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Valuing the biodiversity gains from protecting native plant communities from bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata* (DC.) T. Norl.) in New South Wales: application of the defensive expenditure method

by

J. A. Sinden[ab], Paul O. Downey[c], Susan M. Hester[ab], and Oscar Cacho[ab]

[a] School of Business Economics and Public Policy, University of New England, Armidale, NSW 2351
[b] CRC for Australian Weed Management
[c] Pest Management Unit, Parks and Wildlife Group, NSW Department of Environment and Climate Change, PO Box 1967, Hurstville, 1481, NSW

Presenting author: shester@une.edu.au

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Abstract

Valuation of the gains from protection of biodiversity is difficult because the services that provide the benefits do not normally pass through markets where prices can form. But the services sometimes pass through markets where consumers or producers behave in a market-oriented manner, and so the values implicit in this behaviour can be identified and derived. Estimates of the benefits of biodiversity protection are derived from the costs of protecting native plant communities from a major weed in Australia, by following this approach. In 1999, invasion of coastal areas of New South Wales by bitou bush (*Chrysanthemoides monilifera* subsp. *rotundata* (DC.) T. Norl.) was listed as a key process threatening native plants under the NSW *Threatened Species Conservation Act 1995*. In accordance with the Act, the Department of Environment and Climate Change prepared a Threat Abatement Plan (TAP) to reduce the impacts of bitou bush on biodiversity at each threatened site. The costs of protecting sites vary closely with the number of priority native species and communities at each site. Following standard economic assumptions about market transactions, these costs are interpreted to provide values the benefits of protecting extra species, communities, and sites.

Key words: Bitou bush, *Chrysanthemoides monilifera*, threat abatement plan, valuation of biodiversity, benefit-cost analysis, weed control, defensive-expenditure method.

1 Introduction

Despite at least 50 years of effort, values for the benefits from protection of biodiversity remain elusive, mainly because the environmental services that provide these gains do not pass through markets where prices form. But they often pass through other kinds of markets and, as Sinden and Worrell argued (1979), we can identify these markets to estimate the implied benefit values. The markets may represent consumers purchasing the services, producers supplying them, or even the market transaction itself. As the literature review will indicate, the methods to model consumer purchases have dominated valuation but the values they indicate sometimes lack consistency. Applications of methods to model producer behaviour are less

common because they require data on costs of protection - - but the methods would appear to be particularly relevant because they are based on actual expenditures.

A data set has become available for the costs of controlling a major weed at over 300 sites in coastal New South Wales, Australia, namely bitou bush [*Chrysanthemoides monilifera* subsp. *rotundata* (DC.) T. Norl.]. The data include expenditures to control the weed at the site, and the number of threatened native species, communities and populations at each site. These, and their associated data, are now used to value the benefits of biodiversity protection from the costs of doing so.

Bitou Bush arrived from South Africa around 1908, and has now spread to 80 per cent of coastal New South Wales. In 1999, this invasion was listed as a key threatening process under the NSW *Threatened Species Conservation Act 1995*. In accordance with the Act, the Department of Environment and Conservation (DEC) has prepared a Threat Abatement Plan (TAP) explicitly to reduce the impacts of bitou bush on biodiversity (DEC, 2006). The cost of implementing the TAP, at each of the more than 300 affected sites, is estimated to be $2.85m pa.

The goal of the paper is therefore to estimate values for the benefit of protection of native plant species, native plant communities and the threatened sites themselves. The objectives are to derive these benefits by (a) interpreting the actual total costs incurred to protect sites, (b) isolating the cost of protecting species and communities from the other components of total site cost, and (c) estimating the benefits at the market equilibrium quantity protected as well as at the mean quantity as in (b). The defensive-expenditure method provides a theoretical framework in which to analyse the costs and separate the effects of biodiversity from other factors that influence the expenditures.

2 The valuation of biodiversity

2.1 What is to be valued?
The object of value should be the increase in the quantity of environmental services from protection of the native plant communities from the weed invasion. This is the change in actual quantity of services themselves, rather than the change in the resources that provide the services. Further, the values for the increases in environmental services should be the values that would arise in a competitive market for those services.

The extreme impact of weeds on the natural environment is the extinction of species and associated loss of biodiversity. But biodiversity is not usually traded in a competitive market so there are normally no prices to form the basis of the economic values. In response, methods have been devised to apply market concepts to value biodiversity and the protection of species.

There are three kinds of method to estimate unpriced values, namely those based on (i) demand and willingness to pay, (ii) supply and costs of protection, and (iii) market prices. Market prices rarely exist for environmental services or their associated goods and services, so this review concentrates on demand and supply-based methods.

2.2 Demand and willingness to pay

Contingent valuation and choice modelling use survey questions to elicit household willingness to pay to protect a species, a community, a habitat or an ecosystem. The basic procedure is to ask the survey question, how much are you willing to pay for thing X? For example, Bennett (1982) asked 544 people in Canberra how much they were willing to pay, as a lump sum, to protect the natural ecosystem in the Nadgee Nature Reserve on the south coast of NSW. The 95% confidence interval for their willingness to pay ranged from $20 to $33 for the 60% of the sample who offered bids. Bennett concluded that the protection of this biodiversity is likely to be worth $20 per adult resident of Canberra.

