Contributed Paper

Estimating the Value of Non-Use Benefits from Small Changes in the Provision of Ecosystem Services

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Abstract: *The unit of trade in ecosystem services is usually the use of a proportion of the parcels of land associated with a given service. Valuing small changes in the provision of an ecosystem service presents obstacles, particularly when the service provides non-use benefits, as is the case with conservation of most plants and animals. Quantifying non-use values requires stated-preference valuations. Stated-preference valuations can provide estimates of the public's willingness to pay for a broad conservation goal. Nevertheless, stated-preference valuations can be expensive and do not produce consistent measures for varying levels of provision of a service. Additionally, the unit of trade, land use, is not always linearly related to the level of ecosystem services the land might provide. To overcome these obstacles, we developed a method to estimate the value of a marginal change in the provision of a non-use ecosystem service—in this case conservation of plants or animals associated with a given land-cover type. Our method serves as a tool for calculating transferable valuations of small changes in the provision of ecosystem services relative to the existing provision. Valuation is achieved through stated-preference investigations, calculation of a unit value for a parcel of land, and the weighting of this parcel by its ability to provide the desired ecosystem service and its effect on the ability of the surrounding land parcels to provide the desired service. We used the water vole (*Arvicola terrestris*) as a case study to illustrate the method. The average present value of a meter of water vole habitat was estimated at UK£12, but the marginal value of a meter (based on our methods) could range between £0 and £40 or more.*

Keywords: biodiversity offsets, habitat restoration, land-use change, marginal values, non-use values, stated preference

Estimación del Valor de los Beneficios de Cambios Pequeños en el Suministro de Servicios del Ecosistema

Resumen: *La proporción de parcelas de tierra asociadas con un servicio determinado generalmente es la unidad de comercio de los servicios del ecosistema. La valoración de cambios pequeños en el suministro de un servicio del ecosistema presenta obstáculos, particularmente cuando el servicio proporciona beneficios de no uso, como es el caso de la conservación de la mayoría de las plantas y animales. La cuantificación de los valores de no uso requiere valoraciones de preferencia declarada. Las valoraciones de preferencia declarada pueden proporcionar estimaciones de la disponibilidad del público para pagar por una meta de conservación. Sin embargo, las valoraciones de preferencia declarada pueden ser costosas y no producir medidas consistentes para los niveles variables del suministro de un servicio. Adicionalmente, la unidad de comercio, el uso de suelo, no siempre está relacionada linealmente con el nivel de servicios del ecosistema para estimar el valor de un cambio marginal en el suministro de un servicio de no uso – en este caso la conservación de plantas o animales asociada con un determinado tipo de cobertura de suelo. Nuestro método funciona como una herramienta para calcular valoraciones transferibles de cambios pequeños en el suministro de servicios del ecosistema en relación con la disponibilidad existente. La valoración se alcanza mediante investigaciones*

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de preferencias declaradas, el cálculo de una unidad de valor para una parcela de tierra y la ponderación de esta parcela por su habilidad para proporcionar el servicio del ecosistema deseado. Utilizamos a Arvicola terrestris *como un estudio de caso para ilustrar este método. Se estimó que el valor actual promedio de un metro de hábitat de A. terrestres es de £12, pero el valor marginal de un metro (con base en nuestro método puede variar entre £0 y£40 o más.*

Palabras Clave: cambio en el uso de suelo, desbalances de biodiversidad, preferencia declarada, restauración de hábitat, valores de no uso, valores marginales

Introduction

The total value of all of the ecosystems in the world, given that they are necessary to support human existence, defies valuation. A feasible level at which to practice environmental accounting is within the margins of small changes in the supply of ecosystem services (the benefits humans derive from natural systems). The rate of change in revenue with respect to supply can be used to calculate the value of those small changes, known as the marginal value. Given such a value for ecosystem services, one can determine the most efficient and sustainable allocation of resources.

