THE ECONOMICS OF ECOSYSTEMS AND BIODIVERSITY: SCOPING THE SCIENCE

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FINAL REPORT

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1 INTRODUCTION

1.1 Background

The recently released Stern Review on the Economics of Climate Change (Stern 2007) concluded that it makes economic sense to invest in mitigating greenhouse emissions to avoid the worst effects of climate change rather than face the consequences of failing to do so. This proved highly influential in persuading politicians and economists of the need to act on climate change now.

The G8 decided in March 2007 to initiate an analogous “Review on the economics of biodiversity loss”, in the so called Potsdam Initiative: *'In a global study we will initiate the process of analysing the global economic benefit of biological diversity, the costs of the loss of biodiversity and the failure to take protective measures versus the costs of effective conservation.'.* This proposal was endorsed by G8+5 leaders at the Heiligendamm Summit 6-8 June 2007.”

This study consists of two phases:

- Phase 1 is a preparatory stage including several studies to be ready for the CBD COP9 (Conference of the Parties to the Convention on Biological Diversity) in May.

- Phase 2 is the consolidation stage that will produce the full Review, to be ready in October 2009.

The study is being supported by the European Commission (together with the European Environmental Agency and in cooperation with the German Government), notably through contracted studies such as the present one. Leading economist Pavan Sukhdev as been appointed as an independent Review leader for Phase 2.

1.2 This Scoping the Science project

“The Economics of Ecosystems and Biodiversity: Scoping the Science” project (henceforth, the Scoping the Science project) is one of several Phase 1 projects, running from 14/12/2007 to 30/4/2008. According to the invitation to tender:

“*The objective of the current study is to provide a coherent overview of existing scientific knowledge upon which to base the economics of the Review, and to propose a coherent global programme of scientific work, both for Phase 2 (consolidation) and to enable more robust future iterations of the Review beyond 2010.*”

The specific tasks to be carried out in the context of the project are:

1. Elaborating the conceptual framework for the Review;
2. Establishing a working relationships with research networks and programmes;
3. Reviewing existing ecological knowledge needed to carry out the Review;
4. Identifying critical gaps in knowledge;

5. Presenting key findings;
6. Elaborating a proposal for a 1-2 year programme of work to provide case studies;
7. Elaborating a proposal for a 2-3 year programme of work to fill critical gaps;
8. Building an inventory of relevant research networks, programmes and projects; and
9. Providing an inventory of internet resources.

The outputs of this Scoping the Science project will provide background and recommendations for Phase 2, when the approach to be followed in the Review will be defined in detail.

1.3 This report

This report presents the final results and recommendations from the Scoping the Science project.

The Conceptual Framework for the Review on the Economics of Ecosystems and Biodiversity (Task 1) is detailed in Section 2. Note that this has a significantly broader scope than the remaining tasks in this project. The conceptual framework was developed for the whole Review, covering the full integration of ecological and economic sciences. This task was led by the University of Cambridge.

The bulk of this report covers the most substantial task of the project, of providing a coherent overview of existing scientific knowledge (Task 3). Given the specific mandate that such review should provide an adequate basis for the economic valuation, we ensured that the definition of the review themes selected was appropriately defined for this aim. Section 3 presents the conceptual background and justification for the selection of the set of themes that form the basis of the review of the ecological knowledge. Section 4 presents the results of the thematic reviews. Gaps in knowledge (Task 4) and key findings (Task 5) are discussed in the report for each theme. An extensive network of experts was created in this review (Task 2). This work was led by the University of Cambridge, with the support of UNEP-WCMC.

A synthesis of the recommended priorities for Phase 2 (Task 6) as well as in the longer term (Task 7) is presented in Section 4.19. This task was led by the University of Cambridge, with the support of UNEP-WCMC.

The project also built an inventory of relevant research networks, programmes and projects (Task 8), with the aim of expanding the relationships with these in the future (Task 2). To support this exercise, a review of the recent literature (i.e. publications post the 2005 Millennium Ecosystem Assessment) dealing with economics of biodiversity loss was conducted (Section 5; with the list of identified publications and networks presented in

Annex 4 and in Annex 5). This work was led by Alterra and the Wageningen University.

Finally, the project also provides a review of key internet resources on biodiversity and ecosystem services (Task 9). This task was lead by UNEP-WCMC and its outcomes can be found in Annex 6. This review was specifically focused on web-available material related to biodiversity research and ecology (i.e. as oppose to biodiversity economics). Thus, its outcomes largely complement the information on research initiatives / networks and literature on economics of biodiversity loss above.

2 A CONCEPTUAL FRAMEWORK FOR THE REVIEW

2.1 Why we need a conceptual framework

The scope of the Review on the Economics of Ecosystems and Biodiversity is extremely ambitious: *analysing the global economic benefit of biological diversity, the costs of the loss of biodiversity and the failure to take protective measures versus the costs of effective conservation* (Potsdam Initiative). The scope of the Review is also very wide, as the benefits that we obtain from biodiversity are extremely varied. Tackling this immense task will require building from the knowledge and the skills of an extensive network of research teams with very different expertises and backgrounds.

The conceptual framework is a proposed operational roadmap for Phase 2 of the Review on the Economics of Ecosystems and Biodiversity. It presents a strategy for partitioning the Review into well-defined and self-contained pieces that can be tackled by teams with the appropriate expertise. It then presents the method for linking these pieces together in order to provide a reliable answer to the Postsdam challenge.

The framework is particularly important for ensuring a smooth link between the ecological and the economic aspects of the Review.

The Review will need to face substantial gaps in knowledge and so the results will inevitably be plagued by uncertainty. Dividing the Review into modules allows for the identification of the areas where the main gaps are. Understanding the links between modules will indicate how uncertainty in a particular subject propagates within the review to affect its final results. This facilitates the identification of research priorities for better quantifying the links between biodiversity and economics.

This is a mechanistic framework, for understanding how changes in biodiversity *cause* changes in economic costs and benefits, rather than simply for analysing how changes in biodiversity *co-vary* with economic changes. That is, this framework will help to distinguish causality from simple correlation in the relationship between trends in biodiversity and in economic values.

2.2 Initial considerations and assumptions

2.2.1 Valuing wild nature

The term ‘biodiversity’ is often considered to refer to variability amongst organisms (e.g., species diversity, genetic diversity). Indeed, the Convention on Biological Diversity, defines ‘Biological diversity’ as “*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*”.

However, the economic importance of wild nature does not rely solely on <u>variability</u> (e.g. species and genetic diversity). Indeed, many of the benefits obtained from nature rely much more on <u>amount</u> (e.g., the abundance of particular species) (Figure 1).

