Valuing Marginal Changes in Wildlife Populations and Habitats

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Abstract:

Wildlife conservation represents one of the most difficult challenges to economic valuation and, given the rate of biodiversity loss worldwide, is also greatly in need of the transparent argument for protection which such valuations can provide. This paper presents a simple method for estimating the marginal benefit of an increase in inhabited wildlife habitat.

This is particularly relevant to agriculture where, to date, the question of efficiency in achieving environmental benefits net of the costs has largely been ignored, in part due to the lack of values for marginal benefits. Often, costly land use changes must be undertaken to achieve these goals and eventually, as the quantity of available habitat increases for a given species, the increased resilience of the population gained from another unit of habitat will fall. At some point it will be more fruitful to begin providing habitat for other species in the area, or else to move to land where the species is less well protected. Similarly, at some stage, loss of productive ground may become unjustifiable in terms of the increase in food-miles or pressure to damage overseas (often more biologically diverse and unique) habitats. Currently, no suitable measure is available to aid such calculations.

Equally pressing economic problems are faced by those providing protection against invasive species or wildlife-borne diseases. For instance if culling a given species reduces the probability of disease transmission to livestock, an economic analysis would suggest that culling effort should be set where total cost of culling plus the new risk of transmission multiplied by the cost of infection is minimised. This would minimise the expected costs for all, but is currently unachievable.

Another reason to favour marginal values is that the stated preference techniques used are not cheap. Although conservation work can be co-ordinated on large scales it is usually implemented and funded on a much smaller scale. In the UK for example the 382 species with action plans being protected by 185 local partnerships, which together are made up of over 1500 organisations. This could represent up to 70,670 valuations. This method allows large scale valuations to be used in local CBAs.

Our method still requires stated preference valuations for large scale ecological projects, but the value for a unit of habitat derived is weighted by the probability that wildlife will persist in that unit into the foreseeable future. As such the function which derives the value has diminishing returns to scale, and also includes the increased support which the unit provides to the surrounding habitats. The public may be able to express a willingness to pay for a scheme which professes to “prevent any further loss in the species abundance or distribution” but cannot be expected to value every square meter of habitat. The unit value presented does not, therefore, represent the increase in social welfare that the unit of habitat provides. Nor can it guarantee that any one project will yield a consumer surplus should its costs be lower than the benefits derived from this value. However it will ensure that should every project acting towards the overall goal cost less than the benefits derived then the sum total would pass a cost benefit analysis.

This paper reviews previous attempts to value marginal changes for wildlife. It then describes some of the current challenges facing agri-environmental schemes and the need for marginal values for their benefits. The paper then explains our novel method in general terms before providing a case study of its implementation for the water vole (*Arvicola terrestris*) in the United Kingdom.

Introduction:

The environmental sub-discipline of economics sprang from theories of market failure and attempts to realign instances of market failure. That is to say that while nobody need be, “in charge of the supply of bread to London” (Seabright 2004) the same cannot be said of acid rain supplied in Scandinavia or the conservation of lions in Kenya. These examples of market failure require intervention but since the costs of intervention can be substantial it would be preferable if the intervention were the most efficient possible.

There are often cases in which a single environmentally enhancing action might be considered (for example the banning all Chlorofluorocarbons) and the benefits will be found to outweigh the costs, in the same way that providing the people of London 4 million loaves of bread at 90 pence each might. Ideally, however, bread would be produced at a quantity which maximised consumer pl

us producer surplus and a pollutant would be tolerated at a point at which the full cost of abating the next unit of pollutant cannot be outweighed by the benefits of doing so. Ideally environmental policy should aim to maximise net benefits (Baumol and Oates 1988). Ideally economists should be concerned with the analysis of margins. To help in that task this paper describes a novel method for deriving the marginal value for the non-use values of wildlife conservation work.

Marginal values for the use benefits of biodiversity can be derived in the course of calculating the benefits which the resource yields using existing methodologies.

Bioprospecting values (the in-situ worth of species given the products they might posses for use in medical research) were yielded originally by multiplying the probability that any one species might yield a saleable product by the average product value. Later it was instead calculated by calculating the marginal impact on the probability of finding the blueprints given product (Craft and Simpson 2001). Such calculations allowed (Simpson, Sedjo et al. 1996) to estimate a hectare of Ecuadorian rainforest to be worth ~ US$20.

The value of a species to recreational hunters is relatively easier to calculate as trophy prices are charged. A Canadian big horned sheep (*Ovis canadensis*) for example have cost US US$50,000 per trophy (Loveridge, Lynam et al. 2005)[1] Direct use values from wild animal products are equally straight forward to derive, wild Scottish Salmon costs around UK£30/kilo wholesale at Billingsgate, the London fish market at the beginning of the season. Values from eco-tourism prove trickier for a specific resource and require the use of travel cost or hedonic pricing techniques to discover their impact on the overall value of a resource travelled to or otherwise bought as part of a holiday.

