Some efficiency aspects of native vegetation and biodiversity regulations: A supplementary submission from the Australian Government Department of Agriculture, Fisheries and Forestry to the Productivity Commission Inquiry into the Impacts of Native Vegetation and Biodiversity Regulations

- The spatial heterogeneity of the public benefits of native vegetation and biodiversity regulations means that targeting is essential for benefits to be achieved.
- The spatial heterogeneity of the private costs of regulation means that flexibility in the application of regulations is required if the costs of regulation are to be minimised.
- The spatial interdependence of biodiversity values within a region means that the actions taken or disallowed within a region need to be considered jointly.
- Attempting to achieve biodiversity benefits at the scale of individual properties is unlikely to be cost effective.
- Threshold effects and diminishing returns mean that maximising the area of habitat affected for a given budget is not generally equivalent to maximising the environmental benefits of habitat provision.
- The effectiveness of conservation planning depends critically on the accuracy of the data used. Regional processes could make better use of local knowledge, especially where the processes are seen to be equitable.
- Although transaction costs should be minimised this is not an issue that can be considered in isolation from other policy design criteria.
- Covenants can provide a high degree of security or certainty and have other advantages such as potentially being voluntary and generally acceptable to the community.
- Given uncertainty over the private costs of natural resource management activities the only practical solution may be to strive to minimise payments by using instruments, such as competitive tenders, designed to elicit information on such activities.

Public policy makers face the complex problem of how best to obtain the highest value to society, over time, from the use (including conservation) of Australia's natural resources. The use of natural resources provides Australia with enormous economic benefits, contributing significantly to the country's income and standard of living. Past resource use, which may have been considered appropriate at the time (given available information and prevailing social values), has also left a legacy of resource degradation such as salinity and biodiversity loss.

The potential cost of addressing resource degradation is large and the impacts on regional communities are likely to be significant. Given the large cost involved with fixing all of the resource degradation, it may not be in society's best interests to repair all of the degradation. Instead, it will be important to determine a framework for prioritising and focusing efforts in areas where the benefits of repairing damage exceed the costs. This will ensure that the benefits from the public investments being undertaken are maximised.

The major approach used by governments to address these problems is the use of regulations to govern resource use. This may take the form of restricting activities that typically reduce environmental and biodiversity values. Any constraining regulations have an opportunity cost because resource managers are no longer able to carry out potentially profitable activities. There may also be regulations requiring that certain activities be undertaken in given circumstances. In this case there is a direct cost of undertaking the specified activities. In each case these costs can be seen as a type of forced investment by land managers in the production of public goods such as biodiversity conservation, carbon sequestration or water quality.

Other approaches to addressing natural resource degradation include those that fund or subsidise activities that contribute to the provision of these public goods. Choosing the best mix of regulations, subsidies and other policies requires that a range of options be compared against a range of criteria including:

- the effectiveness of the policy in delivering explicitly stated objectives,
- the efficiency of the policy in minimising the full costs of achieving the objectives and in achieving an appropriate balance between the objectives achieved and the costs,
- the ability of the policy to adjust to changed conditions and information over time and in different locations,
- the security provided to resource managers for future investment, and
- the distributional effects of the policy.

This submission focuses on improving the efficiency of native vegetation and biodiversity regulations although some important welfare and equity considerations are noted. Given that the environmental benefits generated by regulations are largely public and not exclusive to, for example, landholders, important welfare and equity considerations arise when the cost of these investments in the environment are borne by landholders.

Current policy setting: Regulations

Objective of regulations

Native vegetation and biodiversity regulations aim to deliver environmental benefits by constraining resource management options. The regulations aim to restrict some activities while at the same time requiring other activities to be implemented in given circumstances. A wide range of activities are affected by regulations, including land clearing, regrowth management, revegetation, weed and pest management and agro-forestry.

Designing regulations

Apart from the equity concerns, there are a number of issues concerning the efficiency and effectiveness of native vegetation and biodiversity regulations. To be efficient, regulations need to be able to differentiate between different sites on the basis of their relative costs and benefits. However, both the private and environmental benefits and costs of restricting activities can vary significantly between regions and properties.

The level of environmental benefits and costs will be affected by such factors as the current quality of vegetation, the viability of the species or ecological communities in the region, the spatial configuration of the vegetation, future management activities, threshold effects, diminishing returns and economies of scope. These factors are discussed further below.