Markets for tradable ecosystem services are not sufficient to conserve all desirable species. The remaining non-use benefits of ecosystem services must be paid for by the willing (Simpson 1999; Macdonald et al. 2007). It is likely that the ecotourism value of a charismatic species could be maximized if the species inhabited an area much smaller than its historic range. For extractive industries, where a wild population is growing relatively slowly and its products are of a high value, profits may be maximized through extirpation (Clark 1972). Theoretical bioprospecting values (Simpson et al. 1996) are low, as have been the actual amounts spent by medical research companies (Firn 2004).

Currently, the only tools able to measure willingness to pay for non-use benefits of environmental protection are stated-preference methodologies. We developed a "quasi-marginal" value for non-use valuations of flora and fauna. The value could, potentially, be applied to other ecosystem services. We also examined how the method we developed could be transferred to "use" values.

Stated-preference methods cannot always calibrate the value of a single organism or a group of organisms, which makes calculating marginal benefits impossible. The obstacle to calculation in part reflects the "embedding effect" (e.g., when valuing the conservation of all tigers and then a single reserve gives a lower price for the reserve than just valuing the reserve alone) (Kahneman & Knetch 1992; Diamond & Hausman 1994) and respondents' unfamiliarity with expressing a preference for public goods (Arrow et al. 1993). The effect of unfamiliarity on choice can be reduced by appropriate design of questionnaires (Bateman et al. 2002). Nevertheless, respondents still struggle to consistently value a change in a species' abundance from, say, 10,000 to 11,000 and then from 21,000 to 22,000. A more direct approach was suggested by Macmillan et al. (1996), who, in their exploration of the impact of uncertainty on willingness to pay, valued separately "low" and "high" effects of acid rain on abundances of different species. Although valuing relative levels of effect does not provide a marginal value, it presents a question that a member of the public is better able to conceptualize and value.

We based our method on the assumption that if the public can express their willingness to pay for a conservation scheme, then the specific quantities traded might remain the domain of natural science. Our approach, valuing—categorical (e.g., large or small) or qualitative anthropocentric concepts of biological diversity, rather than quantitative changes in species richness or abundance, was used by Christie et al. (2006).

The basic traded unit of an ecosystem service is area of land (or water) (Zonneveld 1989; Bockstael 1996; Barbier 2009). The value of an ecosystem service, however, is rarely correlated linearly with the total area covered and is spatially dependent. In our method the unit value reflects the services provided directly by a particular parcel of land and is weighted by its production of the desired ecosystem service as well as its contribution to the stability of the ecosystem service valued within the surrounding landscape. For conservation of animals or plants this ecosystem service would be the resilience of the local population. In this context, we define *resilience* as the likelihood of a population surviving into the foreseeable future. The resulting calculated unit value does not represent the increase in social welfare that the unit of land alone provides. Instead the unit value ensures that should every parcel contributing to the overall goal cost less than the benefits derived from that parcel, then the sum total would pass a cost-benefit test. For this reason that we precede *marginal value* by *quasi*.

To understand the effects of using a flat fraction of a total value rather than evaluating marginal values, consider a persistent wild population of a given species. Removing a few individuals from a population of a million individuals will have little effect on the future viability of the population. Subsequent removals, however, could eventually reduce the probability of the population's long-term persistence to zero (Groom et al. 2005). The value associated with removing individuals from a population is not constant (Ojea & Loureiro 2009). The first individual removed is unlikely to have the same cost as all