However, these values can often produce a diluted argument for conservation, though a valid and important one. For instance it is likely that the tiger’s eco-tourism value could be maximised even though they may inhabit an area much smaller than their historic range. Similarly it has long been known that in some circumstances (even under optimal controls) it is economically efficient to drive a species to extinction. Finally the bioprospecting values produced by (Simpson, Sedjo et al. 1996) are

[1] It is therefore, incidentally, surprising that natural resource economists have yet (to our knowledge) to produce bio-economic models to maximise the profits from recreational hunting concerns in the same way as they have for fisheries.

particularly low as have been the amounts spent by medical research companies (Firn 2004). The message from these observations is that markets for biodiversity can only take conservation so far; the remainder must simply be paid for by the willing (Simpson 1999).

The difficulty with non-use values

This remainder is found in ecosystem and non-use values and it is the second of these which we will concentrate on here. Currently the only way to measure a non-use value is using stated preference methodologies and this presents a problem when producing marginal values.

Critics of the technique have complained that the value does not alter appropriately when valuing one gorilla troop or ten. This has been attributed to the embedding effect (Kahneman and Knetch 1992) (Diamond and Hausman 1994) but it could also be a result of unfamiliarity or inability of respondents to deal with the choice so quantitatively.

One of the biggest challenges when producing stated preference valuations lies in the fact that respondents are asked to make decisions with which they are unfamiliar (Arrow, Solow et al. 1993). Careful questionnaire design and informative descriptions, including pictures of the goods they are asked to value can be used to overcome this (Bateman, Carson et al. 2002). However even an experienced naturalist would struggle to produce consistent and meaningful values for changes in a species population from say 10, 000 to 11, 000 and then from say 21,000 to 22,000. It is likely that they would require in depth knowledge of the impact that this would have on the resilience of those populations as this is what they are essentially interested in.

There have been a handful of attempts to produce wildlife oriented values which can be applied more generally and to reasonably small changes. One method completely ignores the benefit to the public and instead simply expresses the change in terms of the costs that these changes will incur (Nalle, Montgomery et al. 2004). This kind of valuation can be quite useful; in some cases it can be enough to show that costs are negligible or at least affordable. It is not ideal however as a positive value for wildlife is a far more powerful negotiating tool than simple affordability. Also in order for environmental economics to remain useful, values need to be produced for the consequences of degradation, since so much of its theory relies upon them.

A report to DEFRA on social attitudes to badger culling in the UK attempted to illicit a value for changes in badger numbers using a choice experiment (DEFRA 2005). Respondents were offered choices between states of the world including varying badger populations, levels of tax and also bovine Tuberculosis in cattle. The authors recognized the difficulty for the public in directly valuing a change in populations and decided that the choice experiment format would help to mitigate this problem.

Earlier (Boyle, Desvousges et al. 1994) valued changes in the number of waterfowl deaths with a view to search for problems people have with valuing scope. They found no significant difference between prices given for a change of 2000 and 200,000 migratory wildfowl deaths in the USA. They openly declined the opportunity to suggest that this finding indicated a general weakness in contingent valuation.

Moreover there is evidence to suggest that the majority of properly conducted valuations are sensitive to scope (Carson 1997). Instead it may highlight the difficulty of eliciting quantitative values in this manner for wildlife.

So it seems unlikely that the preferences tested in the badger valuation revealed more sophisticated desires than for more or fewer badgers. In such case the flat value produced and suggested for use in cost benefit analyses for a change of 100,000 badgers (within the range of 100,000 and 400,000) does not seem particularly convincing. What they present is a value for some large change in badger numbers which should not be used in a more quantitative manner. Attempts to tackle this directly might be more feasible. For example (Macmillan, Hanley et al. 1996), whilst investigating the impact of uncertainty on willingness to pay, valued separately “low” and “high” levels of damage from acid rain on species numbers, but this still does not provide a marginal value.

Our method is still based on stated preference valuations. However the value for a unit of habitat is based on ecological grounds. The public may able to express a willingness to pay for a scheme which professes to “prevent any further loss in the species abundance or distribution” it should then be up to ecologists to decide how this money is spent. The unit value presented does not represent the increase in social welfare that the unit of habitat provides. Nor can it guarantee that any one project to protect a species will yield a consumer surplus should its costs be lower than the benefits derived from this value. However it will ensure that should every project acting towards the overall goal cost less than the benefits derived then the sum total would pass a cost benefit analysis.

Method - *Generic*

We present the method in two parts. First we explain theoretically how the method might be applied to any biodiversity conservation work. We then present a case study for the water vole in the UK. The values used in the case study are unfortunately imperfect for the task and merely presented for illustration. Instead we are suggesting that valuations might in future be designed in order to facilitate their dissemination into marginal values.

Our valuation is derived from a stated preference valuation which offers the respondent easily understood and clearly defined choices without recourse to rigorously quantifying the change for the respondent. This stated preference valuation is used to establish the respondent's willingness to pay for a major goal (for example maintaining the current size and distribution of water vole populations). It then uses ecological data to quantitatively define what the major goal should entail. Figure 1 outlines the steps in the process.