Similarly, the private costs and benefits to landholders undertaking or stopping these activities will also vary across sites depending on access to markets and site quality, and through time depending on the weather, market shifts and technical change.

The environmental benefits of restricting an activity at one site might be sufficiently large to outweigh the cost to the landholder of not being able to undertake that activity. In another area, the environmental benefits may be low, or the opportunity cost to the landholder very high, such that restricting the activity may result in a net loss of value to society. Efficient environmental policies need to exploit this variation by targeting activities with the greatest net benefit. When, as is often the case, regulations do not differentiate between sites where restricting the activity will have a net benefit and sites where it will have a net cost, they become less efficient. This needs to be balanced against the increased transaction costs of differentiating between sites.

Investing in biodiversity

For public investment in biodiversity conservation to be cost effective it must achieve the best biodiversity conservation outcome possible for the given level of costs. The first obstacle to achieving this is the lack of a clear statement of the biodiversity conservation objective. Once a clear objective is agreed, different policy options can be compared according to their contribution to this objective, a full assessment of their costs and other criteria as mentioned above. However, the starting point must be a clear statement of the objective.

For the purpose of examining the cost effectiveness of investment in biodiversity conservation it is assumed that the biodiversity objective is a weighted sum of the viabilities of species and ecological communities:

$$V = \Sigma w_i V_i$$

Where w_i is the weight and V_i is the expected viability of the species *i* or community *i*. Viability for a species or community is defined as the probability that the population size or area at the end of some time interval exceeds a given threshold level. A simple case would be where all native vertebrate and vascular plant species whose biology is reasonably well known are included and all the weights are equal. Westphal and Possingham (2003) and Montgomery et al. (1999) discuss cases where only bird species are included and Possingham et al. (2002) use the number of species saved as a simple approximation of *V*. In this case all weights are one and the viabilities are either zero or one. However, varying weights may be appropriate for reasons of aesthetics, ecological function or utility.

This specification of the biodiversity conservation objective is consistent with the nature of the biodiversity conservation task. It allows actions that affect the viability of a number of species to be accorded a greater value than those with a much narrower effect. Similarly, including a broad range of species is essential when species have conflicting habitat requirements. Restricting attention to only a few focal species makes it unlikely that many species habitat requirements will be met and may mean that significant potential biodiversity benefits are missed, particularly when the focal species are the most sensitive or endangered (Westphal and Possingham 2003).

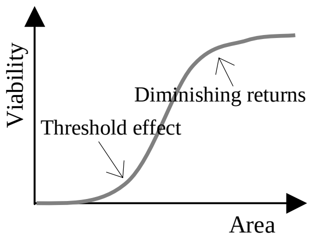


Figure 1. A species viability curve.

Figure 1 illustrates the relationship between the area of good quality habitat for a species and its viability. If actions affecting the habitat area when the species is currently on either the far right hand side or far left hand side — that is, where the species is either secure or critically endangered — are considered then the action may have little impact. However, if the species is on the steeper part of the curve, which may correspond to an endangered, vulnerable or of-concern status, such actions would have a larger impact. Thus, while ignoring the most secure species or ecosystems may not compromise the efficiency of biodiversity investments, prioritising according to degree of endangerment will not yield the best outcome in terms of viability across species. For a given level of public investment there are likely to be species that are too costly to recover when compared with the alternative of increasing the viability of a larger number of species.

Another important factor is the spatial interdependence of biodiversity values within a region. A species' viability for a given area of habitat depends on the size and shape of habitat patches and the spatial configuration of those patches in the landscape. Thus the biodiversity value of a given patch of vegetation cannot be based solely on intrinsic characteristics of the patch. Similarly, for a given level of investment in a region, better biodiversity outcomes can be achieved if these interactions are included in integrated regional planning. Attempting to achieve biodiversity benefits at the scale of individual properties is unlikely to be cost effective.

An explicit, quantitative and flexible specification of the biodiversity conservation objective is necessary in order to examine the cost-effectiveness of conservation activities and regulations. Using such an objective also facilitates the incorporation of biodiversity conservation into regional planning processes and permits a balance to be achieved between the allocation of resources:

- to the different activities supported or restricted,
- to different species and communities, and
- across different regions.