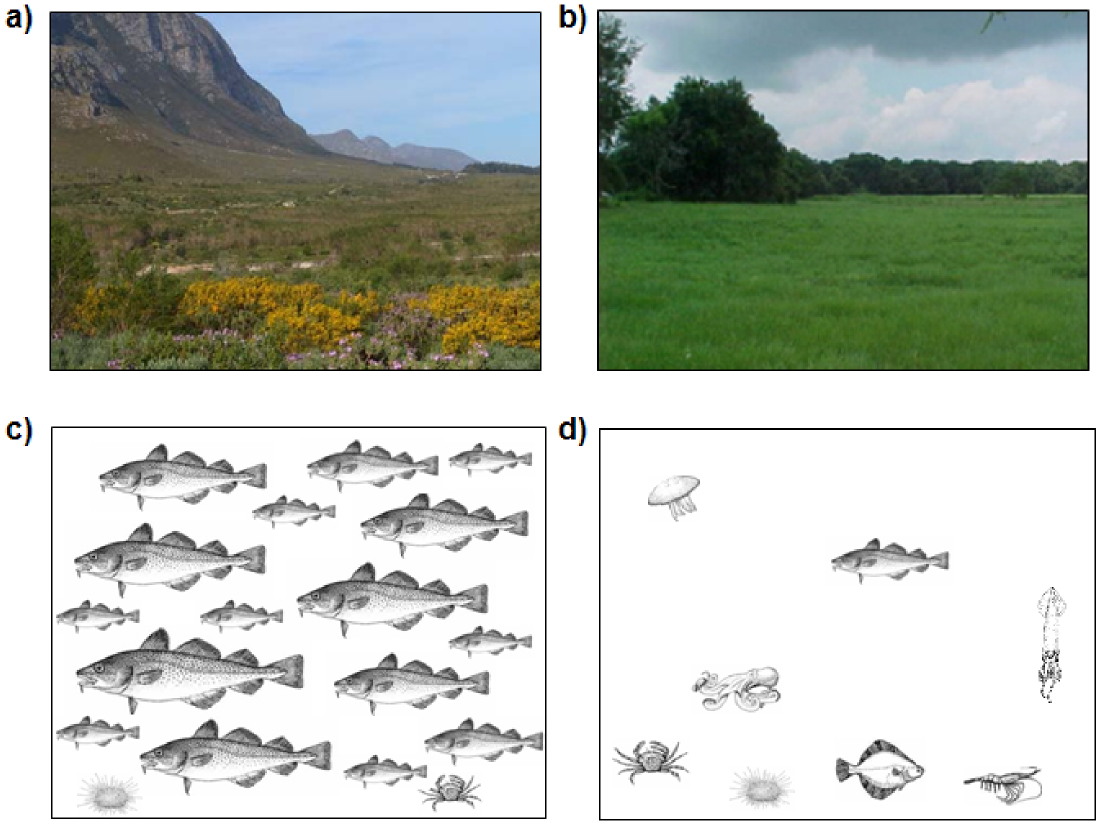


Figure 1. Illustration of the importance of *variability* and of *amount* for the provisioning of different benefits from nature. a) High variability, low biomass fynbos ecosystem. b) Low variability, high biomass improved pasture. Despite its lower biomass, the fynbos ecosystem has a higher value for the provisioning of benefits that depend on diversity, such as compounds for the pharmaceutical industry. c) High biomass, low variability cod-dominated marine system. d) Low biomass, high variability marine system after overexploitation of cod. Despite its lower variability, the cod-dominated system has a higher value for the provisioning of food, a benefit that is highly sensitive to amount (biomass).

Furthermore, the provision of benefits often depends on the *condition and extent of ecosystems* – incorporating many species and their interactions amongst them and with their environment. Ocean fisheries provision, for example, is affected by the condition of coral reefs and mangroves.

It is therefore proposed that the Review on the Economics of Ecosystems and Biodiversity should aim to address not simply the effects of the loss in variability, but also the effects of changes in amount, and in the condition and extent of ecosystems. To avoid confusion, throughout we use the expression *wild nature* to refer to this broader definition of biodiversity.

By ‘wild’ we do not mean ‘pristine’ (in which case it would apply to very few parts of the world); we mean ‘non-domesticated’ (i.e. excluding for example livestock and crops).

There are many ways in which wild nature contributes to human wellbeing. These have frequently been called ‘ecosystem services’, particularly in the context of the Millennium Ecosystem Assessment (MEA 2005). This term includes both ecological processes (e.g. nutrient cycling, water regulation, pollination) and benefits (e.g. food, water, medicinal products). As discussed in Section 3, benefits are the end products of these ecosystem processes, which directly affect human wellbeing, and which can ultimately be evaluated economically (e.g., clean drinking water). We we focus this Review as much as possible on such benefits, because these can be directly valued. In contrast, the value of ecological processes (e.g., pollination, water regulation) can only be established by valuing their contribution to different benefits (e.g., food crops, drinking water). However, this has not always been possible, and so throughout we refer to benefits or (beneficial) processes as the ways in which wild nature contributes to human wellbeing. This terminology, and how it relates to the framework of the Millennium Ecosystem Assessment, is fully explained in Section 3.

2.2.2 Scope of the Review

The aim of this Review is to evaluate the economic consequences of biodiversity (wild nature) loss. It does not address non-economic values of nature, such as those stemming from what many consider the moral right of all species to exist, irrespective of people.

Similarly, ecosystem functions (e.g., nutrient cycles, species interactions) are not valued in their own right (or because they are important for other species), but specifically for the direct or indirect benefits they provide to humans. For the purposes of this Review, only changes in wild nature that have economic consequences are considered.

Hence, the Review – being an economic analysis – will only be able to cover a fraction of the value of wild nature. There are limits to the meaningful valuation (in monetary terms) of nature and its total complement of services and consequent benefits to humans (Turner 1999; Turner et al. 2003). While most use values can be adequately captured with the economic calculus, controversy remains over the evaluation of so-called non-use values (Pearce & Turner 1990). For some analysts, intrinsic values in nature cannot be ‘captured’ within economic analysis (Pearce & Moran 1994) but much depends on the precise definition of intrinsic value. If the interpretation is one involving the assignment of intrinsic values by a human to nature (or to its components) then techniques such as contingent valuation and choice experiments may be able to shed light on the motivations and preference values involved (known as ‘existence’ value). On the other hand, if the interpretation is one in which no human agent is involved then monetary valuation is replaced by moral imperatives (often linked to non-human species rights and/or interests) (Norton 1992). A further complication arises if one accepts that a certain minimum provision of ‘healthy’ functioning ecosystems is essential to ensure a sustainable flow of services and benefits to avoid threshold effects and system collapse. There is then a ‘glue value’, ‘insurance value’, or ‘infrastructure value’ bound up with this (often unknown) minimum ecosystem provision. Total ecosystem value will therefore always be greater than total economic value (Turner 1991).

This document does not reflect our personal conviction, shared by many, that wild nature has an intrinsic value which on its own justifies efforts to conserve it. However, an evaluation of the economic value of wild nature is not incompatible with this conviction. Indeed, if the results of such evaluation are that conservation results in a net economic gain, then that simply adds an economic argument against wild nature loss, alongside the moral argument. If the results are that conservation of wild nature incurs a net economic loss, then that will provide the net size of the bill for conserving wild nature.

Note that 'economic' does not mean simply monetary revenue. It also includes aspects such as changes in livelihood conditions, economic structure, investment risk and social aspects such as poverty, inequality in access, and benefit sharing.

2.2.3 Valuing marginal change

This Review is not about the economic value of wild nature as a whole. In one sense, that calculation is trivial: biodiversity has an infinite value because no human life is possible without it (Toman 1998). This Review is instead about the economic consequences of a *marginal* loss of wild nature, which are substantially lower than the total stock of nature. Indeed, even within the current biodiversity crisis we are (fortunately) very far from erasing all life and the benefits we derive from it. Even substantial transformations in natural habitats may result in only relatively small changes in the provision of some benefits (e.g. water regulation benefits can be retained in the absence of forest cover through appropriate soil conservation practices; Bruijnzeel 2004) and in some cases changes in wild nature may actually improve the provision of benefits (e.g. it is possible that wild meat production is higher in some secondary habitats than in primary forest; Robinson & Bennett 2004).