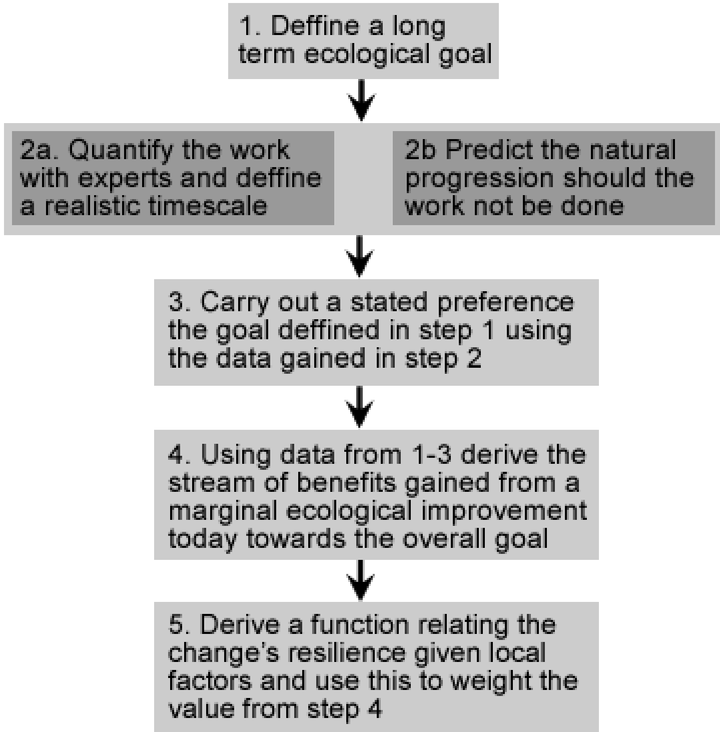

Figure 1: Flow diagram overviewing the steps in the generic process

Step 1 is a task regularly carried out in wildlife conservation. In the UK there are 382 species with Biodiversity Action Plans (BAP) (JNCC). The ecological goal defined in step 1 might be to restore wetlands in England to their 1950's state. This means knowing what the current state of the world is, signified by the blue o in figure 2 and the state of the world which is aimed for is signified by the dotted line.

Once this is understood then professional conservationists would estimate the length of time which this project might take and the progress which might be made at any one time. This provides the vector marked 2a on figure 2 describing the work to be done. In order to value the benefits of the scheme this must be mirrored by estimates for the decline of this resource over the same time period as shown by the line 2b in figure 2.

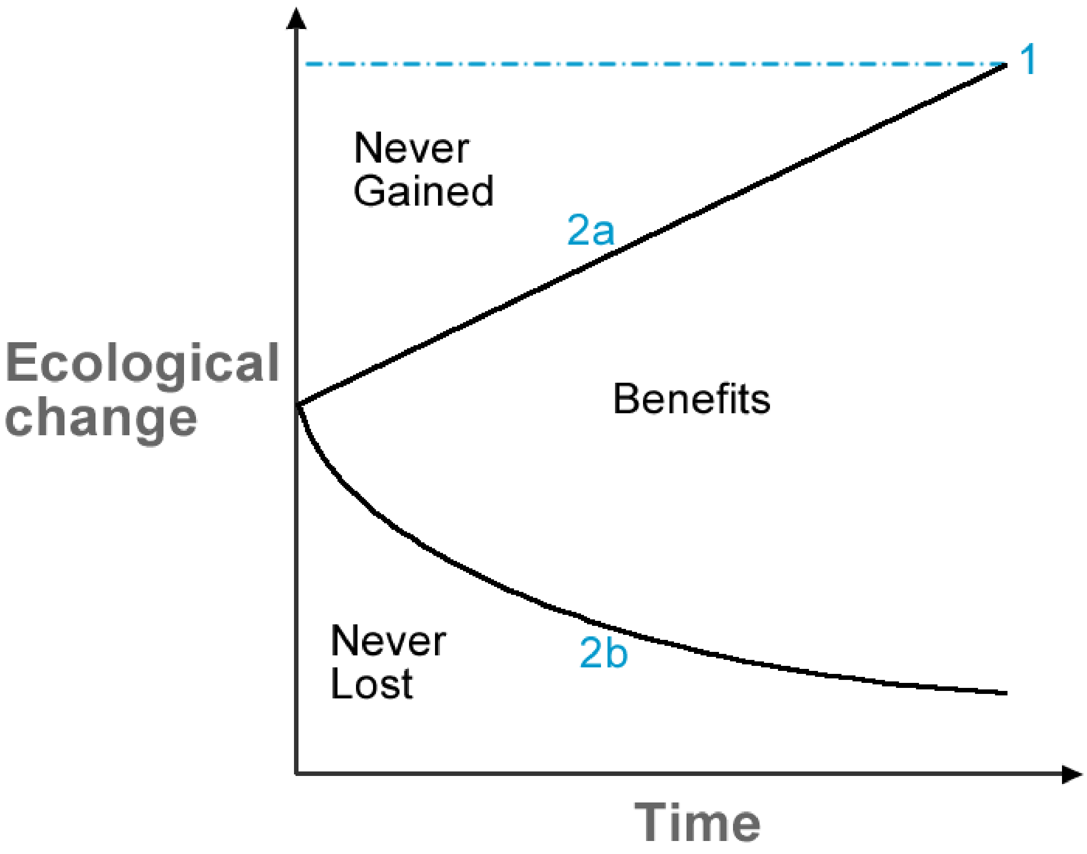

Figure 2: An illustration of steps 1 and 2

This leaves 3 areas on figure 2. The area under 2b which is the resource which is never lost, the area above 2a which is the portion of the resource which will never be gained and that between these functions representing the benefits we wish to value. Step 3, the stated preference valuation, ought now be carried out for this area of benefits. As illustrated in figures 3a and b it becomes quite clear how the value might be parsed into a flat marginal value for the stream of benefits yielding from change in the supply of the resource today.