For perhaps largely practical reasons, existing conservation investments and regulations have used a more piecemeal or ad hoc approach based on intermediate measures of biodiversity outcomes. For example, where only a fixed area of a given vegetation community is allowed to be cleared, the areas that are actually cleared are generally determined by the order in which landholders apply for permits rather than some more efficient rationing process.

The use of area targets is widespread in the implementation of biodiversity conservation policies in Australia. While the arbitrariness of many of these targets is often openly acknowledged the biases they introduce into conservation planning need to be examined. Such area targets are often determined as a given percentage of the estimated pre-European areal extent of the vegetation community. While this is a simple pragmatic approach it should only really be a temporary approach in those cases where there is no better understanding of the determinants of ecosystem viability. Such targets largely ignore critical aspects of the biodiversity value of vegetation as

– the current quality of the vegetation,
– the configuration of the vegetation, and
– other aspects of vegetation management such as fire, weed or feral species management.

Sattler and Creighton (2002) highlight the need to achieve an appropriate balance across a large range of actions that all affect biodiversity outcomes.

Even if these aspects were appropriately included, it is likely that there would be only a weak relationship between the areal extent of a vegetation type in say 1770 and the area required to achieve a given viability threshold. More fundamentally these area statements are only an intermediate measure of environmental benefits. Threshold effects and diminishing returns, as illustrated in figure 1, mean that maximising areas for a given budget is not generally equivalent to maximising environmental benefits. Wu and Boggess (1999) show how maximising the area can actually minimise the environmental benefits of a given investment. For example, this would be the case when limited funding is being allocated across two identical catchments where there is an increasing marginal cost of land and there are threshold effects. Maximising the area means splitting the investment equally across the two catchments. Because of threshold effects the environmental benefits are maximised by targeting the investment to one catchment first. Similarly, where the otherwise identical catchments differ in environmental quality due to past conservation actions, they demonstrate that when there are threshold effects limited investments are likely to generate greater environmental benefits if the 'cleaner' catchment is funded first. Maximising the area funded would target the lower environmental quality catchment first.

The Bush Tender Trial in Victoria and the Conservation Reserve Program in the United States are examples of the use of a managed auction process to more efficiently allocate public investment in biodiversity conservation on private land. The use of auctions has the benefit of being voluntary and permits lower costs to be achieved where there is an information asymmetry concerning the cost, including opportunity cost, of on-farm actions. The biodiversity indices used to evaluate the bids are an attempt to approximate a composite biodiversity index as above and take into account the quality of the vegetation being managed and some ongoing vegetation management actions. However, they fail to take account of the spatial interdependence of the environmental benefits of the bids — whether through spatial configuration effects or through the cumulative effects of habitat provision as illustrated in figure 1. Current research, such as the market based instruments pilot projects mentioned in the first submission, are beginning to address these concerns.

The Regional Forest Agreement (RFA) process adopted the objective of achieving quantitative biodiversity targets at minimum cost. These include area targets for vegetation communities as discussed above as well as targets for endangered and vulnerable species that were more closely linked to the biology of the species in question. A major drawback with this process of target setting was that it gave no guidance for prioritising investments when not all the targets can be achieved.

Opportunities such as where a small reduction in the viability of one community could be traded off for a large gain in the viability of another community were not able to be identified.

A key problem with conservation planning relates to the accuracy of the data used. In the RFA process some areas selected to represent endangered communities in an expanded national park system did not in fact contain the communities they were selected to represent. Importantly, there were processes to revise the allocations in the cases that were identified. Similarly this inquiry has heard of cases where the mapped vegetation underlying regulations were shown to be in error (eg S. Doust and L. Acton on pp. 57, 70 and 73 of the Brisbane transcript (Productivity Commission 2003)). While the National Vegetation Information System is making progress in providing a national vegetation database with consistent classifications and consistent quality, there is clearly a need for regional processes to be able to incorporate more accurate data as it becomes available, and there is scope for such processes to better incorporate local knowledge. This is likely to remain problematic when information relating to public goods such as biodiversity values can lead to private costs.

Another key problem is that the actual effects of vegetation management and biodiversity conservation actions on species and ecosystem viability are poorly understood. This is particularly the case for ecological communities. At the species level, population viability analysis is relatively well advanced if only for a small set of species. For the RFA process it was possible to derive crude viability measures for hundreds of species using modeling and expert knowledge. While current processes seek to make the best use of available knowledge there is clearly a need to be able to adapt strategies and actions as new knowledge is gained. The RFA process at least nominally specified a twenty year period after which the allocation decisions could be reassessed.

