Economic reasons for conserving wild nature

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On the eve of the World Summit on Sustainable Development, it is timely to assess progress over the ten years since its predecessor in Rio de Janeiro. Loss and degradation of remaining natural habitats has continued largely unabated. However, evidence has been accumulating that such systems generate marked economic benefits, which the available data suggest exceed those obtained from continued habitat conversion. We estimate that the overall benefit: cost ratio of an effective global programme for the conservation of remaining wild nature is at least 100 : 1.

Humans benefit from wild nature (*1*) in very many ways – aesthetically and culturally; via the provision of ecological services such as climate regulation, soil formation and nutrient cycling; and from the direct harvest of wild species for food, fuel, fibres and pharmaceuticals (*2*). In the face of increasing human pressures on the environment these benefits should act as powerful incentives to conserve nature, yet evaluating them has proved difficult because they are mostly not captured by conventional, market-based economic activity and analysis.

In 1997, Costanza *et al.* published a synthesis (*3*) of over 100 attempts to value ecosystem goods and services using a range of techniques including hedonic pricing, contingent valuation and replacement cost methods (*4*). Using case studies to derive average values per hectare for each of 17 services across 16 biomes and then extrapolating to the globe by multiplying by each biome's area, the Costanza team estimated the aggregated annual value of nature's services (updated to 2000 US $) to lie in the range $18 – 61 trillion ($10^{12}$), around a rough average of ~$38 trillion. These figures are of similar size to global Gross National Product (GNP), but have been criticised by some in the economic community (*5-9*).

One problem is that such macroeconomic extrapolations are inconsistent with microeconomic theory: extrapolation from the margin to a global total should incorporate knowledge about the shape of the demand curve (*3*, *5-8*). In practice, it is very likely that per unit demand for non-substitutable services escalates rapidly as supply diminishes, so that simple grossing-up of marginal values (as is also done in calculating GNP from prices) will probably underestimate true total values. On the other hand, high local values of services such as tourism may not be maintained if extrapolated worldwide. In addition, while some policy decisions are made using macroeconomic indicators, many others are made at the margin, and so are more appropriately informed by marginal rather than total valuations (*9*).

Another problem with the original estimate is that landscapes can yield substantial (albeit rather different) flows of goods and services after as well as before conversion by humans (which is of course why people convert them). A clearer picture of the value of retaining habitat in relatively undisturbed condition might therefore be obtained by estimating not the gross values of the benefits provided by natural biomes, but rather the difference in benefit flows between relatively intact and converted versions of those biomes.

Net marginal benefits

To address these concerns, we reviewed over 300 case studies, searching for matched estimates of the marginal values of goods and services delivered by a biome when relatively intact, and when converted to typical forms of human use. To ensure we did not neglect private benefits of conversion, studies were only included if they covered the most important marketed goods, as well as one or more non-marketed services delivering local social or global benefits. We cross-validated figures for individual goods and services with other estimates from similar places. Last, we checked that the comparisons across different states of a biome used the same valuation techniques for particular goods and services. Our survey uncovered only five examples which met all these criteria. Here we summarise their findings, with all figures expressed as Net Present Values (NPVs, in 2000 US \$ ha^{-1}), and using the discount rates considered by the authors (see Fig. 1 and online material [*10*] for further details).

Two studies quantified net marginal benefits of different human uses of tropical forest areas. Kumari compared the values obtained from timber plus a suite of Non-Timber

Forest Products (NTFPs), as well as the values of water supply and regulation, recreation, and the maintenance of carbon stocks and endangered species, for forests under a range of management regimes in Selangor, Malaysia (*11*). Compared with two methods of reduced-impact logging, high intensity, unsustainable logging was associated with greater private benefits through timber harvesting (at least at high discount rates and over one harvesting cycle), but reduced social and global benefits (through loss of NTFPs, flood protection, carbon stocks and endangered species). Summed together, the Total Economic Value (TEV) of forest was some 14% greater when under more sustainable management (at ~\$13,000 cf \$11,200 ha^{-1}).