It is also fundamental to understand that for some benefits the value of nature is to *contribute* to the provision of a benefit, rather than to provide the entire benefit. For example, a degree of water purification would take place even in the absence of life, as water can be filtered by passing through soil and rock. It would therefore be incorrect to attribute the value of all water purification in a watershed to its natural vegetation. What is relevant for the purposes of this Review is the *added* (again, marginal) value of natural vegetation to water purification.

The concept of marginality is key for making ecosystem service research policy relevant because it is at the margin that policy and economic decisions operate (Turner et al. 1998). Indeed, it would not be useful, for example, to calculate the total value of the global forests as a tool for informing practical forest policy (Bockstael et al. 2000; Fisher et al. *in press*-b). Having said this, the term 'marginal' is usually used in the economics literature to refer to a very small change. Here we refer to changes that while small in relative terms (compared with the full stock of wild nature) may be quite substantial in absolute terms (e.g., thousands of hectares of change in land use). This creates substantial challenges to economic valuation, particularly if the system function is subject to abrupt, non-linear changes in function (e.g. by 'flipping' from one equilibrium state to another). Incorporating marginality in ecosystem service evaluation requires therefore a good understanding of the drivers and pressures on the systems under study, as well as of how the system is changing or might change from its current state into a different state under a particular policy action (Fisher et al. *in press*-b). Here, we use the term 'marginal' to refer to changes that a single policy level can foresee. Hence, the valuation of marginal changes is anchored into specific 'states of the world' generated from counterfactual scenarios where a specific policy action is either adopted or not (see below, and Section 2.3).

The overall economic consequences of a marginal loss of wild nature can be evaluated by investigating the economic consequences of adopting a specific set of actions that can prevent that specific level of marginal loss. Such evaluation requires comparing different 'states of the world' – one with more, and one with less wild nature.

The 'state' of a given area (e.g.: a 1 km^2 plot in Europe; the entire world), is a particular set of biotic and abiotic conditions for that area, comprising aspects such as: land cover, climate, human distribution, human activities, and conservation actions in place. There is a current state of the planet, a different state existed 20 years ago, and there will be a different state in the future.

Scenarios can be used to conceptualise the state of the planet in the future (e.g., by 2010 or by 2050) or what it would have been today if decisions in the past had been different. Scenarios are the tools (e.g., models, storylines) used to imagine states of the world. Conversely, states of the world are the outputs of particular scenarios.

As discussed in Section 2.3.2, measuring the consequences of a marginal change in wild nature requires contrasting very specific states of the world (counterfactuals), in which everything else is equal except for the implementation or not of a set of actions aimed at reducing losses in wild nature. This has the advantage that it requires being explicit about what the conservation goals are, and what the actions are required to achieve those goals.

2.2.4 The need to be spatially explicit

A key characteristic of this framework is that it is spatially explicit. We think this is essential, for three sets of reasons:

1. The production, use and flow of benefits from wild nature vary spatially, and so it matters to human wellbeing where conservation actions are implemented.

2. A spatially-explicit framework requires stating the assumptions being made when extrapolating across heterogeneous landscapes using limited data.

3. A spatially explicit quantification of benefits and costs allows makes explicit the mismatch between winners and losers in different scenarios, and is thus essential for designing effective and equitable policy interventions.

We develop these points in detail in Section 2.3.3.

2.3 The conceptual framework

The proposed conceptual framework integrates ecological and economic knowledge to evaluate the net socio-economic consequences of policy actions for halting/reducing wild nature loss (Figure 2). This framework is therefore a practical tool for evaluating the effectiveness of different policy actions.

The next section details, step-by-step, how the framework works.

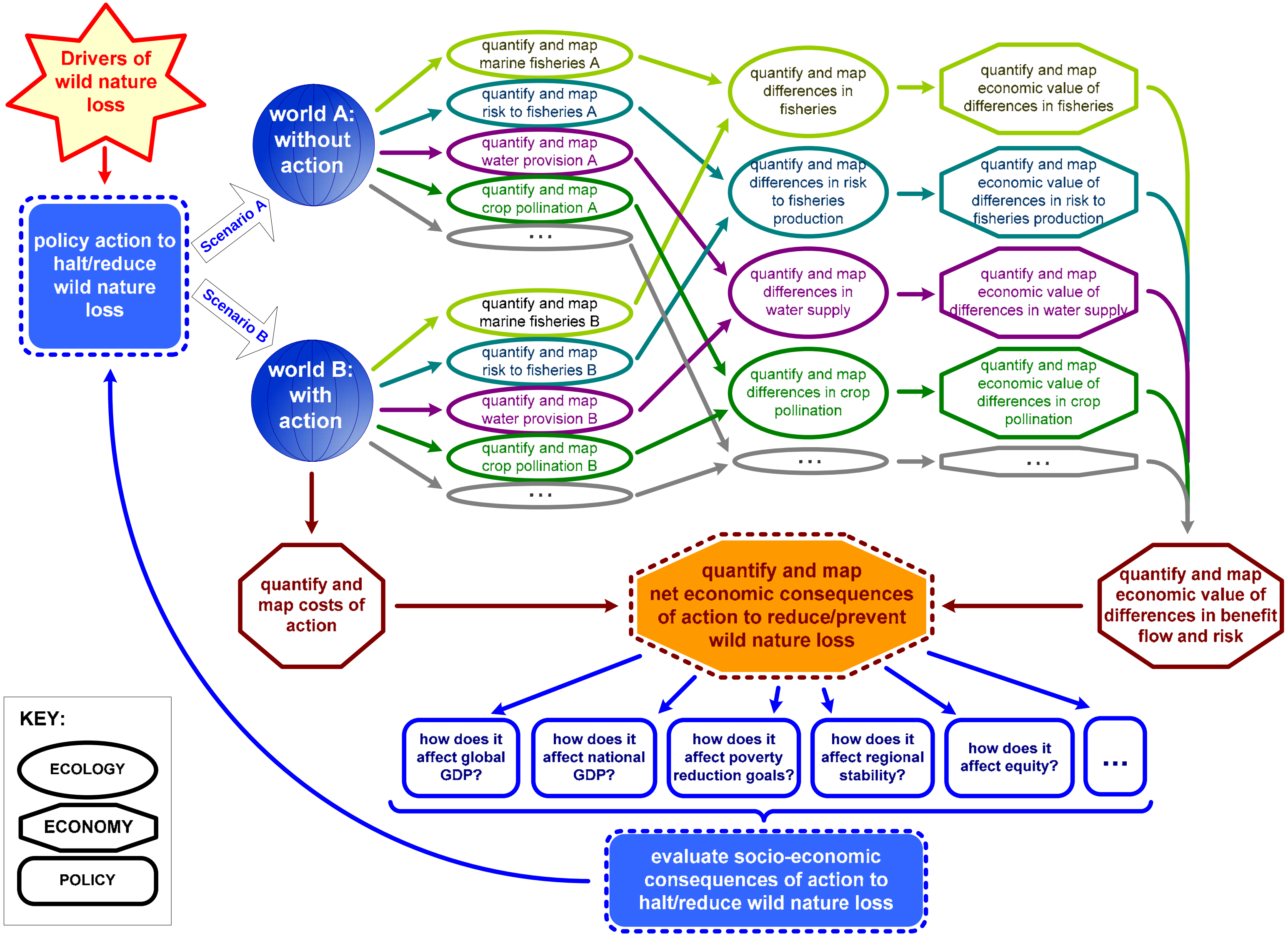
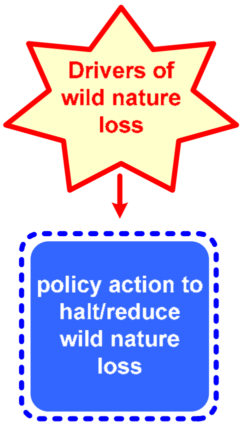


Figure 2. A conceptual framework for evaluating the ecological and economic consequences of policy actions for halting/reducing wild nature loss.