Water Issues

There are a number of dimensions to the water policy debate as it relates to native vegetation management. For the purpose of this submission, attention will be given to the water quality aspect of the problem, especially the salinisation of river and irrigation systems.

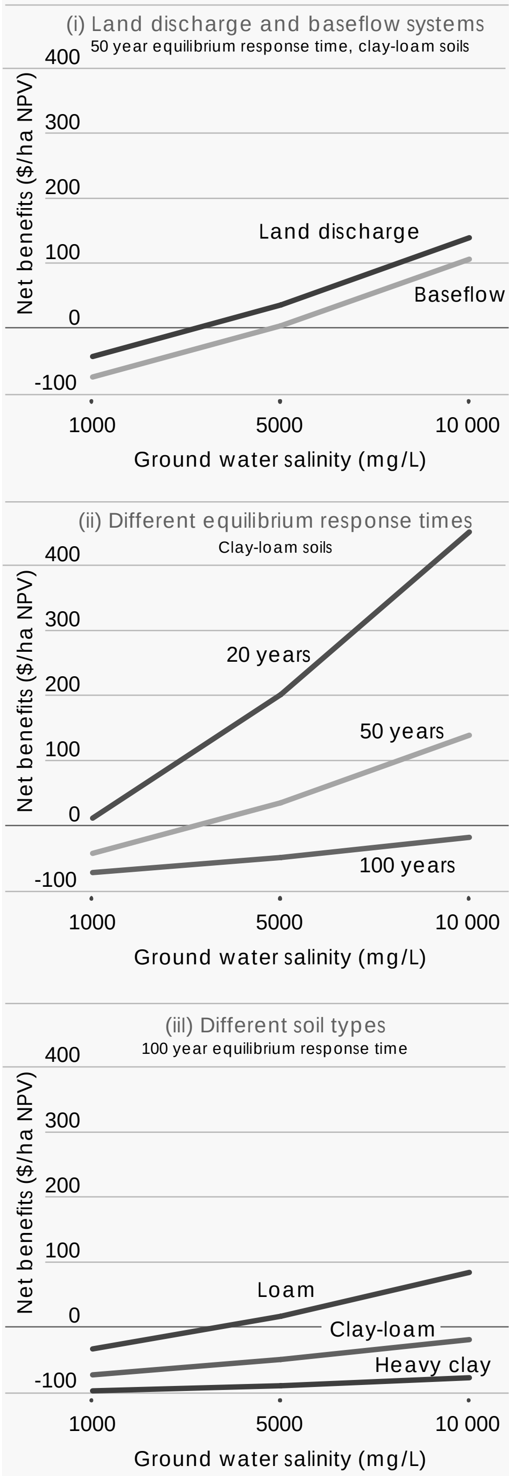
The scientific and physical causes of salinisation are complex, but relatively well researched. As a result of more water entering the water table than is being extracted, naturally occurring salty groundwater finds its way to the surface and into streams and rivers. This causes the overall load and concentration of salt in rivers to increase, reducing water quality both for the natural environment and for consumptive uses such as irrigated agriculture.

From an economic perspective, salinisation occurs because activities that cause, affect or mitigate salinity have spatial and temporal consequences not borne by those undertaking the activity. When this occurs, decision makers often undertake actions that are privately optimal, but may not be optimal from a social perspective. Hence in Australia, many actions that have a negative impact on water quality have been undertaken, while actions that have a positive impact are not provided, or are not provided at an adequate level.

The actions that can mitigate salinity are also well documented. Some of these include revegetation, reducing water extractions from rivers, limiting the water used in consumptive land based activities, constructing drainage systems and pumping ground water.

From a public policy perspective, there are ways of both ensuring beneficial actions are undertaken, and of reducing the number of undesirable activities. However, given the site specific and often complex nature of water salinity problems, these changes in management often need to be targeted to specific parts of the landscape to achieve the greatest benefit and avoid exacerbating other resource degradation problems elsewhere. For example, poorly located revegetation actions to mitigate in-stream salinity may achieve a proportionally greater reduction in surface water runoff than in salt load, thereby contributing to an increase in salt concentrations in rivers.