A study from Mount Cameroon comparing low impact logging with more extreme land-use change again found that private benefits favour conversion, this time to small-scale agriculture (*12*). However, a second alternative to retaining the forest - conversion to oil palm and rubber plantations – in fact yielded negative private benefits, once the effect of market distortions was removed. Social benefits, from NTFPs, sedimentation control and flood prevention, were highest under sustainable forestry, as were global benefits from carbon storage and a range of option, bequest and existence values. Overall, the TEV of sustainable forestry was 18% greater than that of small-scale farming (~\$2570 cf \$2110 ha^{-1}), while plantations had a negative TEV.

Three other biomes yielded single studies meeting our criteria. Analysis of a mangrove system in Thailand revealed that conversion for aquaculture made sense in terms of short-term private benefits, but not once external costs were factored in (*13*). The global benefits of carbon sequestration were considered to be similar in intact and degraded systems. However, the substantial social benefits associated with the original mangrove cover - from timber, charcoal, NTFPs, offshore fisheries and storm protection – fell to almost zero following conversion. Summing all measured goods and services, the TEV of intact mangroves exceeded that of shrimp farming by around 72% (~\$60,400 cf \$16,700 ha^{-1}).

van Vuuren and Roy (*14*) reported that draining freshwater marshes in one of Canada's most productive agricultural areas yielded net private benefits (in large part because of substantial drainage subsidies). However, social benefits of retaining wetlands, arising from sustainable hunting, angling and trapping, greatly exceeded agricultural gains. Consequently, for all three marsh types considered, TEVs were higher when the wetlands remained intact, exceeding figures for conversion by a mean of 58% (~\$8800 cf \$3700 ha^{-1}).

Finally, a synthesis of economic studies examining Philippine reef exploitation demonstrated that despite high initial benefits, destructive techniques such as blast fishing had a far lower NPV of private benefits than sustainable fishing (*15*). The social benefits of sustainable exploitation – from coastal protection and tourism – were also lost upon dynamiting reefs. As a consequence, the TEV of retaining an essentially intact reef was some 73% higher than that of destructive fishing (at ~\$3300 cf \$870 ha^{-1}).

One clear message from our survey is the paucity of empirical data on the central question of the changes in delivery of goods and services arising from the conversion of natural habitats for human use. For ten of Costanza *et al.*'s (*3*) largely natural biomes (including rangelands, temperate forests, rives and lakes, and most marine systems) we found no studies that met all of our criteria. For the four biomes which were analysed, only a handful of well-established ecosystem services were considered, and some particularly valuable services, such as nutrient cycling, waste treatment, and the provision of cultural values, were not examined at all.

Despite the limited data, our review also suggests a second broad finding: in every case examined, the loss of non-marketed services outweighs the marketed marginal benefits of conversion, often by a considerable amount. Across the four biomes studied, mean losses in Total Economic Value due to conversion run at roughly one half of the TEV of relatively intact system (mean = 54.9%; SE = 13.4%; n = 4). This is certainly not to say that conversion has never been economically beneficial – in most instances, past clearance of forests and wetlands for prime agricultural land and other forms of development probably benefited society as a whole. But unless the present case studies or the range of services and biomes examined in the literature is extremely unrepresentative (and we know of no reason why this should be the case), our synthesis indicates that nowadays, conversion of remaining habitat for agriculture, aquaculture or forestry often does not make sense from the perspective of global sustainability.

Continuing losses

These results therefore provide a clear and compelling economic case, alongside socio-cultural and moral arguments (*16*-*18*), for us to strengthen attempts to conserve what remains of natural ecosystems. Yet when we summarised available estimates of recent trends in the global status of natural habitats and free-ranging vertebrate populations, we found that although key data are again disturbingly scarce, they show that rates of conversion are high across most biomes (*10*).

We included in our survey any estimate of global trend in habitat cover based on a series which began in 1970 or later and included a period of at least five years after the United Nations Conference on Environment and Development in 1992. We supplemented this with biome-specific indices based on time series data on populations of wild vertebrates, derived from the WWF 2000 Living Planet Index (LPI) and FAO fisheries data (*19*, *20*). For three biomes we found two estimates using different methods and either largely or wholly independent data. In each case the two estimates were remarkably similar (*10*), and so were averaged to yield single estimates of rates of change. Data such as these, quantifying trends in areal coverage and in populations, in some ways provide a more tractable measure of the scale of the ongoing crisis facing nature than do estimates of extinction rates, which are harder to document and more difficult to link to monetary values.

