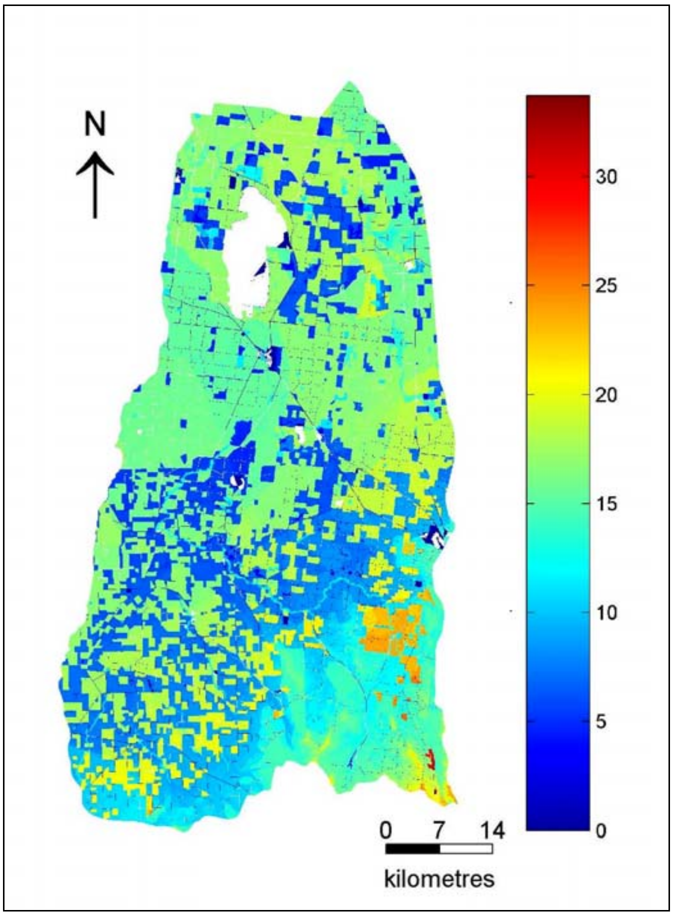

Figure 5. Sequestered Carbon

Joint production and heterogenous outcomes

One of the key motivations for developing the CMF was the hypothesis that environmental outcomes are jointly produced and this feature might improve the cost effectiveness of funds allocated to the environment. In order to determine if outcomes are jointly produced a random sample of sites were assessed for saline land, carbon, terrestrial biodiversity and aquatic function. These sites were then sorted to determine wether they were producing more than one outcome – for the single action revegetation. Analysis of the simulation results derived for all sites with the pilot suggest that 73% generate two or more environmental goods supporting the hypothesis that environmental outcomes are jointly produced from a single landuse change.

Given outcomes are jointly produced there may be scope to reduce total costs if outcomes are correlated. For instance the use of one outcome as a proxy for others may reduce the level of model reporting and complexity. This may save time and reduce the transaction costs associated with estimating outcomes. In order to test if outcomes can be used as proxies for one another the outcomes are tested for spatial correlation.

The table below shows the correlation matrix between the metrics for aquatic function, saline land, carbon and the significance indices for terrestrial biodiversity, for the whole catchment.

Table 6. Whole of catchment spatial correlation matrix

	Aquatic Function	Carbon	Saline Land	BLP	LC
Aquatic Function	1				
Carbon	0.17	1			
Saline Land	0.16	0.06	1		
BLB	0.03	-0.07	-0.09	1	
LC	0.09	-0.06	-0.17	0.64	1

Results presented in Table 6 suggest that there is a very low correlation at the catchment scale between outcomes, and as such we would expect a lot of variability in the EBI (sum of outcomes) estimates reflecting landscape variability.

From Table 6 it can be observed that there is a positive correlation (0.17) between carbon and aquatic function. This is due to a number of biophysical factors. Firstly, revegetation generally sequests greater amounts of carbon than current practice and revegetation has a strong influence on surface water dynamics. For instance revegetation reduces surface water runoff, erosion and recharge, all of which are used to calculate the aquatic function outcome. In this pilot results indicate that revegetation produces both carbon and aquatic benefits 17 percent of the time.

BLB and LC are correlated because they are based on the same base data set (spatial location of current native vegetation) but are not identically correlated because they represent different landscape function and attributes. LC is focusing on the current existence whilst BLB focuses on the future impact of terrestrial biodiversity management or revegetation. They are examining different aspects of *eco-system* function, current function and future function given landuse intervention.

The CMF is shown to provide *ex ante* data on expected outcomes. There is a temptation to use this data to target areas with the aim of reducing the number of site/farm visits thereby saving time (reducing costs) or achieving greater outcomes (areas with *ex ante* high outcome scores).

The following is an example of targeting areas of the catchment based on high outcome scores. Figure 6 below shows the histogram for aquatic function outcomes for each site within the catchment (approximately 1.4 million units each of 50*50 metre resolution). Using tools built into the CMF specific areas of the histogram can be remapped by selecting a range.