Figure 2 Net benefits from reforestation



Another important consideration is that many of these mitigation activities can impose significant costs on agriculture and rural communities more generally. This would be the case, for example, where broad scale reforestation both reduces the surface water yield and increases the salinity concentrations in rivers in the near term to the detriment of downstream irrigators. Even in the long term, the proportional reduction in salt transported to rivers and streams may be less than the reduction in surface water runoff. That is, the cessation of tree clearing, regrowth management or broad scale tree planting such as through plantation forestry has the potential to capture water that would otherwise make its way into rivers and other watercourses. Not only could this make the problem worse, not better, it could impose significant costs on downstream communities as the volume of fresh water is reduced.

Reforestation may generate substantial net salinity mitigation benefits, however, if it is targeted to specific parts of the landscape to ensure it delivers a proportionally greater reduction in salt mobilisation than any reduction in surface water runoff. In this regard, two important landscape characteristics are soil types and the salinity of groundwater as illustrated in figure 2. Revegetation that reduces recharge to relatively fast responding aquifers would also be generally preferred. This suggests the level and type of actions undertaken should vary according to the biophysical characteristics of each region. A carefully designed scheme would ensure that trees were planted, or left uncleared, in areas where they would generate net salinity benefits.

The required level of information to ensure actions are correctly targeted in the landscape is generally significant and costly. As information and data improves, however, the scope for better outcomes increases. Current levels and accuracy of data are generally insufficient at the scale of individual properties although this is being facilitated as regions prioritise areas where the greatest benefits would be achieved.

When considering revegetation as an option to mitigate the effects of salinity, acknowledgment may need to be given to the opportunity costs of any water captured. One way this could be done is to draw revegetation activities within the scope and framework of the formal water market. This would require extending the coverage of water property rights to cover changes in land use. If this were done, large scale revegetation activities would be required in the same way as other consumptive uses to buy any water they need through the market.

Whether the extension of rights to include water use by forestry will actually lead to a more efficient outcome will depend on the transaction costs associated with extending those rights. Any apparent efficiency improvements in moving to such a system may be outweighed by the costs of negotiating, administrating, monitoring and enforcing the extension of rights, as well as by any costs in gathering information on the level of water use and marginal net benefits from water use by forestry (Goesch and Hanna, 2002).

Investing in carbon sequestration

Greenhouse policy aims to control the atmospheric concentration of greenhouse gases by reducing net emissions of these gases. The accounting framework set up to track these emissions includes net emissions resulting from changes in land use, principally where the land use change has occurred since 1990. This includes emissions following from the clearing of native vegetation and sequestration associated with revegetation. Thus there is a well defined framework for measuring the net carbon sequestration benefits of native vegetation management.

The major practical difficulty concerns the estimation of net emissions following land use change, especially as these are affected by past and future management actions. The existing procedures are designed to address the problem of estimating regional or national level emissions. There would be significant additional transaction costs involved with attempting to set up a market for emission abatement services that incorporates land use change.

Since emissions enter a single well mixed global pool of greenhouse gases the net benefit of abating one tonne of carbon dioxide equivalent emissions is independent of the spatial context and of how the abatement was achieved. Thus the marginal cost of abatement in other sectors of the Australian economy and in the world economy, such as the introduction of new technologies in power generation, defines a price for emission abatement. When the cost of abatement resulting from, for example, controlling the clearing of a particular area of vegetation, falls below this price then it would be efficient for this clearing to not take place. For the land clearing and revegetation cases the cost of abatement must include the full opportunity costs of the excluded activities.

The existence of a price based on the marginal cost of abatement in other sectors also means that including carbon sequestration services into regional planning processes is relatively straightforward. For a given price, the value of the net abatement service can simply be added to the commercial values of any land management activities.

General issues for public policy design

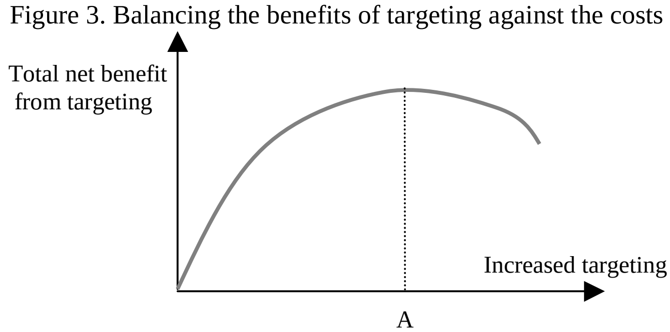
Minimising transaction and public costs

Other things being equal, transaction costs should be minimised to obtain the maximum net social gain from activities to improve resource outcomes. However, this is not an issue that can be considered in isolation from other policy design features. For example, there will need to be a tradeoff between keeping things simple, thereby minimising transaction costs, and tightly targeting activities.