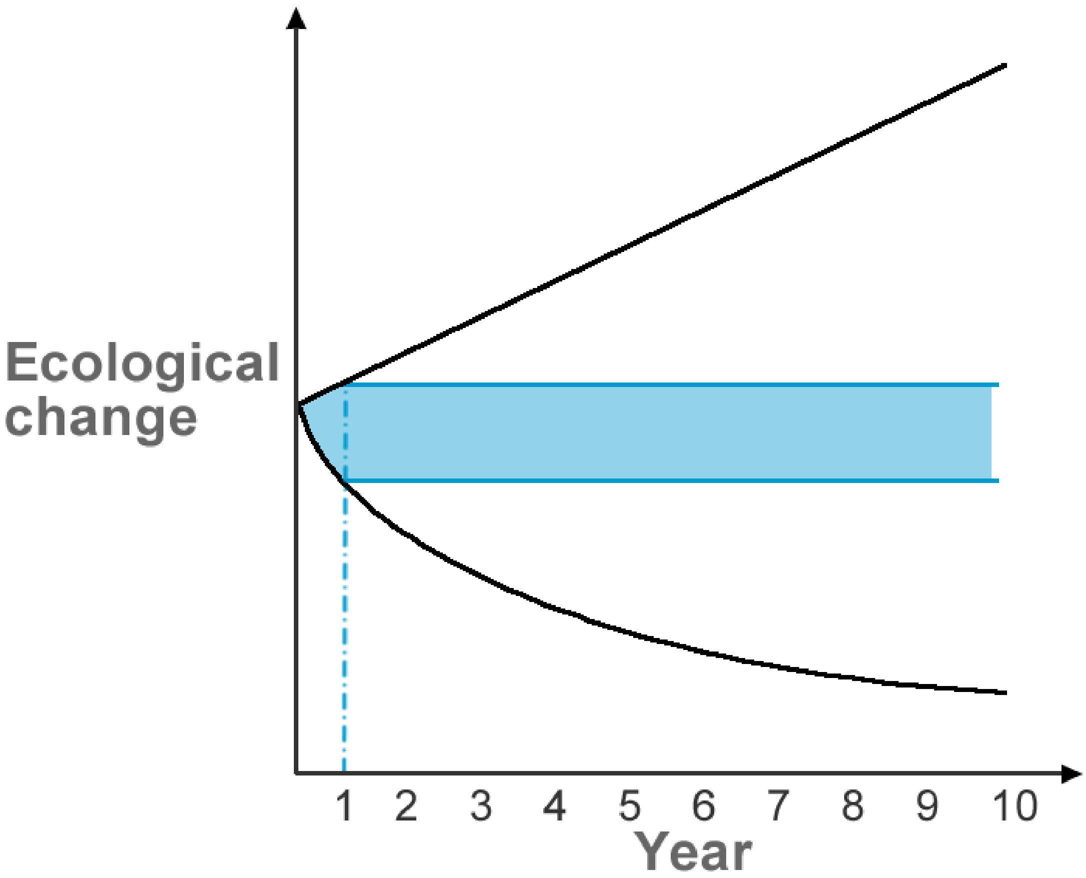

Figure 3a: The benefits yielded in year 1 of a hypothetical project

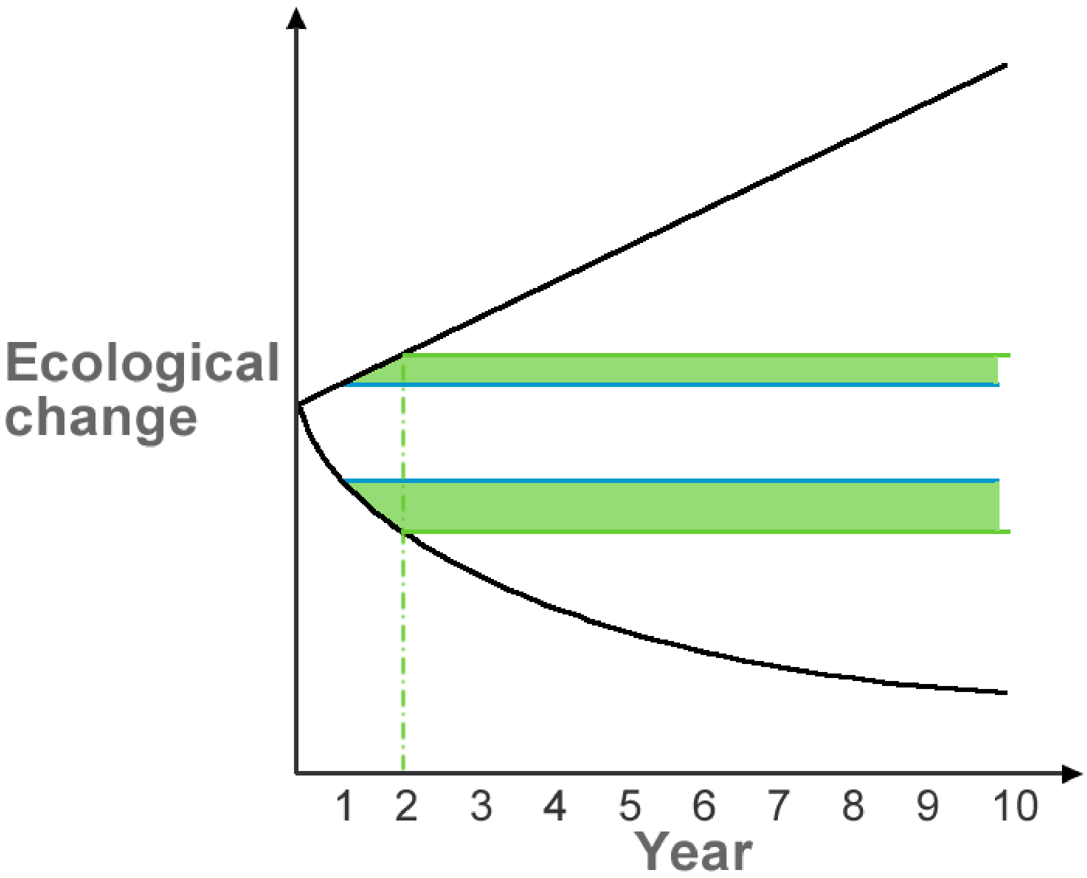

Figure 3b: The benefits yielded in year 2 of a hypothetical project

Figure 3a shows the change in the supply of the resource gained in year 1 and figure 3b shows the change in year 2. Simply calculating the change expected in each year, multiplying this figure by the flat value, discounting this depending on the year and then adding all of the years together produces the full value of the project. Step 3 provided the full value of the project and from steps 1 and 2 the yearly changes can be calculated. From this point some simple algebra will provide the marginal value.

However this marginal value is for a lasting ecological change and an isolated hectare of water vole habitat is unlikely to provide that. For that reason the value must be weighted by the resilience of the ecological change. This might be calculated using a suite of modelling techniques and has a very useful impact on the value. Once weighted an extra unit of habitat will provide benefits not only for itself but also for the resilience of the surrounding habitat.

Method – ***Case Study***

We will now apply this method to the water vole. The water vole is known to many as Ratty from wind in the willows and may be extirpated from much of England by 2020 (Macdonald and Strachan 1999), following 90 years of rapid decline (Strachan, Strachan et al. 2000). This is in part due to the invasive American mink (Lawton and Woodroffe 1991) (Strachan and Jefferies 1993) (Barreto, Macdonald et al. 1998) against which the water vole has no defences. The water vole is a riparian mammal which prefers vegetated banks and emergent vegetation (Lawton and Woodroffe 1991) (Strachan and Jefferies 1993). Agricultural intensification has removed and fragmented many such portions of river. This leads to small isolated populations vulnerable as they no longer benefit from the immigration from neighbouring populations, following stochastic damage to their numbers, leading to local extinctions(Macdonald and Strachan 1999).