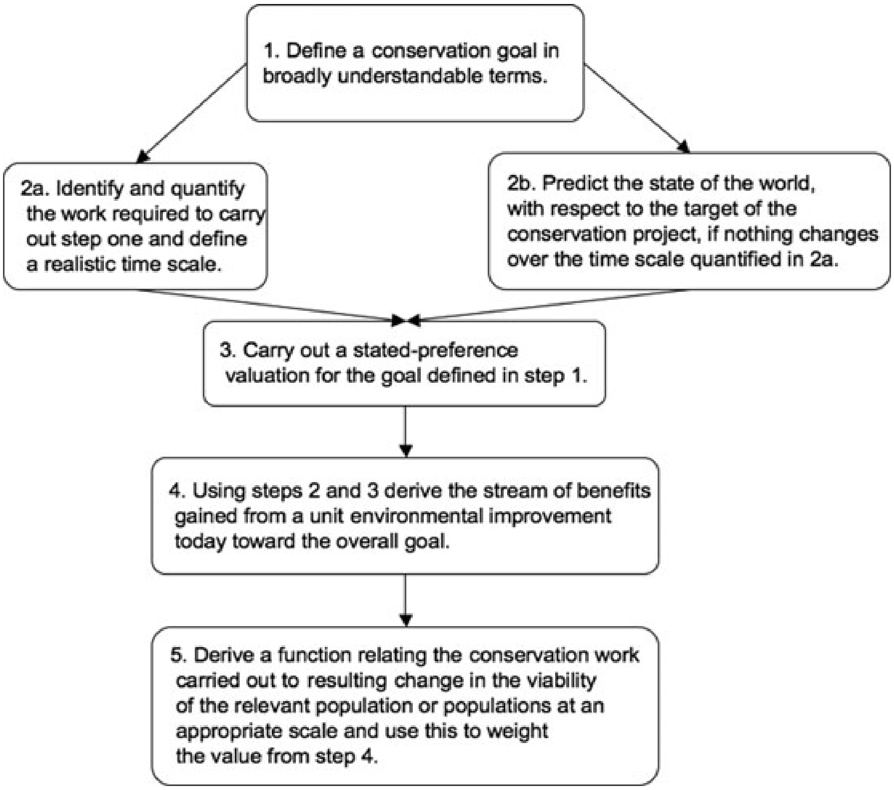

Figure 1. Steps in the calculation of quasi-marginal values.

other individuals removed. Furthermore, it is incorrect to assume that because losing 100,000 individuals costs a given amount, dividing that amount by 100,000 yields the cost of a single individual. Ignoring such facts will overestimate the cost of small changes in the size of a population, underestimate the cost of large changes in population size, and ignore spatial heterogeneity of land cover. Use of a unit value unrelated to total supply as an estimate of the marginal value may be worse than using no estimate because valuations inform decisions about land use.

We assume the value of a parcel of land to the ecosystem within which it is embedded is functionally related to the service it provides and to the ability of that system to resist disturbance. Ecological models can estimate the contribution of a given area to an ecosystem service. We used stated-preference techniques to derive a unit value for a parcel of land but then weighted that value by the predicted change in the provision of the related ecosystem service local to that parcel.

Methods

We begin with a theoretical description of the method. The valuation is derived from a stated-preference valuation that offers the respondent easily understood and clearly defined choices (without access to rigorously quantified information on ecological changes). 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 tiger populations. The method then uses ecological data to define quantitatively the relevant details associated with achieving the major goal. In the tiger example, the current size and distribution of tigers would be measured. There are five steps in the process (Fig. 1).

Step 1: The general goal is defined, which is a common step in conservation projects (Fig. 1). For example, the goal may be to restore wetlands in England to a state comparable to their state in 1950.

Step 2: The change in the provision of the ecosystem service over time as the goal defined in step 1 is quantified. Measuring the change in provision of the service requires thorough knowledge of the current and desired status of the conservation target (dotted line in Fig. 2). Step 2a: Experts estimate the time necessary to achieve this goal and each interim milestone (line 2a in Fig. 2). To value the benefits of restoring the wetlands, in step 2b the change in value of the wetlands over time if nothing is done is estimated (2b in Fig. 2).