Overall we found that five out of the six biomes measured have experienced net losses since the 1992 Rio summit, with the mean rate of change across all measured biomes running at -1.2% per year, or –11.4% over the decade (Fig. 2; see [*10*] for details). Hence the capacity of natural systems to deliver goods and services upon which we depend is decreasing dramatically. Costing the overall value of these losses is fraught with the problems of extrapolation and data availability already discussed. Nevertheless, it is sobering to calculate that if the Costanza *et al.* aggregate figures (*3*) and our estimate of the proportion of TEV lost through habitat change are roughly representative, a single year's habitat conversion costs the human enterprise, in net terms, of the order of $250 billion that year, and every year into the future (*10*). Why, then, is widespread habitat loss still happening, and what can we do about it?

Reasons for continued conversion

In economic terms, our case studies illustrate three broad, inter-related reasons why we are continuing to lose natural ecosystems despite their overall benefits to society (*21*). First, there are often failures of information. For many services, we lack valuations of their provision by natural systems, and particularly of changes in this provision as human impacts increase. While this is an understandable reflection of substantial technical difficulties, we believe that future work needs to compare delivery of multiple services across a range of competing land uses if it is to better inform policy decisions. Our examples show that even when only a few ecosystem services are considered, their loss upon conversion typically outweighs any gains in marketed benefits.

Second, these findings highlight the fundamental role of market failures in driving habitat loss. In most of the cases we studied, the major benefits associated with retaining systems more or less intact are non-marketed externalities, accruing to society at local and global scales. Conversion generally makes narrow economic sense because such external benefits (or related external costs, as in the case of the damage caused by shrimp farming [*13*]) have very little impact on those standing to gain immediate private benefits from land-use change. Hence conserving relatively intact habitats will often require compensatory mechanisms to mitigate the impact of private, local benefits foregone, especially in developing countries. We see the development of market instruments that capture at a private level the social and global values of relatively undisturbed ecosystems - for instance through carbon or biodiversity credits or through premium pricing for sustainably harvested wild-caught fish or timber (*22*, *23*) – as a crucial step towards sustainability.

Third, the private benefits of conversion are often exaggerated by intervention failures. In the Cameroon study, for example, forests were cleared for plantations because of private benefits arising from government tax incentives and subsidies (*12*). The same is true for the Canadian wetland example (*14*), as well as for many other wetlands across USA and Europe (*24*). While over the short term these programs may be rational with respect to public or private policy objectives, over the longer term many result in both economic inefficiency and the erosion of natural services. Globally, the subset of subsidies which are both economically and ecologically perverse totals between \$950 and \$1950 billion each year, (depending on whether the hidden subsidies of external costs are also factored-in [*25*, *26*]). Identifying and then working to remove these distortions would simultaneously reduce rates of habitat loss, free up public funds for investing in sustainable resource use, and save money (*25-27*).

Costing conservation

Tackling these underlying economic problems requires action on many levels, but should in due course result in public and private decision makers acting to reduce conversion of remaining habitats worldwide. More immediately, given concerns about the practicalities of exploiting natural resources sustainably, one of the most important strategies to safeguard relatively intact ecosystems is the maintenance of remaining habitats in protected areas. This costs money, and predictably, our current undervaluation of nature is reflected in marked underinvestment in reserves. To the best of our knowledge the world spends (in 2000 US \$) ~ \$6.5 billion each year on the existing reserve network (*28*). Yet half of this is spent in USA alone. Globally, despite increased expenditure since the Rio summit by both international institutions and private foundations, available resources for existing reserves fall far short of those needed to meet basic management objectives (*29*). Moreover, terrestrial and marine reserves currently cover only around 7.9% and 0.5% of the earth's land and sea area, respectively (*30*, *31*) – well below the minimum safe standard considered necessary for the task of maintaining wild nature into the future (*32*-*34*).