Step 1 is derived from the Biodiversty Action Plan for the water vole as defined in 1996. This stated that it would conserve the current distribution and abundance of water voles and increase their range to 1970’s levels by 2010. Two surveys of water vole numbers allow us to define this quantitatively. The first was in 1989/90 and the second in 1996/98, between these years there was a decline of 67.49% a 5.3% year on year decline (Strachan and Jefferies 1993; Strachan, Strachan et al. 2000). Using these data we estimated that water voles inhabited around 137,000 km of waterways in the 1970’s and 33,000 km waterway in 1996.

Step 2 was derived in a similar way. Step 2a was estimated as a linear year on year increase from 1996 to 2010 of 7% and 2a was estimated as a continued decline of 5.3% per year.

Step 3 is the reason why 1996 values were used as this was when a contingent valuation was carried out for the water vole's BAP (White, Gregory et al. 1997). In 1996 White et *al*. carried out a telephone survey of 109 North Yorkshire residents aiming to elicit a value for the water vole's BAP. The valuation was done using a referendum format and a one-off tax increase as the payment vehicle from which a mean willingness to pay of UK£7.44 was elicited.

Step 4, producing the price of a section of waterway, can now be found from the following function:

$$T = P_i.V_0 + \sum([1+r]^{-n}.P.V_n) \quad (1)$$

Where T is the total willingness to pay for the project per person produced by White et al.'s contingent valuation, P is the unit price for a change in vole habitat, V_n is the total change in vole habitat achieved by the project in year n, y is the length of the project from the time of valuation and r is the discount rate[2].

[2] The discount rate was set at 3.5% as stated in the UK Treasury's Green Book (http://greenbook.treasury.gov.uk).