Step 3: The area between the two functions 2a and 2b (Fig. 2) describes the total change in provision of the ecosystem service over time, which is valued in step 3 with stated-preference techniques. The simplest payment mechanism would be a single one-off tax payment to cover the full cost of the project. The stated-preference interview ought to present the respondents with the time scale of the project and describes the desired goal and the

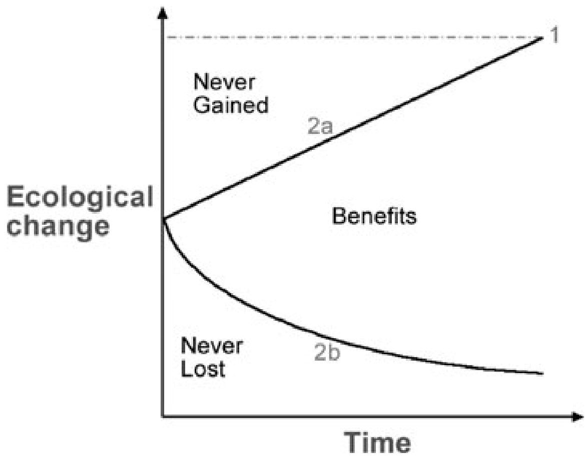

Figure 2. The two functions of ecosystem service provision over time described by steps 1 and 2 in the calculation of quasi-marginal values (2a projects the provision of service over time if the project is carried out, whereas 2b projects provision based on current trends with no intervention). The area below 2b sums to the ecological resources never lost under any circumstances. The area between 2a and 2b shows the total flow of ecosystem services that could be gained from the project.

counter-factual outcome expected if no action is taken. The respondent should be given as much information as possible about the subject, including species uniqueness and function, but care must be taken if the intention is to add potential use values on top of the willingness to pay value. Consider valuing bees or beavers, which might provide pollination and water purification services, respectively. If their values are being calculated separately for these ecosystem services then it would be necessary to inform respondents of this and ask them to concentrate purely on non-use benefits.

Step 4 (Fig. 1): The total value calculated in step 3 is parsed into a flat marginal value for the stream of benefits yielded from a change in the supply of the resource today (Fig. 3). The change in the supply of the resource in years 1 and 2 are shown in Fig. 3a and b, respectively. Calculating the change expected from the conservation work in each year of work, multiplying this figure by the flat value, discounting according to the relevant year, and then adding all of the years together produces the full value of the project. Step 3 (Fig. 1) provides the full value of the project, and from steps 1 and 2 the yearly changes can be calculated. The unit value is then determined algebraically. The unit value we calculated in this manner was for a lasting ecological change. An isolated unit of land is unlikely to provide habitat for a given species in the long term because habitat, demographic parameters, and landscapes are dynamic.

Step 5: The value calculated in step 4 is weighted with a hazard function (Reed & Heras 1992) on the basis of the resilience of the ecosystem service following the ecological change. A suite of modeling techniques can be used to calculate resilience, and in some circumstances the supply of other ecosystem services (e.g., pollination).

Our method is flexible because if the quantitative goals of a conservation project change at any point within the time frame of the original valuation, the calculations can be redone. The values might also be altered if new modeling techniques or new information on the landscape and species of interest become available.

Case Study

We used the water vole (*Arvicola terrestris*) as a case study because their non-use value to the public had already been determined. There is also sufficient ecological