Throughout we refer to a 'policy action' but the framework can be used to test packages of policy actions, as long as adequate counterfactual states of the world are created accordingly.

The term 'wild nature loss' is used throughout to refer to losses in biodiversity (both variety and amount) and ecosystem degradation (Section 2.2.1).

2.3.1 Defining appropriate policy actions based on the drivers of loss



Biodiversity is being lost and ecosystems are being degraded through a diversity of drivers, including habitat loss and degradation, overexploitation, species invasions, and climate change (Baillie et al. 2004).

The starting point of the framework needs to be a good understanding of these drivers. This is crucial to desinging and costing effective policy actions for reducing/halting biodiversity loss and ecosystem degradation (losses in wild nature).

For example, overexploitation is the main driver of losses in the provision of marine fisheries (Pauly et al. 2005). Appropriate policy actions for reducing/halting the decline in the provision of benefits from marine fisheries address this driver directly by regulating fishing effort. This may include:

- Regulation of the temporal distribution of fishing effort by setting fishing seasons that minimise impacts on the fish stocks (e.g., avoiding the reproductive season).

- Regulation of the spatial distribution of fishing effort (e.g. creating marine protected areas with no-take zones, banning of bottom dredging practices below 1000 m).

- Regulation of overall fishing effort, by setting fishing quotas.

- Regulation of fishing targets and impacts, by defining which fishing gear can be used.

The first, and crucial, step in the framework is to make explicit what the overall conservation goal is. For example, it may be "to prevent any additional fishing stock from becoming depleted and to ensure the recovery of at least 30% of currently depleted socks by 2030". This step is crucial because this is where it is defined what specific *marginal* change in wild nature is being considered in this evaluation (see Section 2.2.3). The economic consequences (both costs and benefits) of changes in wild nature (biodiversity loss and ecosystem degradation) are completely determined by this definition.

Having defined this goal, the next step is the identification of an appropriate bundle of policy actions, explicitly enough to be costed, that are likely to deliver the goal. For example, these may include actions such as:

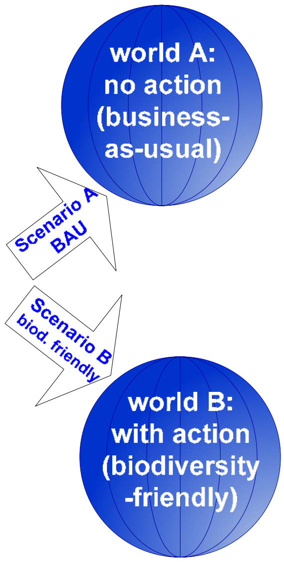
- Decommissing 80% of the tonnage in fishery X;

- Set aside 10% of region Y as reserve networks;

- Replace current fishing gear W with new gear Z.

Naturally, the definition of what the appropriate actions are needs to be done in close collaboration with experts in the field.

2.3.2 Defining the counterfactual states of the world



Evaluating the effectiveness of a particular policy action (or bundle of actions) requires comparing two hypothetical states of the world:

- World A (business-as-usual), where the action is not put in place.

- World B (biodiversity-friendly) where the action is implemented.

The states of the world are obtained through appropriate *scenarios*. The business-as-usual scenario generates predictions for the plausible state of the world in the absence of the specific intervention being tested (e.g., likely situation of the fishing stocks and the fishing industry by 2030 given current predictions in population growth, in demand for fish, in climate change, etc.). The biodiversity-friendly scenario needs to be identical in everything else to the business-as-usual scenario except but for the specific policy action being tested (e.g. likely situation of the fishing stocks and the fishing industry by 2030 *if the policy actions are implemented*, given current predictions in population growth, in demand for fish, in climate change, etc.).

The scenarios (and the states of the world produced) need to have the right level of information at the right spatial scale. For example, if the action is to implement new protected areas, scenario A needs to state where and roughly how large they are; if the action is to manage fisheries appropriately, scenario A needs to spell out how that would be done, for example by creating no-fishing zones and setting sustainable fisheries quotas. If the scenarios do not have the necessary detail, they will not provide clear answers about how plausible changes in wild nature affect the benefits we derive from it, and they cannot be costed appropriately.

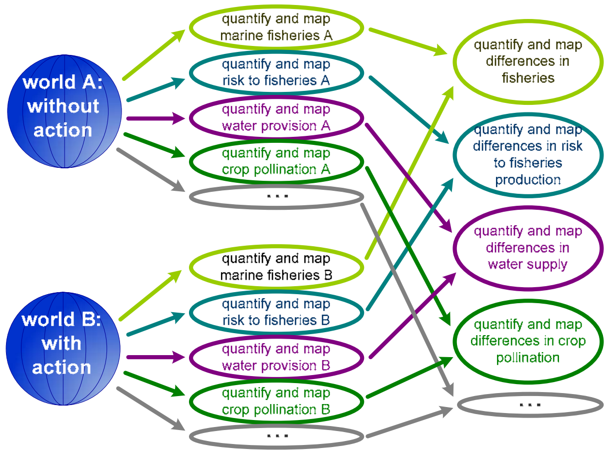
While we refer throughout to 'states of the world', this framework can be applied to evaluate policy actions at any scale, not just global. For example, it may be applied to evaluate the consequences of implementing a particular protected area or a new dam. The methods for comparing states with and without particular interventions are well-established in the field of Environmental Impact Assessment, being for example a requirement in the EU Directive on Environmental Impact Assessment.

Last, while we refer throughout to "two scenarios", many scenarios can be developed and contrasted, although each contrast needs to be between two appropriately matched counterfactuals.

The risks of contrasting inappropriate states

It is fundamental that the scenarios are identical in everything else but the specific policy action being tested. Otherwise, the economic results cannot be directly attributed to a difference in the state of wild nature and therefore are not a measure of the economic consequences of biodiversity loss and ecosystem degradation. For example, a comparison between the state of the world today and a predicted state by 2030 is not appropriate for understanding the net economic consequences of changes in wild nature over that period, because many other conditions would be changing at the same time (e.g. population growth, climate change, technology).

2.3.3 Quantifying and mapping how the biophysical provision of benefits is affected by the policy action



This is the focus of Task 3 in this Scoping the Science project (Sections 3 and 4 of this report).

The first step is to define what are the benefits or beneficial processes that will be evaluated - i.e. the different ways in which wild nature contributes to human wellbeing. These need to be defined carefully to avoid double-counting, which would compromise the results and the credibility of the overall evaluation. Section 3 of this report presents in detail our proposal for a classification of the links between nature and human wellbeing that avoid the double-counting problem and so provides a sound basis for economic valuation.