Using this formula the value for a change in vole inhabited waterways of one metre is worth about 1/5000th of a penny per tax payer in North Yorkshire in 1996 pounds Sterling. So if we were to consider this a good proxy for the whole of the UK this figure can be multiplied by the number of tax payers in the UK (~37 million in 1996 [national statistics]) and adjusted for inflation to make a metre of water vole habitat worth ~ UK£12 in 2005 pounds Stirling.

The numbers of water voles in the 1970's was based on a very rough estimate. To provide some sensitivity testing we carried out the calculation again assuming that water voles existed in all of the UK's waterways in the 1970's. This yielded a price of ~ UK£5 per metre.

There may be cases in which two separate prices are required. Respondents frequently show a much greater willingness to pay for increases in the supply of environmental goods than simply to conserve what is there. (Thaler 1980) named this asymmetry of value the, "endowment effect" because your willingness to pay or accept compensation for any change in your supply of goods is determined by your current endowment. In terms of our valuation it suggests that one price is necessary for increases in inhabited waterways and another for conserving what is currently there. Incorporating this into equation 1 gives equation 2:

$$T = P_i . V_{i0} + P_c . V_{c0} + \sum([1+r]^{-n}[P_i . V_{in} + P_c . V_{cn}]) \quad (2)$$

Where the subscript i denotes an increase in the good and c denotes conservation of an existing good. Without separate values the method reaches an impasse. Ideally both values would be elicited in the original investigation, but a rough ratio between these prices can be estimated from (Horowitz and McConnell 2002). Their meta-analysis produced an average ratio of 10.4 for public goods. So by equating $P_i = 10.4 P_c$ and putting this into the equation 2 we can estimate the P_i to be ~ UK£14 per metre.

The endowment effect depends on a respondent having a right to the status quo. In the case of the predicted demise of the water vole the status quo could be argued to be a decline over time. If the public could prevent further loss by avoiding development then the endowment effect would certainly be applicable, but since the decline will for the most part be due to the status quo in terms of current land use and the American Mink, it is arguable that the initial price of UK£12 is the correct one.

The figures we now have are for a metre of habitat which will be newly inhabited and continue to harbour water voles into the future in step 5. Each metre must be weighted by the probability of persistence of water voles in that metre. This can be done in a number of ways, (Nalle, Montgomery et al. 2004) used PATCH (Schumaker, Ernst et al. 2003), to produce probabilities of persistence. Previous work on the water vole has used GIS models to model persistence and spread (Rushton, Barreto et al. 2000). For our purposes a simple model was produced, based on Population Viability Analysis, using VORTEX (Miller and Lacy 1999).

The parameters for a vole were input into VORTEX and it was run for varying carrying capacities. These carrying capacities were recorded along with the probability of the incumbent population persisting for 20 years. These variables were regressed in a cubic function with persistence (S) being explained by carrying capacity (K).

$$S = G(K) = -1.265 + 3.82K - 0.043K^2 + 0.0002K^3 \quad \text{s.t. } 0 \leq S \leq 100 \qquad (3)$$

This function was then converted into a function of persistence and habitat size. Female water voles generally spread themselves out into 30-150 metre long territories while their often polygamous mates range over 60-3 00 metres(Strachan and Moorhouse 2006). Recruitment rates were estimated from (Stoddart 1970) to be 1.4 and from (Moorhouse and Macdonald 2005) to be roughly 1.5 voles per female at any one time. So we can reasonably estimate 50 metres of habitable waterway to support 1 female 0.5 males and 1.5 juveniles, making the carrying capacity of 1 metre roughly 0.05 water voles. Substituting this into the function we derived we can find a function of sustainability in terms of the metres of habitable waterways available to a population:

$$S = F(M) = -1.265 + 0.19M - 0.043(0.05M)^2 + 0.0002(0.05M)^3 \quad \text{s.t. } 0 \leq S \leq 100 \qquad (4)$$

Where S is again the probability of survival over 20 years and M is the number of metres. This function converts the flat value neatly back to a function of diminishing marginal returns. Here for any set of populations worked with, the total present budget for the work is:

$$\text{Total Budget} = \{[S(M) - S(O)]O.P + S(M).N.P\}/100$$
$$\text{s.t. } M = N + P \quad (5)$$

Where M_i is the total length of habitat available to population i, O_i is the original length of habitable river and N_i is the total length of newly habitable river.

This results in a marginal value in two parts, firstly the impact it has on the resilience of existing habitat and secondly its own value. Figure 4 shows how the marginal value of 10 metres of habitat increases when the habitat is built from nothing. This shows where the marginal value reaches a maximum at around UK£18 per metre as the resilience of the surrounding habitat reaches 100%. However this is a relatively unlikely scenario.