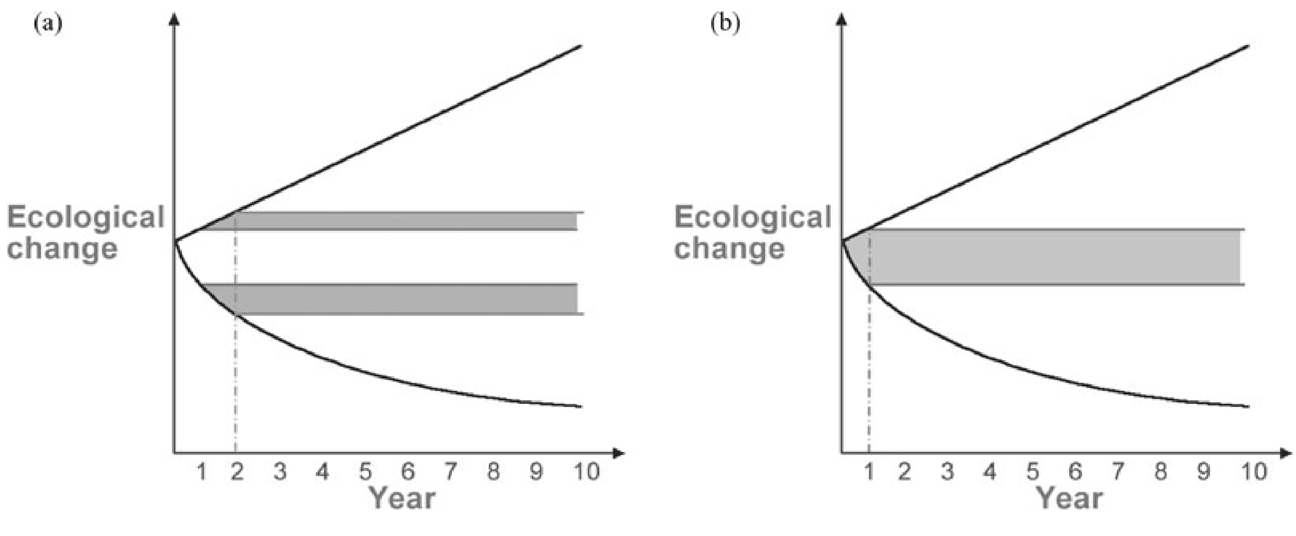

Figure 3. Ecological benefit (shaded area) gained in (a) year 1 and (b) year 2 of a hypothetical conservation project.

information on the status and natural history of water voles in the United Kingdom to create population models. Moreover, the linear habitat of water voles allowed us to use relatively simple modeling techniques to conduct a population viability analysis. Although the case study is useful for illustrating the valuation method, it has some weaknesses. For example, our extrapolations from the contingent valuation and from population data for both humans and water voles may be inaccurate.

The water vole has declined in the United Kingdom over the past 90 years (Strachan et al. 2000) and was on course to be extirpated from much of England by 2020 (Macdonald & Strachan 1999). The main drivers of the decline are predation by the invasive American mink (*Neovison vison*) (Lawton & Woodroffe 1991; Strachan & Jefferies 1993; Barreto, et al. 1998) and habitat loss. Water voles are associated with vegetated banks and emergent vegetation (Strachan & Jefferies 1993). Agricultural intensification has removed and fragmented many such areas along rivers.

We determined the goal (step 1) from the biodiversity action plan for the water vole as outlined in 1996 (White et al. 1997) rather than from the current biodiversity action plan for the species. The stated aim of the plan was to conserve the current (1996) distribution and abundance of water voles and increase their distribution to 1970s levels by 2010. Two surveys (1989–1990 and 1996–1998) of water vole abundance allowed us to estimate their 1970s abundance. Between surveys the number of water voles declined by 67% (a mean of 5.3% per year) (Strachan & Jefferies 1993; Strachan et al. 2000). We used these data to estimate that water voles inhabited around 137,000 km of waterways in the 1970s and 33,000 km of waterways in 1996.

We estimated the change in distribution of the water vole that would be achieved by implementing the biodiversity action plan (step 2a) as a 7% increase per year in abundance of voles and thus the length of waterways occupied (an increase from 33,000 km to 137,000 km of occupied waterways). We estimated the annual reduction in water vole distribution if no action was taken (step 2b) as 5.3%.

A contingent valuation for the water vole's biodiversity action plan was carried out in 1996 (White et al. 1997) (step 3 in our process). White et al. conducted a telephone survey of North Yorkshire residents. They used a referendum format and a one-time tax as the payment vehicle. They found that the mean willingness to pay was UK£7.44/taxpayer (hereafter, £ is UK£).

To estimate the monetary value of a section of waterway (step 4) we used the following equation:

$$T = \sum_{n=0}^{y} ([1 + r]^{-n}.P.V_n), \tag{1}$$

where T is the total willingness to pay for the project per person (£7.44), P is the unit price for a change in quantity of water vole habitat, V_n is the total change in the length of waterways occupied by water voles 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.