Each Victorian household was willing to pay $118 per year to preserve all endangered species in Australia, according to a survey by Jakobsson and Dragun (1996). This payment would give a total value for all Australian households of $850m per year, or $271m for the 2.3m households in NSW. Lockwood and Carberry (1998) estimated a lump-sum willingness to pay of $1.69 per household to protect an endangered species. This is equivalent to $3.89m per species for all 2.3m households in New South Wales or $195,400 per year. Kennedy and Jakobsson (1993) estimated that individual Victorians would pay $40 per head per year to protect the Leadbeater's Possum in Victoria, giving a payment of $194m per year for all Victorians. McLeod (2004) reviewed the values placed on biodiversity protection in this kind of study and found that Australian households said they were willing to pay between $11 and $118 a year to protect an endangered species, giving a state-wide range of $25.3m to $217.4m.

Contingent valuation is, basically, valuation of a single good though a single question. Choice modelling requires respondents to choose between bundles of several goods and where the cost of the bundle is given. The choices are analysed to infer the value of each good or service in the bundle. Rolfe et. al. (2000) apply choice modelling to estimate the willingness to pay for protecting native woodland in the Queensland Upland Desert region. The representative respondent in the survey proved willing to pay $11.39 per year to maintain an endangered species. This is equivalent to $26.2 m for a population of 2.3m households, assuming one respondent per household.

Van Bueren and Bennett (2004) use choice modelling to assess the size of benefits of improvements to waterway health, rural aesthetics, and species protection at a national level. They argue that the implicit prices from the modelling provide the basis for assessing the size of benefits associated with such improvements. Their benefit for protection of 70 (unspecified) endangered native species (plants or animals) was valued at $46.90 per household each year for 20 years, which becomes an annuity of $960,000 per species for 2.3m households.

The Priority Evaluator Technique (Hoinville and Berthoud, 1970) and the Budget Allocation Method (Sinden and Worrell, 1979) were the methodological forerunners of Choice Modelling. Both also require respondents to choose between bundles of goods although both allow the price paid for the bundle to vary whereas in Choice Modelling the budget is fixed at one of a range of amounts. O'Hanlon and Sinden (1978) estimated the existence value for a particular national park to be $6.25 out of a recreation budget of $17.50 giving a total value for protecting all species in the park for all 41,000 family visits to the park of $255, 300 per year.

2.3 Supply and costs

The cost methods interpret expenditures on the supply of environmental services, to derive the benefit values implicit in the expenditures. There are many different kinds of expenditure involved in the provision of these services, including the costs of directly controlling a weed, costs that are avoided in the future when the weed is controlled today, and expenditures to defend the existing biodiversity resources. Interpretations of these costs, to provide benefit values, have included the following.

- Expenditures that are actually undertaken can give minimum estimates of the value of the associated benefit.
- Future costs, that are actually avoided, are a benefit.
- If vegetation is depleted, the associated loss of soil and water conservation may lead to loss of agricultural output which is a cost.
- Managers undertake control expenditures to maintain a level of environmental service, and these defensive expenditure method estimates the values of these gains from the expenditures themselves.

All these methods are based on actual costs and so should provide reliable values of their particular values, but all are likely to give minimum estimates of benefit.

Governments undertake expenditures to protect environments in various ways, although their motivations may not always match those of producers in the standard market model. Their annual support of government agencies is complemented by one-off payments for special purposes. For example, in 2002, the Queensland Government agreed to contribute $200m to protect threatened but not endangered native vegetation (Morton et al, 2002). This action would, apparently, save 5280 species from extinction, so willingness to pay per species is a lump sum of $38,000 or an annual payment of $1900.

Managers will undertake expenditures on control programmes to defend the existing environment when they expect an environmental gain. The defensive expenditure method estimates the values of these gains from the expenditures themselves - - the expenditure on the marginal unit is a minimum measure of the benefit of that unit. For example, Sinden and Griffith (2007) estimated the value of the environmental gains from protecting threatened sites and threatened species from expenditures to control of 35 weeds across Australia. They reported that the annual expenditure to manage each of 35 weeds in Australia increases by $64,800 for each extra native species and $5,800 for each extra site protected.

The opportunity cost of protection is, of course, the loss in agricultural income to protect the native vegetation. While it is a different kind of cost, governments impose regulations on landholders to protect native vegetation, and this reduces their options for development and so reduces land value. In an area of northwestern New South Wales, Sinden (2004) combined (a) functions of change in land value due to changes in vegetation, with (b) functions of the loss of native species due to changes in vegetation levels. At current levels of vegetation, the present polices to maintain 30 per cent of each property in native vegetation would cost at least $148.5m over all 1.227m ha of farmland - - as a loss in land value. This is equivalent to a cost of $15.7m for each species protected across this farmland. Simulations to allow for

uncertainty in species numbers, and in the coefficients in the model, showed that there was a 0% chance the cost is below $15.7m., a 25% chance it is below $29.1m, and a 50% chance it is below $37.4m. The annuity equivalent of $15.7m is $314,000 which applies across the whole area.