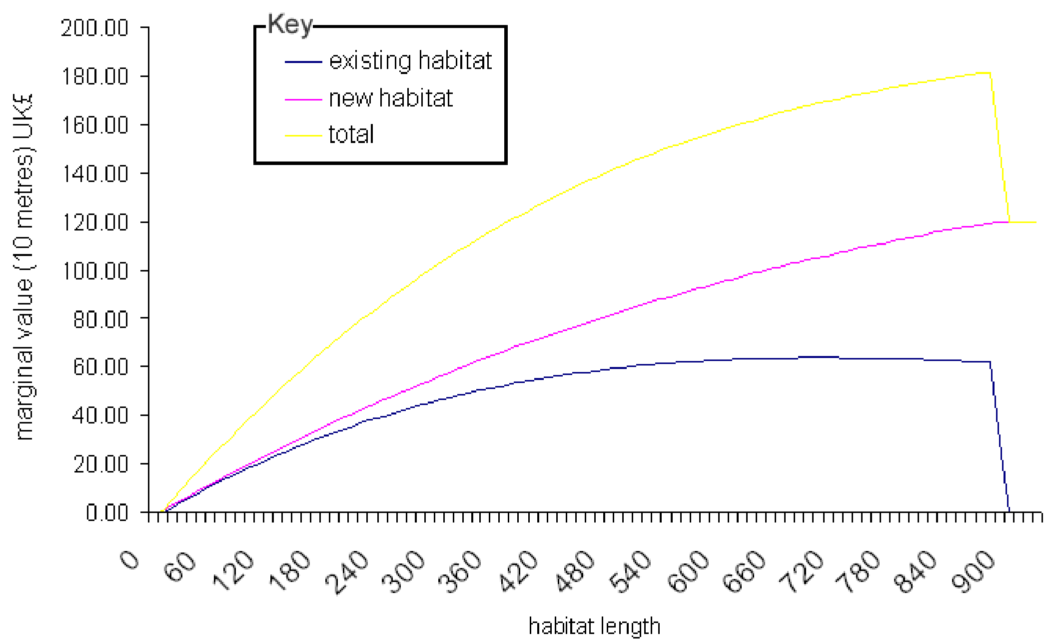

Figure 4: Functions describing value of marginal changes of 10 metres new habitat

For this reason we would like to provide an example for how this tool might significantly aid the local conservationist work with farmers to maximise the efficiency of the work they do. In this example there are 3 fragmented sections of waterway of around 150 metres in length which would otherwise provide excellent water vole habitat. If water voles could emigrate between them then the resilience (in our model) of this habitat would be around 65% whilst separately it is around 0.25%. The waterway between them is on a dairy farm making it relatively costly to take any land out of production and meaning that fencing would be required. Table 1 shows that it costs more than twice as much to provide habitat on arable land than through a dairy farm suggesting that this area would ordinarily be avoided. On the other hand, using the models defined, the value of 300 metres of habitat linking these stretches would be worth UK£34 per metre which exceeds the cost of water vole conservation on the Dairy farm.

Table 1: Estimate of the present value of the costs of water vole protection on farmland in the UK (2005)

	Through arable land	Through dairy farms
Farm Enterprise Opportunity Cost a	£3.49	£15.66
Establishment a	£1.50	£1.50
Fencing a	-	£2.87
Mink control b	£8.33	£8.33
TOTAL	£11.97	£27.02

a estimates derived from Nix farm handbook 2004

b estimates derived from conversation with the UK game conservancy trust

Discussion

The first thing to note is that it is the habitat which is valued and not the population size. If we were to find that welfare alters each time a harvest mouse swims uncomfortably down the gullet of an owl it could prove problematic. We therefore consider changes in habitat size or quality and relate that to its ability to support wildlife.

From this standpoint a first best method might be a marginal value which reflects a habitat change's impact on the viability of the overall population defined earlier. A value, obtained in this manner for a change in habitat might be thought to reflect the utility procured on the public's behalf. However, this would be true only if the public were risk neutral. To counter this perhaps an average Von Neumann Morgenstern utility curve for the public could be derived and used. However there is a weight of evidence to show that this theory does not accurately reflect human behaviour (Kahneman 2002). In this case we might wish to use prospect theory but as of yet this can only provide some guidelines to human decision making and no methodology for defining functions to reflect decision making in a suite of cases. Finally even if prospect theory accurately describes the individuals reaction to risk there may be an argument to suggest that what is rational for the individual is not necessarily so for the collective. This argument would suggest that policy makers should just be playing the odds in order to maximise benefits.

This knotty argument is mooted in this case by a further more practical difficulty. Calculating the marginal benefit of another kilometre of river inhabited by, for instance, otters would require knowledge of the total population and distribution of otters at any one time. While coarse estimates might be available for some species, for most the requisite detail will be unattainable.

Our solution to these difficulties was to derive a flat average value for a unit of habitat, where this unit is in a condition such that the viability of the target species is essentially assured there. This value is then weighted by an estimate of the probability of persistence for that particular unit of habitat given the local, measurable, conditions. It is arguable that this weighting should reflect the apparent risk aversion of the populace (e.g. (Macmillan, Hanley et al. 1996) (Kahneman 2002)) with even lower weight given to populations which are less than 100% resilient. This presents a technical problem which has not so far been properly solved for any good but could perhaps be found in a generalisable mathematical interpretation of prospect theory

Currently, the marginal benefit of an extra unit of habitat has two parts: the direct increase from the extra unit and the increased stability it imparts upon its neighbouring units of habitat. This value will not directly represent the increased utility which that unit has provided in the way economic valuations usually intend (unless by coincidence). It does represent an objective measure of the benefits of small changes in species numbers, which will, if the plan which was originally valued is completed, in aggregate represent the actual welfare change. It is therefore a practical and useful tool for prioritising spending on conservation as well as campaigning for further funds. For that reason however it is perhaps more appropriate to call this a marginal budget rather than a marginal value. For the purposes of this paper however we have for the most part called it a marginal value, since we would hope to use it in place of a marginal value in efficiency calculations.