The value of a change in vole-occupied waterways of 1 linear meter was worth roughly 1/5000th of a penny per taxpayer in North Yorkshire (1996 pounds Sterling). We multiplied this value by the number of taxpayers in the United Kingdom (approximately 37 million in 1996, National-Statistics 2010) and adjusted for inflation to determine that a linear meter of water vole habitat was worth approximately £12 (2005 pounds Sterling).

Our estimated length of waterway occupied by water voles in the 1970s was based on an uncertain extrapolation from later population declines. To evaluate the sensitivity of the price to our assumptions, we recalculated P, assuming that water voles existed in all U.K. waterways in the 1970s. This recalculation yielded a price of £5/m, which we considered a conservative estimate.

In some cases two separate values for a unit of land cover need to be determined. Frequently, respondents show a much greater willingness to pay for increases in the supply of environmental goods than to conserve the existing supply. Thaler (1980) called this the "endowment effect" because willingness to pay or accept compensation for any change in the supply of goods is determined by the consumer's current endowment. To control for the endowment effect we produced one price for restoring water voles to habitat from which they have been extirpated and another for conserving water voles where they currently persist. Incorporating the endowment effect into Eq. 1 yields

$$T = \sum_{n=0}^{y} ([1 + r]^{-n}[P_i.V_{in} + P_c.V_{cn}]), \tag{2}$$

where i denotes an increase in the good and c denotes conservation of an existing good. If separate values are not elicited in step 3 an impasse could be reached, but a ratio between these prices can be estimated from Horowitz and McConnell (2002). Their meta-analysis produced an average ratio of 10.4:1 for public goods (making $P_i = 10.4\ P_c$). So, we used Eq. 2 to estimate P_i as £14/m.

The endowment effect is related to the status quo, in our case the current endowment of vole provision. The public in our example is currently endowed with a decline in water voles over time. If the public could prevent further loss, say by avoiding changes in land use such as agricultural intensification or other development, then the endowment effect would be applicable. Because the predicted decline of water voles is associated with current land use and the effects of the American mink, the initial price of £12 is arguably correct.

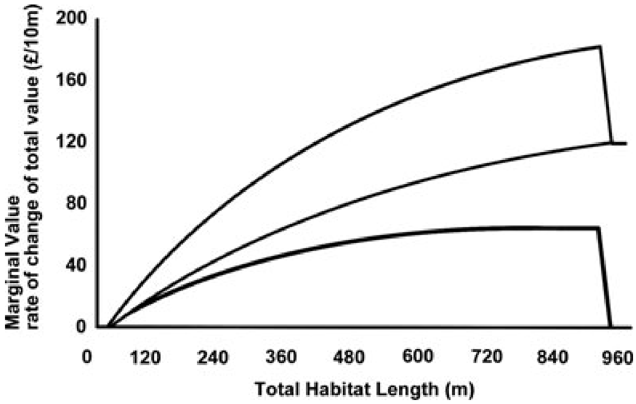

Figure 4. Functions describing the rate of change of the total value of waterways for water vole conservation as the quantity of habitat increases by 10-m increments.

The P, P_i, and P_c refer to a meter of water vole habitat that will be colonized and occupied over time. In step 5, we weighted each meter by the probability of persistence of water voles in that meter. The water vole has been the subject of spatially explicit population models (Rushton et al. 2000), but we built a simple model of population viability, with VORTEX (Miller & Lacy 1999).

We ran VORTEX parameterized for water voles for different carrying capacities. We recorded the carrying capacities and the corresponding probability of the population persisting for 20 years estimated by the model. Given the short-life span and fecundity of the water vole, 20 modeled years is sufficient. We regressed carrying capacity and persistence probabilities in a cubic function in which persistence, S, was explained by carrying capacity, K.

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

(The abbreviation s.t. is, "subject to" and refers to a constraint on the values of the equation.)