2.4 An overview
Values for environmental services should be the values that would arise in a competitive market for those services. So the method to value them should model the structural characteristics of the market which include:
- many buyers, all facing budget constraints,
- many producers all facing input constraints, and
- exchanges between buyers and sellers.

All the methods attempt to model the operation of the competitive market and all follow the supply/demand model of Figure 1. But the willingness-to-pay survey methods have no effective budget constraints, whereas the methods based on actual cost methods do. So, a priori, the methods based on the actual costs of control appear to offer reliable ways to value biodiversity (Sinden 1994, and Sinden and Thampapillai 1999). The defensive expenditure method appears to be a promising way to analyse such cost information.

A review of the limited set of empirical values for biodiversity gains obtained so far indicate that:
- values from willingness to pay methods exceed the values from cost methods,
- values from willingness to pay methods vary from survey to survey and
- values from willingness to pay methods are high.

But methods based on demand and willingness to pay estimate points on a demand curve, such as M in Figure 1, whereas supply and cost methods estimate points on the supply curve such as M. This may be another reason why the cost methods appear to give lower values then the willingness-to-pay methods for the value of biodiversity gains.

3 The defensive expenditure method

Individual households or firms often act to maintain the existing level of utility or profit. They might, for example, soundproof houses against noise or change the source of inputs to maintain resource quality. The benefit of such actions must exceed the cost, otherwise they would not be undertaken. So, in principle, we can derive the benefits from these defensive expenditures, hence the intuitive appeal of the method.

3.1 The theoretical model
The theory to apply these beliefs can be presented first from the standpoint of consumer who seeks to maximise utility (Shortle and Abler, 2001).
$$U = U(e, q) \qquad (1)$$
with respect to the income constraint
$$Y = D(e) + p.q \qquad (2)$$
where U is utility, e is environmental quality, q is the quantity of all non-environmental goods and services that are consumed, p is the price of those goods and

services, D( ) is the defensive expenditure function that defines the necessary costs to "consume" the environmental quality, and Y is income.

The benefit from defending a given environmental quality against an adverse change is equal to the extra income (EV) that would be required to restore the consumer's utility back to its original level, after the adverse change has occurred. In estimating this measure of Hicks equivalent variation, the utility before the change must equal the utility after the change:

$$U(Y_0, e_0) = U(Y_1, e_1) \qquad (3)$$

where e is the level of environmental quality before ($_0$) the change and after ($_1$) the change. The ideal measure of defensive expenditures in terms of consumption (DE_c) is therefore the change in income that restores the original level of utility. So from equation (3) we have:

$$DE_c = (Y_1 - Y_0) = EV \qquad (4)$$

This concept of the benefit is hard to measure when the demand for the environmental quality is hard to estimate, and when the observable data do not capture any change in demand that follows the change in environmental quality.

A minimum bound to the benefit can be estimated from the costs necessary to maintain the level of environmental quality rather than from the extra income necessary to maintain the level of utility. This minimum (DE_{cm}) is the expenditure necessary to maintain the initial level of environmental quality (e_0) before and after the change. For the consuming household and following Shortle and Abler (2001):

$$DE_{cm} = D(q_o, e_0) - D(q_o, e_1) \qquad (5)$$

where e_0 is the quality of the environment that is to be maintained.

Equation (5) can also be derived through the production measures to be undertaken to guard against the loss of a natural site from a weed invasion. The defensive expenditure (DE_p) is now the cost of the extra inputs needed to maintain the initial environmental quality of the natural site after the potential invasion.

$$DE_p = D(e_0, w_0) - D(e_0, w_1) \qquad (6)$$

where e_o is the initial level of environmental quality, the weed invasion w. This approach requires only information on the defensive expenditure function D(e, w), and the management actions before and after knowledge of the invasion.

3.2 Application to valuation

The concept of defensive expenditures has been widely used to describe the broad protective purpose of many cost outlays (McMillan, 2004, Escofet and Brava-Pena, 2007). In this use, the concept helps to provide and organise information for choices between policies. For example, Rogers, Sinden and De Lacy (1997) estimate the loss in income from irrigated crops when water is diverted to maintain riparian forest in its existing state. In the present work however, we apply the notion more specifically to value particular gains in biodiversity when the native plant communities are protected.

The method assumes that producers and consumers are rational, and so no expenditures are undertaken unless benefits exceed costs and all expenditures are undertaken when benefits exceed costs. In the context of protecting sites from the spread of bitou bush, the market moves to Q_e where demand and supply are in equilibrium (Figure 1). This scenario implies we are we defending the existing levels of environmental services at the existing site rather than defending the flow of environmental services by increasing the services at other sites.

Consider the protection of many similar reserves of native plant species, where the costs to control the weed vary. The costs include access construction, spraying, hand removal, planning and monitoring. The method assumes that decision makers know how a weed invasion affects the flow of environmental services, and that defensive actions do not change the flow of environmental services from the pre-invasion level.