The value derived for a metre of water vole habitat, may appear unrealistically large. However it is important to remember that this represents the present value of water voles inhabiting that section of river forever. Moreover it is relatively close to the real cost of protecting the water vole. Table 1 gave two estimates of the present value of the cost of protecting a metre of water vole habitat on farmland dedicated to different example enterprises. There is no technical reason for the benefits to be of a similar magnitude to the costs, but it is normatively satisfying that they are not hugely disparate. It may well be that some of the costs presented here will have benefits beyond those for the water vole but extra species cannot be naively added to this value.

In the study which produced the vole value (White, Gregory et al. 1997) a fellow riparian species was also valued. The otter was found to have a WTP of UK£11.91 per person against the Vole’s UK£7.44, whilst the value for both was UK£10.92. This suggests an amenity misspecification bias (Mitchell, Carson et al. 1990) more precisely a policy package bias. It would be difficult for the respondent to disentangle their valuation of the water vole from their value for the riparian habitat in which they reside given the method of elicitation. The values may therefore be close to the value for healthy riparian habitats.

To overcome this policy specification bias when including other environmental benefits to the same habitat it might be more appropriate to enlist a Choice Experiment. In this way, for instance, the value of a metre which contains water voles with some level of persistence can be compared to a length with both water voles and otters or neither. This clearly cannot be done for every species which inhabits a waterway however and value judgements would have to be made. To this end a number of indicator species could be carefully chosen for a variety of representative biomes. A choice experiment might then be carried out for plans to protect each of these habitats and the work weighted by the resilience of the indicator species in these biomes.

The accuracy of the model used to measure this resilience will of course affect the accuracy of the value used. This is true of the methods behind any valuation where constraints of time, money or of a technical nature are binding. There is a great deal of flexibility in producing values in this manner. In some ways this is a strength which allows it to be used in a variety of situations under varying constraints. But the authors of course advise that any environmental valuation is used with care, with special attention being paid to the techniques used, when interpreting the results.

The importance of marginal values to economic analysis is hard to overemphasise but there are specific reasons why this methodology might be particularly useful. The stated preference techniques used to value large scale changes are not cheap. Some of the most expensive valuation studies cost in excess of UK£1 million and the majority costing roughly UK£200,000-£500,000 (Bateman, Carson et al. 2002). Although conservation work can be co-ordinated on larger scales, it is usually implemented and funded on a much smaller scale. In the UK for example the 382 species with action plans being protected by 185 local partnerships, which together are made up of over 1500 organisations (JNCC). This works out at up to 70,670 potential contingent valuations, which, if done properly could cost in the order of UK£14 billion. This is a worst case estimate but its excessive size reflects the problem well since a fraction of this cost would be untenable. This paper suggests the use of a value that could be developed nationally but used effectively for raising funds or awareness at smaller scales by any organisations working on the UK's BAPs or any other small scale conservation enterprise.

The methodology may also be highly useful to those working at a broader scale. It was created as part of a scoping study for the Rural Economy and Land Use programme into maximising the net benefits of conservation work on farmland net of the full economic costs. Recreating habitats to support the 382 species with BAPs in the UK across the 185 local areas is challenging. The requirements of these species must be provided in a structured manner across a landscape with many different landholdings. Each landholding is owned and run by individuals with differing motives and beliefs and whose land and farm businesses needs differ. Intricate plans for agri-environmental work increase the initial costs for those charged with running them and complicate engagement with landholders. On the other hand the more flexible and easily implemented environmental schemes currently in use are liable to fail to prevent the demise of a variety of species (Butler, Vickery et al. 2007).

The primary indicator for the success of agri-environmental schemes is farmer uptake (DEFRA 2002). This has in part led to a mismatch between the causes of and prescriptions to tackle biodiversity degradation in agricultural landscapes in Europe (Dutton, Edwards-Jones et al. submitted). A valuation tool such as the one outlined here might help to make an economic argument for more directed schemes where the efficiency of placing a few metres of habitat through relatively costly land can be realised such as in the example given at the end of the methods section.

This might also support the costs of modelling and monitoring. The former will be necessary to maximise the efficiency of schemes administered at the landscape scale in order to estimate the reaction of wildlife and so avoid leaving some species unprotected as may be occurring currently (Butler, Vickery et al. 2007). The later will be necessary to prove the worth of the schemes.

Currently billions are being spent on agri-environmental schemes (Green, Cornell et al. 2005) and we cannot be sure what benefits if any are being reaped. When the European union is considering converting set aside to biofuel production (FDF 2006); there may be potential for farmers to produce pharmaceuticals (Bowles 2007); and discussions over C.A.P are likely to be dominated by arguments over free trade rather than the green pillar a full economic analysis of the net benefits of the schemes could prove of vital importance to their future.

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