For each particular benefit (e.g. fisheries) or beneficial process (e.g. water regulation; see Section 3) one needs to understand how its provision is affected by the policy action (i.e. what are the marginal benefits of the action). In order to do that, we need to be able to quantify the predicted provision for each of the states of the world considered. For example, we need to predict fisheries production under each scenario; these can then be compared to understand how the policy action is likely to affect fisheries.

The importance of being spatially-explicit

It is crucial that the contrast between benefit provision under different scenarios is done not simply in terms of the overall global value, but by attempting to quantify and map the spatial variation in the provision of the benefit or beneficial process. Indeed, a key characteristic of this framework is that it is spatially-explicit. This is essential, for three sets of reasons:

i. The *production* of benefits or beneficial processes varies spatially, because it is based on spatially-variable underlying ecosystem processes. For example, fisheries production will depend on ocean productivity (Section 4.6), while carbon storage (contributing to carbon regulation; Section 4.16) depends on the vegetation cover (Figure 4). The *flow* of benefits from the point of production (e.g., an upstream forest) to the point of use (e.g., a downstream city) is highly influenced by spatial constraints. The *use* of these benefits is spatially heterogeneous too, depending critically on the patterns of distribution of end users (e.g., cities, or agricultural areas), which creates substantial spatial variability in the value of benefits even within areas with similar natural production and flow. Finally, alternative states of the world capturing different political interventions need to be spatially explicit, as the results of such interventions depend critically on where they are implemented (e.g., the location of no-take marine areas). Integrating all these components (production, flow, use, and state) clearly therefore requires a spatially explicit framework.

ii. Frequently, available information will be insufficient for the elaboration of detailed maps of particular elements of the framework (e.g., value of wetlands for storm protection). This does not render the framework less useful: quite the opposite. Having to describe each element spatially requires stating the assumptions being made when extrapolating from limited data. If a single data point is all that is available, then the corresponding world map may have a single colour. Oversimplifications become obvious, and so are more likely to receive attention in future work. Further information will help to refine the spatial representations (e.g., if other data indicates that value of wetlands for storm protection is likely to vary with Gross Domestic Product/km^2). A spatially-explicit framework can always be aggregated into coarser spatial units, for example to provide values per biome, or global values. The reverse, however, is not possible.

iii. Services often flow from their point of production to users elsewhere (Figure 5) – so the benefits of their conservation may accrue to different actors than the benefits associated with their loss (Figure 6). For any given action there will likely be winners (e.g., areas where fisheries production increases) and losers (areas where production decreases). Information on these would be lost from the global aggregated value, but is fundamental to an appropriate economic valuation (as in most cases benefit value is context-dependent). A spatially-explicit approach is therefore crucial to fully evaluate the broader social consequences of each action in terms of impacts on livelihoods, development goals, and equity, and for designing effective and equitable policy interventions.

Quantifying and mapping the production of benefits/processes

Obtaining global, or at least large-scale, maps of the production of a particular benefit (or beneficial process) requires a good understanding the factors that drive production. These factors will include a mix of abiotic (e.g., climate, soil type, topography) and biotic (e.g. ecosystem type, species diversity) variables, as well as factors that are determined by human actions (e.g. extent of habitat). If these relationships are well understood, it should be possible to generate a mathematical model that can reasonably predict benefit production given information on the underlying factors. Such a model corresponds to what in economic terms is called a *production function*. At this stage, however, we are interested in modelling the biophysical production of benefits (or beneficial processes) rather than their economic value.

For example, timber production from natural forests per area, per year, may be given as the forested area multiplied by a function (*f*) of factors such as primary productivity, forest type, topography, etc. (Figure 3).

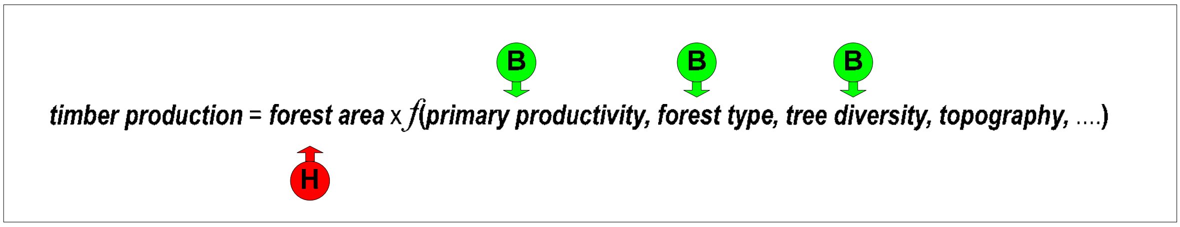


Figure 3. Hypothetical production function for timber form natural forests. Green arrows correspond to biodiversity/ecosystem inputs; red arrows correspond to human inputs.

Note that for many of these benefits their production depends not only on wild nature, but also on abiotic (e.g. topography) and anthropogenic inputs (e.g. forest area). The main aim of this Scoping the Science project is to understand the marginal (or added) value of wild nature for the production of these benefits.

Given such a production function and appropriate biophysical data (maps of primary productivity, of forest types, of tree richness, topography) it would be possible to create a global surface of potential timber production for each state of the world. From one state to the other, some key variables would change (e.g., forest conversion), therefore resulting in spatial differences in timber production. A contrast between these maps would then be used to quantify, and subsequently value, the spatial variation in differences in timber production between states of the world.

The production map should be expressed in the appropriate physical units for measuring the provisioning of the service in question (e.g., tons of carbon/km^2 for carbon storage, m^3 of water/km^2/y for water provisioning, numbers of visitors/km^2/y for recreational services).

In practice, a detailed model of a production function will seldom exist at a global scale. However, more simplified models do exist in some cases, or they can be produced from available data. For example, a global model of wild meat production (Section 4.8) has not been created yet, but a reasonable number of individual studies have calculated productivity for different areas around the world, and we believe that a first-cut global model could be produced in time for Phase 2 (Section 4.8). The key question we addressed in each of the thematic reviews (Section 4) was how far is scientific knowledge from being able to create such models. In a few cases, maps of benefit/process production have already been developed (e.g. Figure 4).

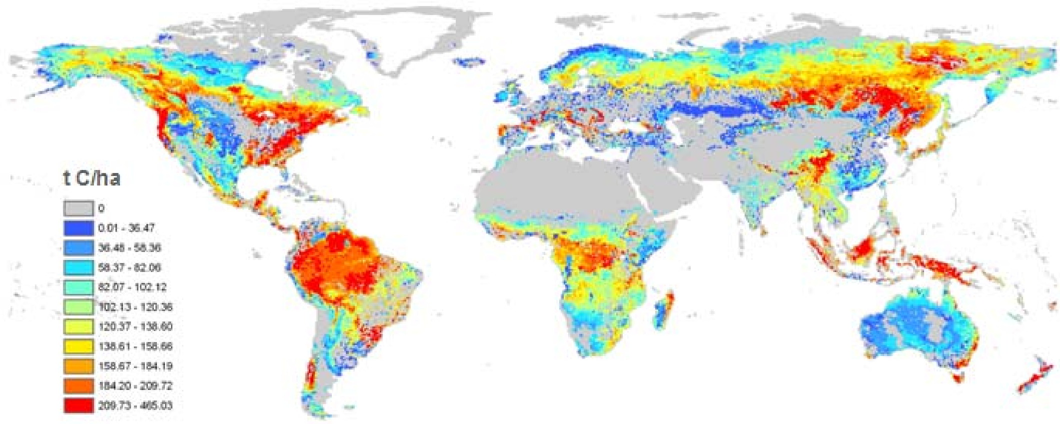


Figure 4. A preliminary map of the global distribution of current ecosystem productivity for the service 'carbon storage' (Naidoo et al. *in press*).