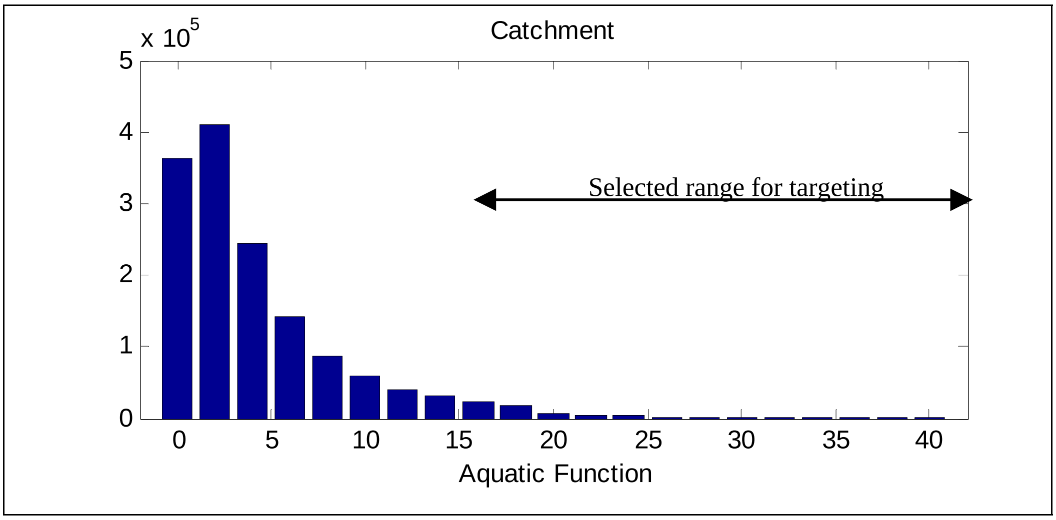

Figure 6. Catchment - Aquatic Function Histogram

For this example land areas that scored aquatic function greater than 15 were mapped to show their location within the catchment (see Figure 7 below). This shows there is a concentration of land in the south east of the catchment scoring high for aquatic function. It may be possible to target these areas for land use change reducing the costs by not visiting other areas of the catchment, were aquatic function the primary outcome of interest. However, it was shown above that there is a very low correlation between outcomes, so targeting this area may reduce the overall quantum of outcomes.

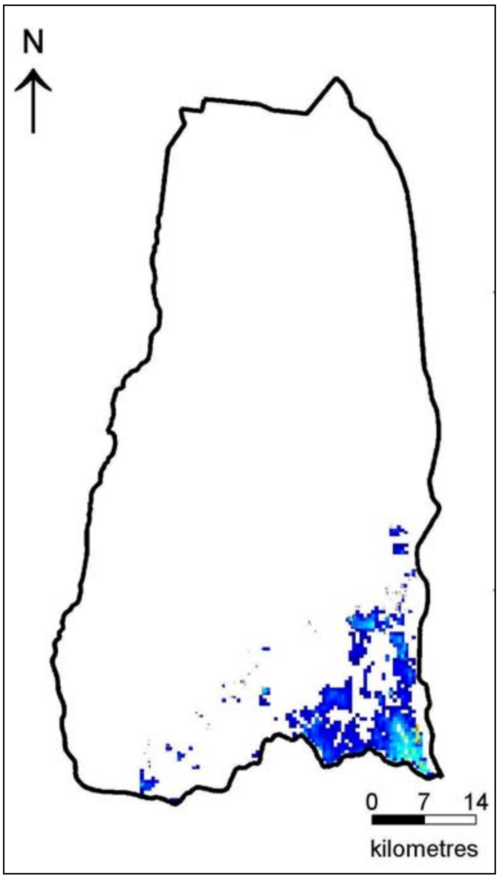

Figure 7. Targeting high scoring aquatic outcomes

It may be possible to identify other areas with a lower aquatic function score but increase the scores of one or more of the other outcomes, generating greater outcomes in aggregate, assuming the purchaser is indifferent between outcomes.

Implementation and training

A possible barrier to adoption of the CMF is its scientific complexity. The framework needed to be used by field officers either on site or locally. An interface was developed with the field officers that enabled them to down-load site information into the CMF for processing. The interface provided the officers with the ability to validate landuse (both current and proposed) run the biodiversity algorithms and finally report the outcomes.

One field officer was assigned to each sub-catchment and they undertook 8 hours training in the use of the interface. One of the officers had previously conducted single outcome assessments (biodiversity) using a paper based system, which they found to be time consuming with significant potential for error. Further it was very difficult for them to trace the process if bids needed to be altered or for audit purposes. They reported the interface to be non-threatening and there is no longer the need for reams of paper to complete the biodiversity assessment because the CMF had been programmed to complete the process with their input. On average site visits are taking one day to complete. This includes travel, site assessment, post processing data and administration.