To estimate the resources needed to meet this shortfall on land, we reworked recent calculations (*28, 35*) of the costs of properly managing existing terrestrial protected areas and expanding the network to cover around 15% of land area in each region. We found that a globally effective network would require around an annual outlay of between ~ $20 and $28 billion (including payments to meet private opportunity costs imposed by existing and new reserves, spread out over 10y and 30y respectively [*10*]). New work derived from the costs of existing marine reserves suggests that an equivalent initiative for the world's seas, this time covering 30% of total area (*34, 36*), would cost at most ~ $23 billion per year in recurrent costs, plus ~ $6 billion per year (over 30 years) in start-up costs (*10*). The estimated mean the total cost of an effective, global reserve programme on land and at sea is some $45 billion per year. This sum dwarfs the current $6.5 billion annual reserve budget yet could be readily met by redirecting less than 5% of existing perverse subsidies (*25*, *26*). The crucial question is whether this is a price worth paying.

Although limited data make the answer imprecise, they indicate that conservation in reserves represents a strikingly good bargain. We assumed that the mean proportional loss of value upon conversion recorded in our case studies is representative of all biomes and services, and that previous gross per hectare values of those services are roughly correct (*3*). If these assumptions are valid, then our hypothetical global reserve network would ensure the delivery of goods and services with an annual value (net of benefits from conversion) of between ~ $4400 and $5200 billion, depending on the level of resource use permitted within protected areas, and with the lower number coming from a network entirely composed of strictly protected reserves (for working, see [*10*]). The benefit : cost ratio of a reserve system meeting minimum safe standards is therefore around 100 : 1.

Put another way, the case studies, Costanza *et al.*'s (*3*) service values or our reserve costs would have to be off by a factor of 100 for the reserve programme envisaged to not make economic sense. We consider errors of this size to be highly unlikely, as most of our assumptions are conservative (for other sensitivity analyses, see [*10*]). For example, in terms of the values of services, we assume that unit values will not increase as supply diminishes, that nature reserves do not increase the flow of services beyond their boundaries (whereas some clearly can [*34, 37*]), and that all of a biome's services not included in the Costanza *et al.* survey (*3*) are worthless. On the reserve costs side, we assume that management costs do not decrease once local communities' private opportunity costs are met, and that expanding reserve systems yield no cost savings through economies of scale or dissemination of best practice. Because these assumptions are biased against conservation, we consider our 100 : 1 ratio as a low estimate of the likely benefits of effective conservation.

Development and wild nature

In advocating greatly increased funding for the maintenance of natural ecosystems, we are not arguing against development. Given forecast increases in the human population of over three billion by 2050 (*38*) and the fact that some 1.2 billion people still live on less than a dollar a day (*39*), development is clearly essential. However, current development trajectories are self-evidently not delivering human benefits in the way that they should: income disparity worldwide is increasing and most countries are not on track to meet the United Nation's goals for human development and poverty eradication by 2015 (*39*). Our findings show one compelling reason why this is the case – our relentless conversion and degradation of remaining natural habitats is eroding overall human welfare for short-term private gain. In these circumstances, retaining as much as possible of what remains of wild nature through a judicious combination of sustainable use, conservation, and, where necessary, compensation for resulting opportunity costs (as called for at Rio [*40*]) makes overwhelming economic as well as moral sense.

References and Notes

1. By "wild nature" we mean habitat in which biodiversity, non-biotic components and ecosystem functioning are sufficiently intact that the majority of ecosystem services typically derived from such a habitat are still being sustainably and reliably supplied. Our usage differs from other usages, such as those adopted in cultural or anthropological studies. Because our focus is on wild nature, we excluded the Cropland and Urban biomes when using data from Table 2 of (*3*).
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41. This review is the result of a workshop convened by the Royal Society for the Protection of Birds and sponsored by the RSPB and the UK Government's Department for Environment, Food and Rural Affairs. We thank Neal Hockley, Pippa Gravestock, Jörn Scharlemann and Catherine Tiley for help with research, and Mark Avery, Richard Cowling, Gretchen Daily, Alistair Gammell, David Gibbons, Jeffrey McNeely, and Callum Roberts for stimulating discussions.

Figure legends

Fig. 1. The marginal benefits of retaining and converting natural habitats, expressed as Net Present Values (in 2000 US \$ ha^{-1}) calculated using the discount rates and time horizons presented. Values of measured goods and services delivered when habitats are relatively intact and when converted are plotted as black and white columns, respectively. (From [*11*-*15*]; see [*10*] for further details.)