We converted Eq. 3 into a function of persistence related to quantity of water vole habitat. Female water voles disperse along 30- to 150-m long territories, whereas their often polygamous mates range over 60- to 300-m-long territories (Strachan & Moorhouse 2006). We used a recruitment rate of 1.4 (Stoddart 1970) and estimated there were 1.5 males per female (Moorhouse & Macdonald 2005). Thus, we assumed that 50 m of waterway supported one female, 0.5 males, and 1.5 juveniles, which means the carrying capacity of 1 m of territory is roughly 0.05 water voles. Substituting carrying capacity for the length of waterway required to support that number of voles into Eq. 3 relates resilience to length of waterway:

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

where S is the probability of survival over 20 years and M is the number of meters. We used Eq. 4 to weight P derived from Eq. 2 to convert the flat value into a function with diminishing marginal returns.

We used Eqs. 1–4 to value a project to restore water vole habitat. Remember that our original valuation exercise was for a large-scale project, whereas the weighted price derived allows phases of the project to be valued. We used the following equation to determine a budget for a phase of the project, which if kept to in the latter phases ensures the overall project does not exceed total willingness to pay:

$$\text{total budget} = \sum_{i=1}^{n} \{[S(M_i) - S(O_i)]\, O_i.P + S(M_i).N_i.P\}/100 \text{ s.t. } \forall i \quad (M_i = N_i + P_i), \tag{5}$$

where M_i is the total length of habitat available to the population following restoration, O_i is the original length of habitable river (existing before intervention), and N_i is the total length of newly habitable river.

We used Eq. 5 to provide a marginal value in two parts: the impact restoration has on the resilience of existing habitat and the weighted value of the restored waterway itself. The marginal value of 10 m of habitat increased when the habitat was newly constructed in the absence of an existing populated waterway (Fig. 4). "Existing habitat" describes the value added to previous lengths of river by the "new" 10 m, and the "total" adds the new 10-m value to the change in the existing habitat. The marginal value reaches a maximum at around £18/m because the resilience of the surrounding area reaches 100% (Fig. 4).

When building on existing habitat, these marginal values can increase. As an example of how our approach might help maximize project efficiency, consider two parallel waterways separated by 150 m of dairy farm. Each waterway has 425 m of vole habitat, and the vole populations associated with each have a 62% probability of persistence for the foreseeable future. It was more than twice as costly to protect water voles on dairy farms than on arable land given the need for fencing and the increased value of land to the farmer (Table 1). Nevertheless, our approach revealed there might be a strong economic case for protection on the dairy farm. Connecting the two vole populations with 150 m of habitable waterway would create a total habitat length of 1 km (425 m + 425 m + 150 m), which would support a population with near 100% probability of persistence for the foreseeable future. Habitat along each waterway would increase in value by £1938 (450 m [length] × 0.38 [change in resilience] × 12 [price per meter]), and the newly constructed habitat would be worth the full £12/m. The full value of restoring 150 m would be £5677, meaning

Table 1. Estimate of the present value of the costs of protection of water vole on farmland in the United Kingdom per meter of riverbank on both sides of a river (2005).

Costed item	*Cost on arable land (UK£/m)*	*Cost on dairy farms (UK£/m)*
Farm enterprise opportunity cost[a]	3.49	15.66
Establishment[a]	1.50	1.50
Fencing[a]	n/a	2.87
Mink control[b]	8.33	8.33
Total	11.97	27.02

[a] *Estimates derived from Nix (2004).*
[b] *Estimates derived from personal communication with personnel of the U.K. Game Conservancy Trust.*

that each new meter created would on average be worth £37.85.

Discussion

The provision of an ecosystem service rarely is linearly related to the area of land committed to the use associated with service provision. Accordingly, some calculation is necessary to ensure the price of the traded unit is related to the service rather than merely the unit of land associated with the service. In our method the marginal benefit of an extra unit of habitat for an animal or plant has two parts: a direct increase in habitat quantity and an increase in probability of population persistence. The method can allow a relatively small increase in habitat quantity to have a high value where that increase greatly increases the probability of persistence of the population.

Our method yields a value that could be used as an approximation of the marginal value of change in quantity or quality of habitat from a stated-preference survey. This value is an objective measure of the benefits of small changes in abundance and would in aggregate represent the associated change in human "welfare." Beyond the period of time or remit of the original valued conservation plan this value is not strictly applicable, but it might be used as a value of "benefit transfer."