In a recent case study of the Burdekin catchment in Queensland, the use of different management actions to address increased eutrophication and sedimentation of the Great Barrier Reef was examined (Love and Mues 2003). The study discussed the issues to be canvassed through public policy design and demonstrated that some investment options to address these problems have relatively low cost.

In general, improving the targeting of policies will involve greater transaction costs as additional effort in planning, administration and compliance monitoring will be needed. There will come a point when the additional benefits from a further improvement in targeting will be outweighed by the costs (point A in figure 3).

Figure 3. Balancing the benefits of targeting against the costs



Enduring benefits

Policies should also create appropriate ongoing incentives for improved natural resource management. Policies that promote only one-off investments in environmental assets, such as fencing off areas of remnant vegetation, but do not provide appropriate ongoing incentives to control pests or weeds in the fenced off area or to maintain the fencing, are less likely to deliver enduring benefits.

One option for policy makers is to use covenants to secure long term benefits. A covenant applies conditions to the title of the land to ensure it is used in a certain way. Where a public payment or subsidy is to be paid in return for a landholder taking actions to provide external benefits, a covenant could be used to stipulate and enforce long term commitments.

The advantage of covenants is that they can be tailored to different situations and can include contracts for how the land is to be managed. Responsibility for maintaining the land under the covenant could fall to the landholder or another party to the contract. Covenants may provide a high degree of security or certainty as they offer recourse to civil and criminal systems for enforcement and interpretation. They also have other advantages such as potentially being voluntary and generally acceptable to the community (ABARE 1997).

Generating additional benefits

Possibly the greatest challenge for policy makers is to design policies that encourage more of particular activities in a way that avoids making payments to landholders who already undertake those activities. That is, if external benefits were to be achieved at minimum cost, it would be highly desirable to avoid paying subsidies for actions that the landholder will probably do anyway.

In practice, this is very difficult. There are information asymmetry problems where the landholder is better placed than government to judge the profitability of the activity. Further, individual landholders have little incentive to share this information. Alternatively, collaborative regional processes may be able to more efficiently and effectively draw on the collective knowledge of landholders.

In situations like this, the only practical solution may be to strive to minimise payments for activities by using instruments designed to elicit this information, such as competitive tenders. A competitive tender is where landholders lodge their asking price (bids) to provide a specified bundle of external benefits. The appeal of tenders is that they have the potential to provide environmental benefits efficiently and at low cost (Stoneham, Chaudri, Ha and Strappazzon 2002; Heaney and Beare 2002). Competition among bidders for available public funds is designed to ensure that bids are lodged close to the opportunity cost of the landholder.

A critical factor determining the cost of the public investment will be the flexibility that each provides in discriminating between actions offering different levels of external benefits. Policies need to make efficient use of existing private and public knowledge, balance current investments in on-ground actions with research, and efficiently incorporate new information.

Conclusion

Improving the efficiency of native vegetation and biodiversity regulations will require governments to take a more outcome focused approach that more carefully targets support and regulatory constraints. If the application of regulations is to become more flexible there needs to be a clear statement of the overall objectives that the many pieces of regulation aim to achieve.

Since the majority of the impacts of changed land management, whether on rural communities or on environmental benefits, occur at a regional scale it is appropriate that efforts to improve the delivery of these benefits be delivered through processes based at the regional level. However, such processes need to be designed to deliver on a set of clearly specified objectives and need to be appropriately resourced. They also need to be supported with the provision of data and scientific and economic expertise.

The most promising approach to the targeted allocation of investment for the delivery of environmental benefits appears to be through the use of auctions. Not only do auctions address information asymmetries, joint assessment of bids can account for the spatial interdependence of benefits. Ideally these auctions would be embedded in collaborative regional processes so that the scope of the auctions is restricted to the activities likely to generate significant net benefits.

There are important uncertainties in our understanding of the full benefits and costs of changing the way natural resources are managed. There needs to be a framework for monitoring both the costs and benefits of actions so that future actions may be changed. This framework would also need to provide resource security for private investment.

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