Fig. 2. Recent estimates of the annual rate of change in the area or abundance of vertebrate populations for six biomes. Note that the biomes that have declined deliver very valuable ecosystem services (*3*). *Values plotted are the mean of habitat and population-based estimates; † little confidence can be attached to this value (*10*).

Online material

Case studies

All values were converted to 2000 US $ using a GDP deflator index (*S1*). Where case studies gave ranges of values, we took midpoints. All figures were taken directly from the sources cited, except for the coral reef example, where the time schedule for yields from destructive fishing was estimated from Figure 2 of ref. [*S2*] as 36 tonnes km^{-2} in year 1, and then 3 tonnes km^{-2}, rising to 5 tonnes km^{-2} by year 10.

Rates of loss

We searched the published literature and available databases for global estimates of recent trends in the area of largely unmodified habitats in all the relevant biome categories of Costanza *et al.* (*S3*) except rock and ice and open ocean. We supplemented this with biome-specific indices based on time series data on populations of wild vertebrates, derived from the WWF 2000 Living Planet Index (LPI) and FAO fisheries data (*S4*, *S5*). Unless otherwise stated, annual percentage rates of change in area or index value were calculated by taking the values a_1, a_2 in the first year t_1 and the last year t_2 of the series under consideration and calculating $100*(1 - (a_2 / a_1)^{(1/(t2 - t1))})$.

Tropical forests: We used the estimate in the FAO Forest Resources Assessment 2000 (*S6*) of a global net change of -7% in the area of tropical forest for the period 1990-2000, yielding an annual decline of 0.8%, although we are aware that some authorities consider this an underestimate. The LPI Index (*S4*) for tropical forest vertebrates showed a decrease of 26% between 1970 and 1999, yielding an average annual decline of 1.1%.

Temperate and boreal forests: The FAO Forest Resources Assessment estimates that temperate and tropical forests have increased in extent by 1% during the period 1990-2000, yielding an annual increase of 0.1% (*S6*). The LPI Index (*S4*) for temperate forest vertebrates showed a change of +4% between 1970 and 1999, yielding a small annual increase of 0.1%.

Mangroves: Valiela *et al.*(*S7*) estimated on the basis of a comprehensive assessment of mangrove resources that at least 35% of the global area of mangrove forests has been lost in the past two decades. Their data yield an annual decline of at least 2.5%.

Swamps, floodplains, lakes and rivers: There are no global estimates for rates of change in the extent of these habitats or for overall changes in their condition. The WWF LPI (***S4***) for inland water vertebrates showed a decline of 51% between 1970 and 1999, yielding an average annual decline of 2.4%.

Grasslands, rangelands, deserts and tundra: There are no global estimates for rates of change in the extent of these habitats or for overall changes in their condition. The available data for vertebrate populations are currently inadequate to allow development of a reliable LPI for any of these biomes.

Coral reefs: Although Bryant *et al.* (*S8*) report that around one quarter of the world's reefs are believed to be at high risk of degradation, there are no reliable global estimates for the rate at which coral reefs are actually being lost or degraded.

Seagrass and algal beds: There are no global estimates for the extent of algal beds, nor for rates of change in extent. No comprehensive survey of seagrass beds has been carried out, although it has been estimated that there may be between 500,000 and 1,000,000 km^2 in total (M. Spalding pers. comm.). Short and Wyllie-Echeverria (*S9*) stated that perhaps 900 km^2 of seagrass beds had been lost globally between 1985 and 1995, although the basis for this is not clear. Extrapolation would give an annual decline of 0.01-0.02% although little confidence can be attached to this figure.

Estuaries: We found no global assessment of rates of loss or degradation of estuarine habitats.

Coastal shelf: The only measure of coastal shelf habitat modification for which we found global estimates was disturbance of the sea-floor by bottom trawling. However, it is not clear what proportion of bottom trawling caused long-term habitat degradation, so we have not used this estimate.

Marine: The marine component of the WWF LPI (*S4*) does not distinguish between different marine biomes. Overall it indicates a 36% decline in abundance of marine fish, mammals, birds and reptiles over the period 1970-1999, yielding an average annual decline of 1.5%. Further evidence for decline is provided by fitting a curve to FAO data (*S5*) on changes since 1974 in the proportion of all the world's marine fish stocks that are exploitable (i.e. categorised as fully, moderately or under-exploited). Most fish stocks reduced to unexploitable levels show little evidence of recovery within 15 years of their decline (*S10*) and so can be regarded as effectively lost to exploitation for the foreseeable future. The fitted curve suggests that exploitable fish stocks have effectively been "lost" at the rate of 1.5% per year.