Harrington and Portnoy (1987) developed a household production model to assess health pollution issues. The model included expenditures on goods explicitly purchased to defend the existing levels of air quality and the environmental service flows that rest on that quality. They point out that inclusion of defensive expenditures may overstate willingness to pay for pollution control if the expenditures enhance health quality and so the new equilibrium involves a higher quantity of the service because the quality increases.

3.3 Application to the problem

The defensive expenditure method is now applied to value the economic benefits of biodiversity gains from management of the weed. The basic procedure is to derive a cost function to model the supply curve AS in Figure 1, and so estimate DE_p in Equation (6). The method is likely to provide minimum values as argued above, and also because a greater budget allocation might lead to higher expenditures on control (Eiswerth, Shaw and Yen, 2005)

4 Data collection

Data on costs, site characteristics and quantities of biodiversity protected, were available for over 300 sites identified in the bitou bush TAP. But data were not available for all the variables at each site. For example, cost information was sometimes available only for a group of sites. When such data could not be reallocated per site, all sites in the group had to be removed from the sample. This process left 90 sites with a full set of data. This information is now described and the variables are summarised in Table 1, with their mean values over the 90 sites.

4.1 Measures of cost

For each site, annual cash expenditure, external grants, and in-kind contributions were available for 2005-2006. They represent the total resources necessary to protect each individual site and the biodiversity they contain. So, total economic cost was defined as follows:

Total Economic Cost (COST) = cash expenditures + external grants + in-kind costs   (7)

Cash expenditures are recurrent funds spent on wages and operating costs. The external grants are incomes from other state and Commonwealth agencies such as the National Heritage Trust. The in-kind costs include volunteer labour costed as (number of volunteers * a wage rate), staff time of the government agencies, and on-costs to manage the volunteers.

The total economic cost of implementing the TAP in 2005-06 was $2,845,500. This included expenditure by the DECC, Department of Lands, the five coastal Catchment Management Authorities (CMA), Lord Howe Island Board, numerous coastal councils and the University of Wollongong. The on-ground cost in 2005-06 was $2,489,000 which included the costs of control activities, site planning, monitoring of priority sites, surveys, training volunteers, and direct co-ordination.

4.2 Measures of biodiversity and weed threat

The data included the following measures of biodiversity:
- the total number of high priority, medium priority and low priority species (NOSPEC), and
- the total number of high priority, medium priority and low priority ecological communities, (NOCOM).

The data also include the number of priority endangered populations, so the total number of ecological entities (TNEE) is the sum of all these species, communities and populations per site.

The TAP contained data on the degree of risk from bitou bush to each ecological entity at each site. This information was based on the potential impact of bitou bush and associated weeds, and the proximity of the site to existing areas of weed invasion. The information was scored on a five-point scale (1 = low threat to 5 = high threat) for each entity at each site. So the average score per entity was calculated at each site and taken as a measure of the potential impact of the weed (IMPACT).

The potency of the threat at a site depends on both the willingness and ability of managers to protect the site. In turn, willingness depends fundamentally on the number of native species, communities, and populations at the site, and ability depends on the difficulty of control of bitou bush including the difficulties of access, and the level of the invasion so far. All of these factors are captured as the number of ecological entities protected per worker (NEPPW) per year per site. We assumed that the cost per worker, including on-costs, was $90,000 per year and so the number of workers was calculated as ((cash +external grants)/90,000). The variable:

$$NEPPW = TNEE/\text{Number of workers} \qquad (8)$$

measures the number of entities protected per employee per year at each site. Increases in the value of NEPPW indicate increases in the effectiveness of control measures in protecting biodiversity. So the cost per site should decrease with increases in this variable.

4.3 Tenure, location and area

There are 38 different kinds of land ownership across sites, and management may vary with tenure. So the three main tenures sites were identified and coded as TNP = 1 if the site was administered by the National Parks and Wildlife Service and 0 if otherwise, TLG = 1 if Local Government and 0 if otherwise, and TDL = 1 if Department of Lands and 0 if otherwise.

The 90 sites were located along 1100 kilometres of coast and covered five Catchment Management Authority areas. Environmental conditions, the nature of the weed invasion and management responses, are likely to vary over this length of coastline and by CMA region so dummy variables were introduced for the catchments with more than ten sites. The location variable LNR was coded as 1 = Northern Rivers Catchment and 0 = otherwise, LHC was coded as 1 = Hunter Central Rivers Catchment and 0 = otherwise, and LSR was coded as 1 = Southern Rivers Catchment and 0 = otherwise.

5 Analysis

The data are now analysed to identify:
- the total cost per site and per ecological entity protected,
- the cost specifically due to biodiversity protection at the "mean", and
- the cost specifically due to biodiversity protection at market equilibrium.

At each of these three stages, the benefits are interpreted. Thus we estimate the total benefit per site from all "sources", the benefits due to the "mean" quantity of biodiversity protection, and the benefit due to the 'equilibrium" quantity.