Discussion

The CMF has significantly reduced the transaction costs associated with accurately determining environmental outcomes for any site within the landscape. The CMF can be readily calibrated to any catchment providing there is sufficient data for calibration. Further, the framework can be readily updated, as new data becomes available.

Generally, fixed-price grants based programs have focused on one environmental outcome and required information to support spatial allocation decisions, or worst still allocate funds based on lowest cost without any consideration of outcomes. The CMF has reduced transaction costs and accounts for multiple environmental outcomes.

The CMF has incorporated biophysical processes to account for erosion, water, carbon, saline land to estimate environmental outcomes. Further the landscape context (LC) considers the current location of native vegetation and the biodiversity landscape preference (BLP) considers the future spatial needs of key mobile fauna species. The CMF is the only framework (the authors are aware of) that has brought together both types of information.

The framework has demonstrated the importance of joint production in environmental outcomes and the heterogenous nature of the landscape in terms of aggregate outcomes. This information has been incorporated into an auction-based approach (EcoTender) offering the possibility for significant cost savings.

Results presented in this paper demonstrate that a unit change in recharge (arising from reafforestation) has very different impacts on saline land depending on where it occurs in the catchment As such recharge is not a proxy for saline land area when considering the off-site impacts of landuse change. The use of recharge as a proxy would reduce the cost effectiveness of available environmental funds, if saline land area were the objective.

The correlation results presented in Table 6 and those specific to aquatic function (Figure 6 and Figure 7) indicate that the CMF is capable of exploring the trade-offs between metric. However, targeting areas based on outcomes alone, ignores the cost side of the problem. It may be the case that all high cost land use changes are located in south east of the catchment. If cost effectiveness is the objective (minimising the total cost per unit outcome) then it may be beneficial to go elsewhere in the landscape.

It is not until the cost information is available that a decision can be made about the most cost-effective distribution of funds across the catchment. Using outcome information alone may result in much higher total costs.

Interpreting the biophysical information into economic costs – for instance converting yields to opportunity cost – is tempting but potentially very costly, as it ignores the heterogenous nature of landholders costs. In many instances biophysical information has been used to estimate costs for targeting purposes.

Data from previous auctions for conservation contracts show that when landholders were engaged in a competitive bidding process for conservation contracts, their bids displayed much larger variation than can be explained by variation in land capability (Stoneham *et al*., 2003). The average bid per hectare in BushTender was $274/ha but the standard deviation of bids was $349/ha. Whilst there was some variation in the quality of land between bidders, the auction was confined to a relatively homogeneous (with respect to agricultural production) Box Iron Bark vegetation classification. This result is significant because it means that the cost of land-use change is different on each farm and this information needs to be truthfully revealed rather than estimated using bio-economic models that treat landholders as homogeneous agents.

If outcomes are appropriately developed to reflect the importance of location now and in the future and account for all on-site and off-site impacts, the only remaining exercise is to employ a mechanism that reveals the true cost of making the changes, hence the use of auctions.

The CMF was developed to support a pilot project and as such, there are a number of areas that would benefit from further research and effort. From an economic point of view there has been no account for diminishing returns or preferences between outcomes. The framework has shown the dependency between spatial locations for individual sites but has not included empirical approaches (for instance synergies between sites) to exploit the opportunity for further cost savings.

In the sub-catchments the pilot is operating in no water is used for productive purposes. A reduction in stream flows could have deleterious economic impacts if the water is collected and used for productive purposes. There are a number of policy issues that need to address the trade-off between water for environmental purposes and water for productive or consumptive purposes if this approach were to be adopted in an area used for water collection.

Conclusion

The methodology developed in this study links landuse and management with biophysical crop growth and environmental processes on a site-specific basis with the capacity to assess the off-site impacts at both the field and sub-catchment scales. This approach accounts for spatial variability and connectivity within the landscape. Results presented in this paper demonstrate the value of adopting a holistic catchment modelling framework to inform a market-based auction process. The project applied the model framework to estimate multiple environmental outcomes, both on-site (biodiversity, erosion and carbon sequestration) and off-site (catchment yield and water quality), arising from landuse change at the land management scale. This approach provides a predictive capacity to estimate and extrapolate the likely outcomes arising from alternative land management.

The framework has shown that recharge alone is not a suitable metric for the allocation of environmental funds for the prevention of saline land. Further the CMF has shown that the use of a single metric is not sufficient to capture the heterogeneity of landscape change at the farm scale. Combining this information with auctions for landuse change provides the opportunity to purchase environmental outcomes more cost effectively than current grant based approaches.

The Catchment Modelling Framework provides policy makers with a new tool to analyse landscape intervention and make informed decisions about the outcomes resulting from investment at the paddock scale. The framework is practical and feasible for application is the field and provides a cost effective, replicable and transparent method for the assessment of environmental outcomes to support programs for the allocation of environmental funds.

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