Mapping (potential) sustainable production, rather than current use

For some benefits (those that involve harvesting, such as fisheries, timber production, wild meat production) there is a potential for overexploitation leading to depletion. In this case, the relevant production map needs to correspond to a map of *sustainable* productivity, rather than current benefit flow. For example, when considering food production from fisheries, the relevant map is one of sustainable fisheries production, rather than one of current catches. Indeed, the latter is likely to reflect excessive fishing effort in some areas, which over the long-run will lead to a reduction in benefits.

On the other hand, a map of current benefit flow would ignore areas where there is no present use but which may be valuable nonentheless. For example, areas too remote to contribute for current timber production may be considered to hold value for future timber (option value, Section 3.7).

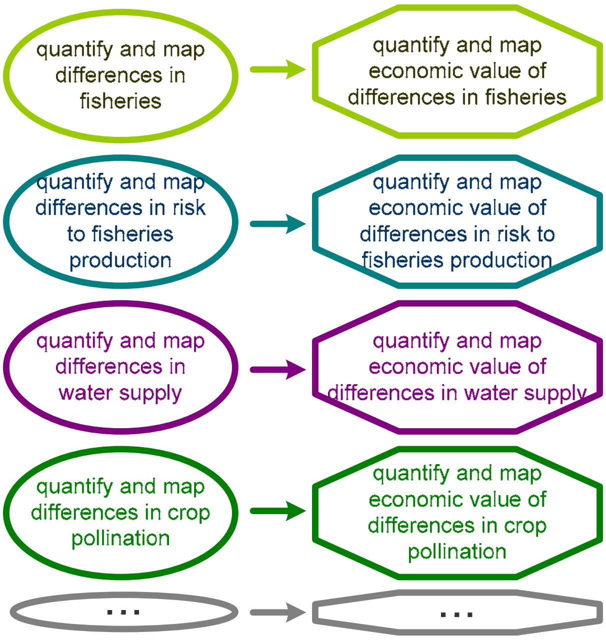
Quantifying and mapping risk

Biodiversity loss and ecosystem degradation affect not only the flows of services and benefits, but also the resilience of systems. It is therefore crucial to attempt to quantify the extent to which a particular action (or the lack of its implementation) affects the likelihood that each service or benefit will be compromised. Knowledge on resilience and risk is even scarcer than on benefit/process production, but whenever available is should be incorporated in the economic valuation. In the thematic reviews (Section 4), we collated information on the possibility of non-linearities in the production of each benefit/process, and particularly the possibility of threshold effects in which a small change in the state of wild nature could result in a disproportionate change in benefit/process provisioning.

For example, there is now some information on the conditions that trigger collapse in marine fisheries (Section 4.6), and so it may be possible to obtain at least some indication of the extent to which fisheries are at risk of collapse under each scenario. Again, whenever possible, this information should be quantified in a spatially-explicit way, even if the mapping units are likely to be quite coarse. For example, it will clearly not be possible to map risk of fisheries collapse at a fine scale, but expert opinion could probably provide at least a qualitative assessment of the risk of collapse for each of the FAO major fishing areas.

Key here is considering the sensitivity of benefit/process productivity to likely changes in biodiversity. It may not be vital (at this stage) to know, for instance, that reducing forest cover from 2% to 1% of the land surface will probably trigger a collapse in timber productivity if the relevant difference in states of the world (under policy action vs. inaction) is 25% vs. 15% forest cover.

2.3.4 Quantifying and mapping the economic value of changes in benefits derived from the policy action



The information on how the biophysical production of each benefit/process changes under each scenario becomes the basis of the economic valuation, using appropriate tools.

It is beyond the remit of this project to discuss in detail how such valuation can be made, but we present here some considerations that will affect the way the valuation is done.

Different types of values

Different benefits (or beneficial processes) correspond to different types of economic values, which affects the methods used to quantify their economic value. For example, benefits obtained through direct, consumptive use (e.g., fisheries, wild meat, medicinal plants) can be valued using market prices directly or through replacement costs; non-consumptive use benefits such as nature tourism can be valued using travel costs; and hedonic price methods can contribute to the evaluation of indirect use values such as water purification (Turner et al. 2003).

In Section 3.7 we discuss how the benefits/processes proposed for the Review relate to the framework of Total Economic Valuation (Pearce & Turner 1990, Pearce & Moran 1994).

Importance of understanding benefit flow to the economic valuation

Benefits often flow from their point of production to users elsewhere (Figure 5). For example, water purification takes place throughout a watershed, including areas that may not be populated; the benefits from this process (clean water) are used in downstream populated areas such as cities and agricultural fields.

Benefit flow has a very substantial effect on valuation because it influences the degree of offer and demand for each benefit. For example, water prices are determined at the regional scale because water flows within watersheds (Figure 5d). So, given similar demand, it is expected that water will be cheap in areas where there are abundant supplies and expensive in areas where water is scarce. Given the same supply, water will be more valuable if there is local higher demand (for example, large expanses of agricultural fields). In contrast, the value of carbon (for global climate mitigation) is the same worldwide. Indeed, the sequestration of a tonne of carbon (or the avoidance of the release of one tonne of carbon) creates the same level of benefit irrespective of where in the world it happens (Figure 5e).

The flow of services may be affected by one or more possible mechanisms: physical processes (e.g., currents, winds, diffusion); biological processes (e.g. bee movements, fish migration); and anthropogenic processes (e.g. trade, governance). The spatial configuration of service flow falls into five general categories (Figure 5):

- Locally produced benefits: when the point of service production is the same as the point of use (e.g., soil production);

- Omnidirectional neighbourhood benefits: when service use takes place within a buffer areaa surrounding the point of production (e.g., pollination);

- Directional neighbourhood benefits: when service use takes place in the neighbourhood of the area of production, but only in a given direction (e.g., storm protection);

- Long-distance directional benefits: when service users are located far from the point of production, with services flowing in specific directions (e.g., water provisioning flowing downstream); and

- Globally-distributed benefits: when the service can be used anywhere irrespective of the point of production (e.g. climate change mitigation by carbon sequestration).

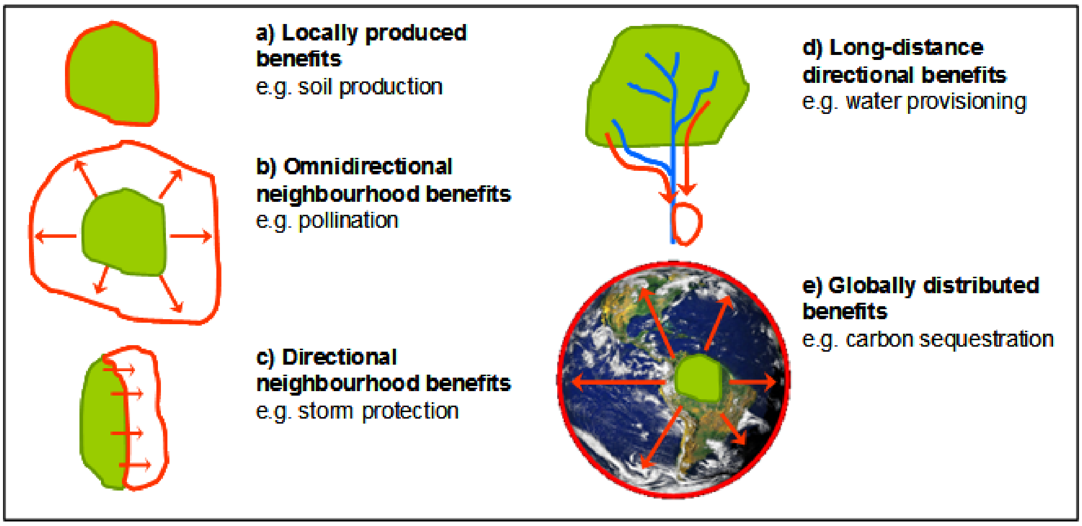


Figure 5. General categories of service flow in relation to spatial configuration (adapted from Fisher et al. *in press*-a).