Thus for each of three biomes we have two estimates derived by different methods and either independent data (tropical forest and temperate/boreal forest) or largely independent data (marine LPI and fish stocks). In all cases the two estimates were remarkably similar. The rates of change were therefore averaged for these biomes to yield a single estimate. Five of the six global biome-specific estimates of change in habitat area or population show declines, which are distributed about a mean of 1.2% per year (SE = 0.5%; n = 6).

Costs of losses

If the Costanza *et al.* aggregate figures (*S3*) for largely natural biomes and our estimate of the proportion of TEV lost upon conversion are roughly correct, then a single year's average losses in the 1990s cost society approximately \$37.6 trillion x 54.9% x 1.2% ≈ \$250 billion every year into the future.

Costs of conservation

The hypothetical terrestrial reserve network would cover ~15% of each region (*S11*, *S12*). The costs include resources needed for the effective management of existing and new reserves; the costs of adequately compensating local residents in developing countries for the unmet private opportunity costs of existing, strictly protected reserves (spread over 10 years); the costs of surveying and then leasing or acquiring new reserves (spread over 30 years); and the private opportunity costs of greening forestry or farming in buffer zones around the margin of reserves, covering an additional 1.5% of each region's total area.

The cost of the hypothetical marine network was derived from a survey of current and unmet expenditure for 71 Marine Protected Areas (MPAs; A. Balmford, P. Gravestock and C. Roberts, unpubl. data). Total management costs of MPAs can be predicted from their size (regression gives $\log_{10}$[annual cost, in 2000 US $] = 5.00 + 0.20 [$\log_{10}$(area, in km^2)], with $r^2 = 0.79$). The management costs of the hypothetical global network were then estimated by combining this relationship with the log-normal size distribution for 991 existing reserves (*S13*), which together cover 0.50% of the seas, and assuming 30% coverage is achieved by simple replication of this current network (note that this will overestimate total costs because plausible spatial patterns of network expansion inevitably lead to reserve merging and hence economies of scale). One-off set-up costs of MPAs were estimated at 7.3 times annual management costs (from n=4 reserves, including [*S2*]), and were spread evenly over a 30-year implementation period.

Total costs for both the terrestrial and marine reserve system varied through the implementation period, from $32 to $54 billion per year, with a mean of $45 billion.

Benefits of conservation

If Costanza *et al.*'s (*S3*) per hectare values of ecosystem services and our 54.9% estimate for the relative loss of TEV upon conversion are approximately correct, the proposed reserve network would safeguard annual flows worth $23.8 trillion x 54.9% x 30% ≈ $3900 billion at sea and (because a network covering 15% of land area would cover ~16.9% of the largely natural biomes [*S3*]) $13.8 trillion x 54.9% x 16.9% ≈ $1300 billion on land, or ~$5200 billion in total. However, under strict protection, those flows accruing from resource extraction would not be available. Remaining services constitute ~91.4% of all services by value, according to Table 2 of ref. (*S3*) (conservatively assuming all recreation is incompatible with strict protection). Hence a strict reserve network would safeguard annual flows with a net worth of ($23.8 trillion x 91.4% x 30%) – ($23.8 trillion x 45.1% x 30%) ≈ $3300 billion at sea, and ($13.8 trillion x 91.4% x 16.9%) – ($13.8 trillion x 45.1% x 16.9%) ≈ $1100 billion on land, or ~$4400 billion in total.

Sensitivity analyses

Our qualitative conclusions remained robust to varying discount rates in the case studies between 3% and 10%, and to excluding tourism and live fishing benefits in the reef example. The ~ 100 : 1 ratio was also robust when (because it was not addressed in the case studies) we excluded all benefits and costs from open oceans, and decreased only as low as ~ 40 : 1 even when we made the unlikely assumption that nutrient cycling (the largest service not examined in the case studies) differed from all measured services in being delivered equally by intact and converted biomes.

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