5.1 The total benefit per site and per ecological entity protected

5.1.1 Benefits per site
How much has been spent to defend the sites against bitou bush? Are some sites relatively costly to protect while others are relatively cheap? And what do these expenditures mean for the valuation of the benefits from biodiversity protection? The 90 sites were ranked in order of increasing cost which is expressed as the present values of expenditures over the five years of the Plan (with the annuity equivalents in parentheses). The following results were observed.
- The minimum cost per site was $2,165 ($110).
- The mean cost per site was $121,348 ($6,070).
- The maximum cost per site was $1,068,586 ($53,430).

The site with the highest cost was SR 20, Bherwerre Peninsula in Booderee NP, which contained three high and medium priority species and seven different ecological communities. Since this amount has been spent to save these ten ecological entities, protection must be worth at least this amount and so the benefits of protection appear to be worth at least $53,430 per year for this kind of site. This particular site has what is believed to be the largest infestation of the weed on the south coast of New South Wales, and is at the edge of the Southern Containment Zone that is kept clear of the weed to prevent its spread.

5.1.2 Benefits per entity protected
A more relevant indicator of the cost of biodiversity protection might be the cost per ecological entity protected, so the 90 sites were now ranked on this criterion (Figure

2). The costs, again as present values over five years (with annuity equivalents) were as follows.

- The minium cost per entity was $100 ($5).
- The mean cost per entity was $8,121 ($410).
- The maximum cost per entity was $50,000 ($2,500)

The figure shows that the cost per entity increases only slowly up to about 70 of the sites, but then increases steeply for the last 20 sites.

The three sites with the highest costs per entity are shown in Table 2. Sites NR45 and NR55 each contains one priority ecological entity. These are *Plectranthus cremmus* and Littoral Rainforest respectively, both of which occur at other sites. Site SM3 contains two priority ecological communities, namely Grassy Headlands and Coastal Banksia Woodland. Both of these also occur at other sites. If the biodiversity is protected at other sites are there additional reasons for protecting these high-cost sites? For example, do the sites form part of the National Containment Zones to stop spread of the weed or are they in designated buffer zones around other, more crucial areas? But they have been protected, so we conclude that the entities in NR45, NR55 and SM3 are worth up to $2500 per year to the community.

5.2 Benefits from biodiversity protection at the "mean" quantity of biodiversity

5.2.1 The general model
The values implied from the observation of expenditures assume that all changes in cost are due to changes in biodiversity alone. For example, they assume that the entire $2,500 per entity per year from SM3 accrues to biodiversity alone. These values are good starting points for policy choices, but they will exceed the true benefits for biodiversity protection alone if other factors affect the cost. So we now separate the effects of biodiversity from the potential impacts of weed threat, the difficulties of control, differences in land tenure, and differences in location.

The analytical task is to identify the way the cost of protection varies with these different characteristics of the site, including the biodiversity that is protected. The following general function, which models the supply function AS in Figure 1, was therefore specified.

Cost = f (number of priority species protected, number of priority communities protected, potential impact of the weed invasion, difficulty of weed control, land tenure, and location)   (9)

The following data are available to estimate equation (9). Following the symbols of Section 4 which are summarised in Table 2:

COST = f (NOSPEC, NOCOM, IMPACT, NEPPW, TNP, TLG, TDL, LNR, LHC and LSR)   (10)

When estimated, this model provides the information to value the environmental benefits from managing weeds to protect biodiversity.

5.2.2 The method of estimation

11

The relationship of COST to the explanatory factors in Equation (10) is likely to be curvilinear, as postulated in Figure 1. The costs per entity of Figure 2 exhibit a geometrically-increasing trend, as did the costs per site when they were graphed. So the dependant variable (COST) was transformed to logarithms to the base 10, but all the explanatory variables (NOSPEC to LSR) were left in their arithmetic form.

The estimated functions are shown in Table 3. To begin with all the variables were included in an ordinary least squares regression to give Equation (11) and then particular variables were excluded on the basis of diagnostic tests to give Equations (12) and (13). In particular, LNR and LHC were correlated at 0.616 per cent in Equation (11). Although the coefficient of determination of Equation (11) was higher at 0.757 and these are both dummy variables, LHC was excluded to give Equation (12) to avoid any possibility of multi-collinearity. Variables TNP, TLG and TDL are all included in Equation (11). But TNP was related at 0.463 per cent to TLG, so TLG was excluded from Equation (12) for similar reasons. The highest correlation of the remaining explanatory variables in equation (12) was 0.493 for LNR and LSR so LNR was now excluded to give Equation (13) because it had the highest correlations with NOSPEC and NOCOM (0.324 as opposed to 0.156, and 0.470 as opposed to 0.342).

The coefficients per variable vary little across the three equations of Table 3. Equation (12) has slightly higher adjusted R squared statistics and slightly higher, or equal, t-statistics on all its variables so this equation is preferred. The highest remaining correlation in Equation (12) was 0.493 between LNR and LSR.