Quantifying and mapping benefit use

While benefit production is essentially an ecological process, benefit use is intrinsically human-centred. Socio-economic data are therefore fundamental to quantifying and mapping the economic value of variations in benefit production.

First, socio-economic data are key to understanding benefit flow for some ecosystem services. Some anthropogenic processes such as the establishment of particular governance systems (e.g. marine economic exclusive zones, territorial boundaries) put constraints to the natural flow of benefits, while others (e.g. trade) create routes for their distribution.

Second, the use of benefits (underlying their economic valuation) naturally depends on the spatial pattern of users. Individual humans are the ultimate users of all benefits that are valued economically, although value may be through direct use (e.g. drinking water), indirect use (e.g., crop irrigation water), or even through non-use (e.g. water affecting habitat quality of species that have intrinsic value). Even within the same area, different people may have different levels of reliance on ecosystem benefits (thus valuing them differently). For example, poor farmers may depend more directly on benefits provided by nearby forests than other sectors of society.

Third, socio-economic data are key to understanding what drives demand for the benefits obtained from wild nature. The ultimate value of benefits services depend on how much people are willing to pay, which in turn depends on variables such as the relative scarcity of the benefit, the income of the users, and the extent to which the benefit may be replaceable or not.

Finally, the costs of conserving the biodiversity and ecosystems underpinning the benefit will depend substantially on the human pressure in the areas where those benefits are produced.

Relevant socio-economic data to be gathered for the Review are likely to include, amongst others: maps of human population density; geopolitical maps; maps of landuse (with agricultural and urban areas); data on relative income; poverty maps; and data on trade.

Patterns of benefit use will evolve from the interaction between the patterns of service production, the mechanisms of benefit flow, the distribution of users, and the relative demand for the benefit across different users. For example, the benefits of water purification by a forested area are used downstream, given the mechanism of flow for this service (Figure 6). In mapping benefit use it also becomes clear that not all productivity is necessarily used. For example, water purification benefits by a given forest are not used if there is no-one downstream (Figure 6).

Understanding determinants of demand

The economic value of benefits results from the interactions between availability and demand. Such value is therefore likely to be highly dynamic, as both availability and demand change over time. Changes in availability may result from either changes in productivity (including natural fluctuations but also declines caused by overexploitation of the ecosystem) or changes in flow (e.g., changes in governance or in trade). Changes in demand may be caused by changes in flow (e.g., when new roads create a market for wild meat, Section 4.8); socioeconomic changes (e.g., population growth, increased/decreased affluence); sociological changes (e.g. changes in diet or in preferences); technological changes (either by creating alternatives to the benefit, for example water treatment plants, which reduces demand; or by increasing the accessibility of benefits, such as deep sea fisheries, thereby stimulating increased demand).

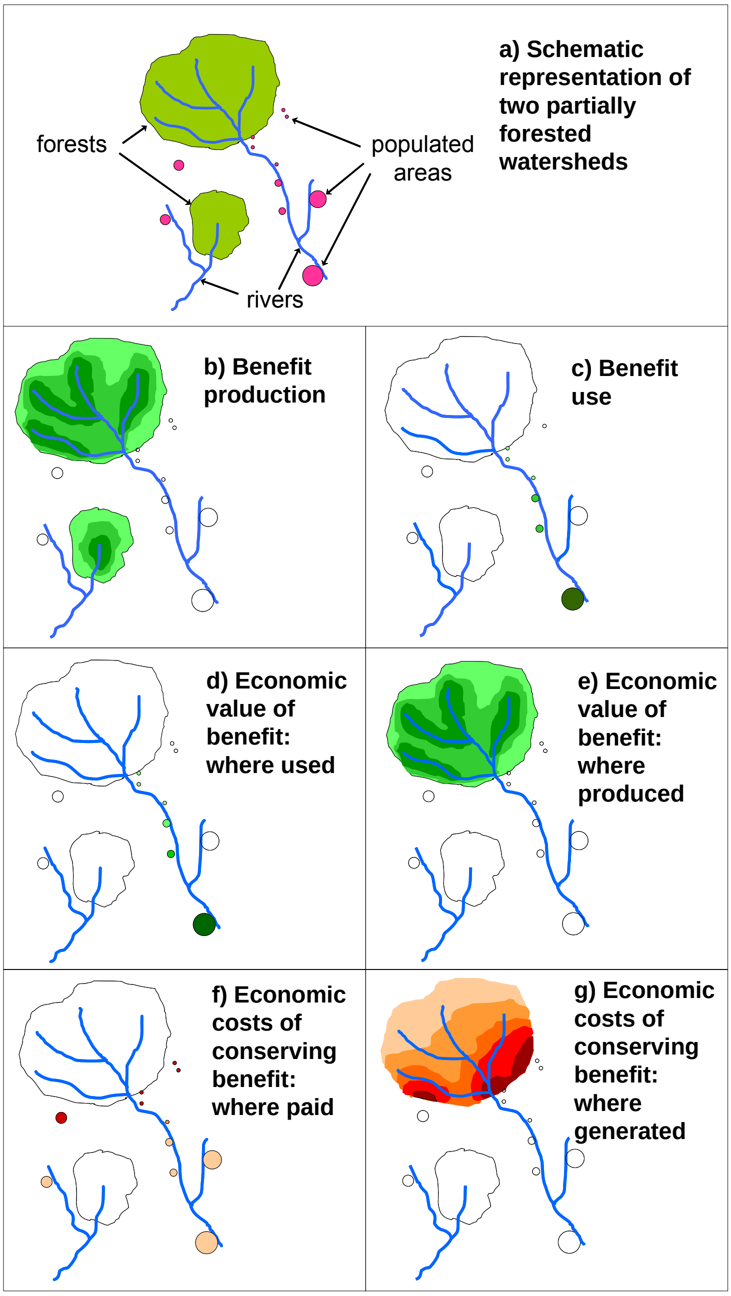


Figure 6. Illustration of the different types of maps of benefit production and costs considered under the framework. a) The scheme represents two forest patches, two river systems, and a set of populated areas. The beneficial process 'water purification' is used as an example. b) Benefit production: water purification takes place upstream, in areas with natural vegetation, such as forest patches. Riparian areas (near the river) are particularly important (darker shades correspond to higher benefit production). c) Benefit use: users of clean water are downstream populated areas and their agricultural fields. Given that the benefit flows along rivers, only those populated areas downstream from a forest patch benefit from the purification service provided by forests (darker shades correspond to higher levels of use). d) Economic value of the benefit, where used: clean water only has an economic value where used, with value depending on demand. e) Economic value of the benefit, where produced: the overall economic value of clean water, established according to its use (d) can be 'back-mapped' into the area of production (b). Areas where production is higher are attributed a higher economic value. Only areas producing services that are being used have economic value. f) Economic value of conserving the benefit, where paid: conserving the forests that help to purify the water has costs. For example, a protected area may be created to prevent logging. Part of the costs may be shared by tax payers throughout the area (light red). However, populations near the forest may incur other costs, such as the opportunity costs of not being able to log the forest or to convert it to agriculture (dark red). g) Economic costs of conserving the benefit, where generated: the overall economic costs of conserving the benefit (f) can be 'back-mapped' into the forest. Costs of conservation (shades of red) are in this example higher near population centres, where there may be more conflict with other forms of land use (and hence higher opportunity costs).