Valuing Risk

Our method is a compromise between tractability of calculation and accuracy of the value of an ecosystem service. An improved method might produce a marginal value to society of a change in quantity or quality of habitat as it affects the viability of the species at the national rather than local level.

Calculating the marginal benefit of a unit of habitat occupied by a given species with respect to the species as a whole would require knowledge of the global population and distribution of that species at any one time. For most species this information is not available. We derived a flat average value for a unit of habitat and then weighted the value by an estimate of the unit's contribution to the species' probability of persistence.

Whether this weighting should reflect the apparent risk aversion of society is debatable (e.g., Kahneman 2002). Our method assumes the public is risk neutral. A risk-neutral public might consider a project that is likely to result in 50% probability of persistence of the global population of water voles equivalent to a project that would result in near 100% probability of persistence of half the global population of water voles. Economists are still trained to relate the probabilities and outcomes of a project to human welfare through a Von Neumann Morgenstern utility curve. Nevertheless, Von Neumann and Morgenstern's theory may not accurately reflect human behavior (Kahneman 2002). Prospect theory is an alternative way to describe preferences with risks. Currently, prospect theory provides only guidelines to human decision making, not a widely accepted method for defining functional models (although attempts have been made to do so; Starmer 2000). Even if prospect theory accurately describes an individual's reaction to risk, what is rational for the individual is not necessarily rational for the collective public. The risk-neutral preference assumption may be unavoidable and desirable. Other researchers have made similar simplifications when including risk in economic decisions (Reed & Heras 1992; Barbier 2009).

Best Practice

White et al. (1997) also estimated the public's willingness to pay for otters (*Lutra lutra*). Respondents were willing to pay £11.91 per person to protect otters against the vole's £7.44; willingness to pay for both was £10.92. This result suggests an amenity misspecification bias (Mitchell & Carson 1990) or, more precisely, a policy package bias. This is because, given the methods of White et al. (1997), it would be difficult for a respondent to disentangle a valuation of the water vole from a value for the riparian habitat of the water voles. The values attributed to the voles may therefore be close to the value for functioning riparian systems. This form of confusion can be minimized by providing respondents to a valuation survey with a suite of public goods to value and by allowing respondents to change their minds.

Our case study illustrated the valuation approach. Best practice would be to value land-cover types rather than one species and to provide respondents with information on the suite of species typically associated with that land-cover type. Ecological modeling might focus on keystone species or ecosystem engineers.

Valuation should be as comprehensive as is financially practicable (Bateman et al. 2002). It is important that the valuation presents information on the time scale of the project, aims of the project, and the expected outcome if the project is not implemented. Care must also be taken not to double count if use values are added to the willingness to pay values because respondents may include use values in their valuation.

Calculated use values related to land cover may also be weighted as we describe here. The weight would not necessarily be directly related to the resilience of an ecosystem; rather, it would be related to the services the ecosystem provides. In the case of bees, for instance, services might be the pollination provided to agriculture.

Technical Advantages of the Method

Use of stated-preference techniques to value spatially extensive environmental changes is expensive. Some valuation studies cost more than £1 million, and the majority cost roughly £200,000–£500,000 (Bateman et al. 2002). In some regions, conservation usually is funded and implemented at smaller extents. For example, in the United Kingdom the 382 species with biodiversity action plans are protected by 185 local partnerships, which together comprise over 1500 organizations (UKBPSC 2010). Conducting 70,670 contingent valuations, one for each species and partnership, could cost approximately £14 billion. Instead, we suggest the use of fewer values that could be developed nationally or internationally but used effectively by organizations working on conservation enterprises on smaller scales.

When market externalities are not valued, benefits are ignored by the market. Without tractable methods for producing marginal values, at best coarse average values are used to value the environment (benefit transfer). Such coarse values can at times be worse than no value if they support decisions that are unlikely to achieve a conservation goal. We believe our method produces values for marginal changes in the supply of non-use environmental benefits that can fill this gap.

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