5.2.3 Interpretation of the estimated equations
The estimated equations provide information to explore the way bitou bush affects management and to value the biodiversity gains from management. Equation (12) indicates that more is spent on weed control when:
- there are more priority native species at a site,
- there are more priority communities at a site,
- the potential impact of the weed is greater, and.
- the site is administered by local government.

Less is spent per site when:
- the number of entities protected per employee-year increases,
- the site is administered by the National Parks and Wildlife Service, and
- even less is spent when the site is administered by the Department of Lands.
The National Parks and Wildlife Service has had management and expenditure programmes at their sites over many years, and is likely to be more efficient now in control than other agencies. So the requirement for expenditure at their sites in the TAP is likely to be less *ceteris paribus*, hence the negative sign on TNP.

All the dummy variables for location (LNR, LHC, LSR) have negative signs. But only LNR and LSR significant and LSR has the highest negative coefficient. So the costs per site decrease below the mean for the Northern Rivers Catchment Management Authority but they decrease even more in the Southern Rivers CMA

5.2.4 The benefits from biodiversity protection

The benefits from biodiversity protection may now be valued as follows. The extra cost that is expended to protect an extra ecological entity at the mean is a measure of this benefit. The extra cost is represented by M at Q_m in Figure 1 and can be derived as follows.

In the semi-logarithmic models of Table 3, the dependent variable has been transformed into logarithms while the explanatory variables retain their arithmetic form. So:

$$\text{the extra cost of the first extra entity} = \partial y/\partial x = \beta\,(Y) = \beta\,(COST) \qquad (14)$$

where β is the coefficient on the explanatory variable and COST is the arithmetic value of total cost per site calculated at the logarithmic mean. The mean total cost per site in logarithms is 4.849, the arithmetic value of this mean is $70,650, and the coefficient for the number of species is 0.05949. The extra cost is therefore $4203 as a present value, or $210 as an annuity. In a similar way, the extra cost of a community is a present value of $5377 or an annuity of $270.

Following the defensive expenditure approach and the assumptions of market transactions, these extra costs are the estimates of the minimum benefits of the gains in biodiversity protection. The annual benefit of protecting a species is therefore $210 and the annual benefit of protecting a community is $270 per year. These two benefits are listed in Table 4. The values per site were calculated as (mean number of species per site x this value per species) plus (mean number of communities per site x this value per community). The value per extra site, calculated in this way, was $1020.

5.3 Benefits from biodiversity protection at the "equilibrium" quantity of biodiversity

The values of biodiversity gains for use in a benefit-cost analysis should be the values that would form in a competitive market in equilibrium, if such a market existed. The values should therefore be derived by modelling the equilibrium transaction at *E* in Figure 1. In contrast, the values of the previous section have necessarily been derived for the "mean" number of species or communities protected per site. This would be represented at say *B* for quantity Q_m in the figure. We would expect the "equilibrium" values at *E* for quantity Q_e to be higher, even much higher, than the values for the mean quantity Q_m.

Management at all the sites in the bitou TAP has been funded so the equilibrium values can be approximated from the highest marginal cost per species or per community that has been expended. This, of course, assumes that the budgets were sufficient to allow managers to undertake all control where the perceived benefit exceeded the cost, so the last (or most costly) exchange becomes the marginal transaction for the unit at Q_e. We can therefore derive supply curves for the extra species and communities protected over the range from the mean number (2.54 species or 1.80 communities) to the maximum number (12 species or 9 communities) protected. The procedure, to apply Equation (14) and derive these marginal costs, is now illustrated for the protection of species.

Instead of just β and the mean value of COST as in Equation (14), we now have:

the extra cost of the nth extra entity $= \beta \, (COST + n *\beta)$ (15)

where n varies from 1 to 12 for the extra species and 1 to 9 for the extra communities, β is again the coefficient on the explanatory variable (NOSPEC or NOCOM), and COST is the arithmetic value of the total cost per site in logarithms at each number of extra entities. The term in parentheses is calculated first as a logarithmic sum and then converted to its arithmetic equivalent before multiplying by the coefficient. If we are calculating the values at the means as in section 5.2.4, Equation (15) reduces to Equation (14) because *n*, the number of extra entities, is nil.

Equation (15) was applied with increasing levels of *n* for species and communities separately to give the two marginal cost curves of Figure 3. The "equilibrium" values were $1020 per species protected (for the 12th species), $1300 per community protected (for the 9th community protected). The value per site, as mean number of entities times these maximum values, was $5020 (Table 4).

6 Discussion

6.1 The benefit values

The benefits of Table 4 represent the value of the extra unit of biodiversity at the mid-point for the mean number protected (Row 1) following Q_m in Figure 1, and at a proxy for the equilibrium number protected (Row 2) following Q_e. The value of the equivalent biodiversity gains per site is $1020 per year at the mean quantity of species and communities, and $5020 at the equilibrium quantities. This particular mean site value is comparable to the value of $5860 from Sinden and Griffith (2007), which was derived from an analysis of management costs to protect against similar weed invasions in similar situations.

These are of course annual values per site. Over 300 sites in the TAP, the benefit value is $306,000 per year (300 x $1020) at the mean quantities, or $6.12m as a present value. At the "equilibrium" quantities, we have $1.506 m per year or $30.12m as a present value.

In 2001, the Centre for International Economics undertook a benefit-cost analysis of bio-control of bitou bush. They estimated the willingness to pay per household for the extra biodiversity on the same kind of sites to be between $1 and $2 per year for each household living within 50 kms of the affected areas. We now assume that all 2.3m households in NSW are affected by the actions of the NSW National Parks and Wildlife Service. With a 3 per cent rate of inflation from 2001 to 2006 (the date of the TAP), the CIE (2001) value of the biodiversity protected lies between $2.667m ($1 x 2.3m x 1.16 for inflation) and $5.332m ($2 x 2.3m x 1.16). This willingness-to-pay estimate lies close to, but rather above the preferred defensive–expenditure estimate of $1.506m per year - - as we would expect.

Values derived through actual expenditures are necessarily minimum estimates of the benefit. But they are defensible estimates because they are based on actual

expenditures. They are also economic values, which attempt to value the equilibrium price that would emerge in a competitive market if one existed. They are, obviously, not ecological or biological values in any sense.

6.2 The valuation process

In the preliminary phase of model building, the individual number of high, medium and low priority species, as well as the total number of high, medium and low priority species in aggregate, were each regressed against COST. But only the total proved significant. The same result occurred for the numbers of communities, so the analysis was based on the total number of priority species at each site (NOSPEC) and the total number of priority communities at each (NOCOM). The analysis therefore implicitly assumes that all species are of equal economic worth and all communities are of equal economic worth.

The defensive expenditure method assumes that decision makers are rational and so undertake protection activities when the benefits exceed costs, and have sufficient budget to undertake all activities where benefits exceed costs. Then the estimated "equilibrium values in Table 4 would be the "true" market values of protection of biodiversity. In reality of course, budgets are limited and other factors such as regional rather than state-wide targets, motivate the choice of activities. But in the TAP, choice of activity rested closely on the protection of biodiversity.

The analysis has provided values for individual species, communities, and sites. But are these values in their biologically-appropriate order and relative magnitude? Under the NSW Threatened Species Conservation Act, native biodiversity can be listed as individual species, a specific population of a species, or an ecological community. A population is "smaller" than a species while a community comprises a group of species in its biophysical habitat. The number of populations proved to be unrelated to costs but the value for community values exceeded the value for species - - as perhaps might be expected.

We have taken the maximum number of species per site (twelve) over the set of sites, and the maximum number of communities per site (nine) over the set of sites, as the proxy for the "equilibrium" number of each (*Qe*). The true equilibrium numbers will exceed these levels because existing expenditures have been taken to protect existing sites but they may also have been taken to protect other sites with more species and communities. Thus, further work should be undertaken to reduce the uncertainty surrounding *Qe*.

6.3 The policy implications

How might these results assist with policy choices?
- The biodiversity values obtained here, together with the methods of analysis developed in Sinden et al (2008) and Hester et al (2006), could be applied to assess the economic worth of other threat abatement plans that concern similar kinds of weeds and vegetation.

- The processes for selecting a set of sites for future threat abatement plans might now be reconsidered, to review the inclusion of high-cost sites where all the species and communities occur elsewhere.

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Table 1 Variables used in the analysis of the costs for 90 sites

Symbol	Description	Mean
COST	Total economic cost $ per year for five years	28,025
	Measures of biodiversity	
NOSPEC	Total no of high, medium, and low priority species	2.54
NOCOM	Total no of high, medium and low priority communities	1.80
NOPOP	Total no of high priority populations	0.04
TNEE	Total number of ecological entities*	4.39
	Measures of weed threat	
IMPACT	Potential weed impact (mean score, 1 low to 5 high)	3.40
NEPPW	Number of entities protected per worker per year	98.72
	Land tenure	
TNP	Tenure =1 if administered by NPWS, 0 otherwise	0.48
TLG	Tenure =1 if local government, 0 otherwise	0.19
TDL	Tenure =1 if Department of Lands, 0 otherwise	0.07
	*Location**	
LNR	Location =1 if Northern Rivers CMA, 0 otherwise	0.51
LHC	Location =1 if Hunter Central CMA, 0 otherwise	0.27
LSR	Location =1 if Southern Rivers CMA, 0 otherwise	0.19

* All biodiversity identified here
** CMA indicates Catchment Management Authority. Variables for the Sydney Metropolitan and Hawkesbury Nepean CMA's are excluded, because there were too few sites in each.

Table 2 The cost per entity for the three most costly sites

Site	Number of entities	Costs per entity ($)	
		Present value	Annuity equivalent
NR45	1	30,000	1500
NR55	1	41,730	2100
SM3	2	50,000	2500

Table 3 Equations* to explain variations in cost: 90 sites

Explanatory variables	Equation		
	11	12	13
Management factors			
NOSPEC	0.06182 (3.8)	0.05949 (3.8)	0.05156 (3.4)
NOCOM	0.07831 (3.2)	0.07611 (3.1)	0.09151 (3.9)
IMPACT	0.055989 (1.8)	0.06000 (2.0)	0.05550 (1.8)
NEPPW	-0.00145 (7.8)	-0.0014 (8.0)	-0.0014 (7.7)
Land tenure of site			
TNP	-0.13837 (1.3)	-0.1672 (1.9)	-0.1643 (1.9)
TLG	0.06691 (0.6)		
TDL	-0.33469 (1.9)	-0.3909 (2.4)	-0.3119 (2.0)
Location of site			
LNR	-0.30940 (1.5)	-0.1744 (1.8)	
LHC	-0.15741 (0.7)		
LSR	-0.40301 (1.8)	-0.2814 (2.3)	-0.1975 (1.7)
Intercept	4.8553	4.7479	4.6411
R	0.757	0.755	0.743
R^2	0.574	0.570	0.553
Adj R^2	0.520	0.527	0.514
N	90	90	90

* These equations are all models of the function:
COST = f (NOSPEC, NOCOM, IMPACT, NEPPW, TNP, TLG, TDL, LNR, LHC, and LSR), together with their R and R squared and adjusted R squared statistics. The t-statistics are in parentheses.

Table 4 Values for biodiversity gains from analysis of 90 sites

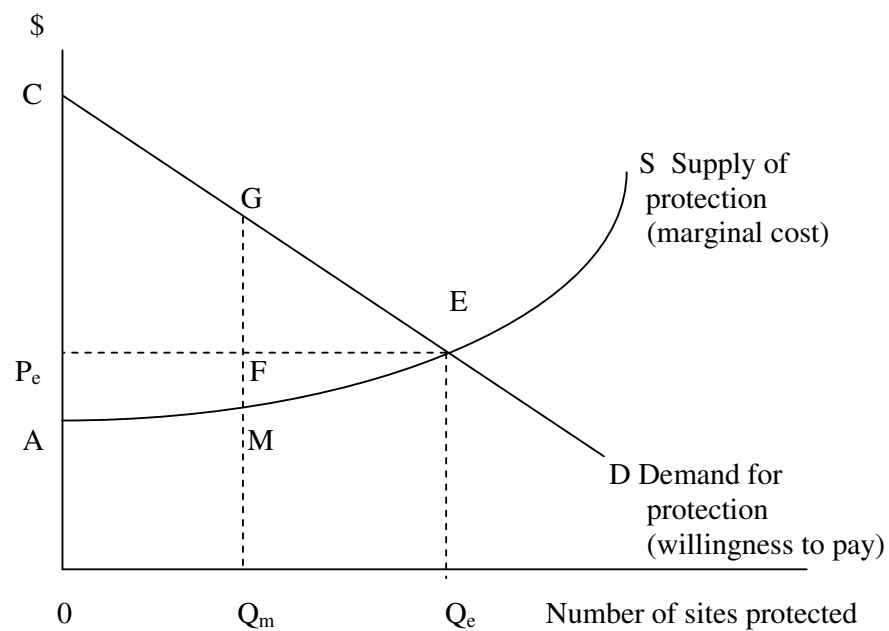| Situation	Values ($ annuities)		
	Per species	Per community	Per site
1 At the mean	210	270	1020
2 At "Equilibrium"	1050	1300	5020

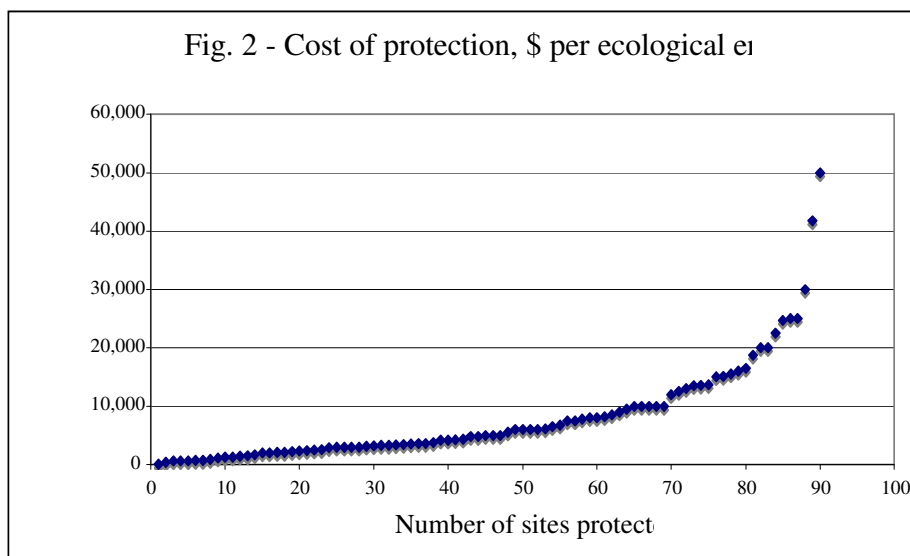**Figure 1 Application of the defensive expenditure method**

Fig 3   Marginal costs of protecting native plants: series 1 is individual species and series 2 is individual communities