While many of these changes are somewhat unpredictable, a good understanding of what drives demand (coupled with a good understanding of what drives availability) is fundamental for being able to predict/model variations in the value of benefits over space, for alternative states of the world.

Quantifying and mapping economic value for each benefit

A map of the economic value of a particular benefit (or beneficial process) needs to be anchored by actual data points of the value of benefits across space. These may include, for example, studies of the local market value of particular goods (e.g., wild meat) or of the damage costs avoided of certain processes (i.e. carbon sequestration). These can then be extrapolated across space based on the knowledge of the drivers of economic value (demand, offer, flows) combining spatial information on production and use.

The quality of this map – the extent to which the numbers contained in it are reliable – will of course reflect the quality of the knowledge gathered in each of the previous steps. In any case, a basic map can be produced even with very little information (and indeed this has been done before; Costanza et al. 1997). If a single data point is all that is available on the economic value of mangrove for storm protection, for example, then a global map could have all mangroves of the world coloured in the same way. It is, however, very likely that substantially better judgements can be made based on the information gathered in the steps described before. For example, we know that protection from physical storm damage only benefits the neighbourhood of the mangrove, in a directional way depending on the wind (Figure 5). Socio-economic data should highlight which of those areas are more likely to suffer substantial financial damage from storms (e.g., urban centres, industrial areas). Understanding the drivers of demand should help to understand how the monetary value of the service varies with, for example, income.

The value of the economic benefits for each service can initially be mapped onto the point of use, where benefits are enjoyed. For example, economic benefits of pollination would be mapped on the relevant agricultural areas, economic benefits of storm protection on the relevant population centres, and economic benefits of water purification on relevant downstream populations or crop fields (Figure 6d). The same overall value can also be 'back-mapped' onto the area where the ecosystem service is initially produced (forest plots where crop pollinators are based, mangroves that provide valuable storm protection, forests that regulate water for downstream populations; Figure 6e). Only areas producing services that are being used (and therefore that have economic significance) would be included, and their relative value would be defined according to their contribution to the overall economic value (Figure 6d).

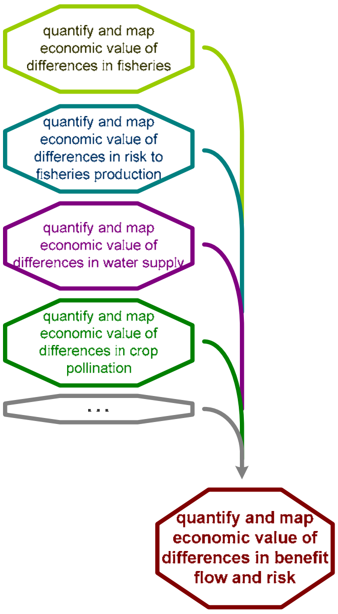
Value over time: discount values

Ideally, a map of the global value of a given benefit should be more than a static picture of value at a given time, but show the overall value accumulated into the future. This would produce a more valuable indication of relative value of the underlying ecosystems. For example, areas where natural resources are currently subject to overexploitation (e.g., overharvested fish stocks) may be currently yielding high economic value at the expense of compromising future productivity. In contrast, in areas where exploitation is being managed sustainably (e.g., through fishing quotas), current production may be lower but more likely to be sustained into the future. Aggregating values over time will require economists to make decisions about discount rates (levels, shapes, fixed or declining, etc). These could potentially defer to the decisions made in the Stern Review (Stern 2007) over the same issues.

Different measures of economic value

Economic value is frequently interpreted in terms of monetary value. However, a diversity of other measures can be considered that offer different insights into the way in which changes in wild nature affect human wellbeing. For example, economic value may be quantified as the fraction of local income, rather than in absolute (e.g. US$) terms. This would highlight areas where the local value of the benefit is low in absolute economic terms but high in terms of livelihoods. For example, the absolute economic value of wild meat is not particularly high but it contributes significantly to the protein intake and food security of millions of people in Africa (Section 4.8).

2.3.5 Quantifying and mapping the overall economic value of changes in benefits derived from the policy action



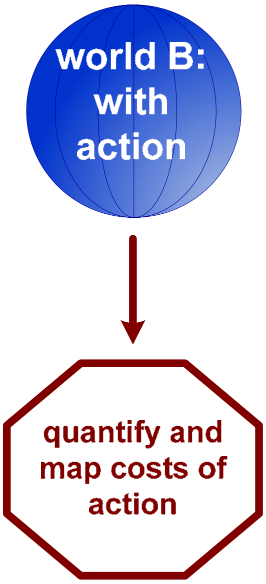
Once converted into a common currency, combining the information for all benefits and services provides an overall quantification of the economic value of differences in benefit and service flow, and in risk, from implementing the policy action (world B) or failing to do so (world A). This calculation may account for differences in purchasing power parity. This information is quantified in a spatially-explicit way, indicating in which parts of the world the net value of benefits increased or decreased.

The spatially-explicit approach followed in the framework means that trade-offs between benefits are effectively accounted for. Indeed, some benefits are competing, rather than additive, such that an increase in the value of one benefit may come at the expense of a reduction in the value of other. For example, the protection of a forest from logging may ensure the continuation of clear water delivery, at the expense of reducing water quantity (Section 4.9). This is dealt with in this framework by being spatially explicit, and by the way states of the world are defined. Most of the differences in benefit production will be noticeable through changes in land use from one state of the world to the other, and these will in many cases indicate which services are being increased and which ones are being lost. For example, a conversion of forest to agriculture will predictably increase the production of services associated with crops (e.g., soil, biological control, pollination) but will reduce services associated with forest (e.g., timber production).

More complex interactions between services could in principle also be accounted for – as long as they are explicitly recognised. For example, if recreational value in a given area (say, a lake) is only possible if regulation of water quality is also in place, then maps of benefits of the two services may be combined by making the value of the former service conditional on the existence of the latter. It is therefore possible to account for leverage amongst ecosystem services (where the occurrence of one service unlocks the conditions for the occurrence of another).

For 'extractive' types of benefits (e.g. fisheries, timber, wild meat), the production will depend on whether the extraction of natural resources is taking place in a more or less sustainable way. Again, this will need to be considered within each scenario. It might be particularly interesting to compare scenarios of sustainable production and of overexploitation. As discussed above, the overall value of benefit production will need to be measured not just for a particular year but integrating the value of production over time. A key decision (likely to affect the results of such comparisons) will be what discount rate to apply.

2.3.6 Quantifying and mapping the costs of policy action



The benefits derived from the conservation of ecosystem service production are of course only part of information needed to decide if such conservation makes economic sense. As established by the Potsdam mandate, these need to be weighed "*versus the costs of effective conservation*".

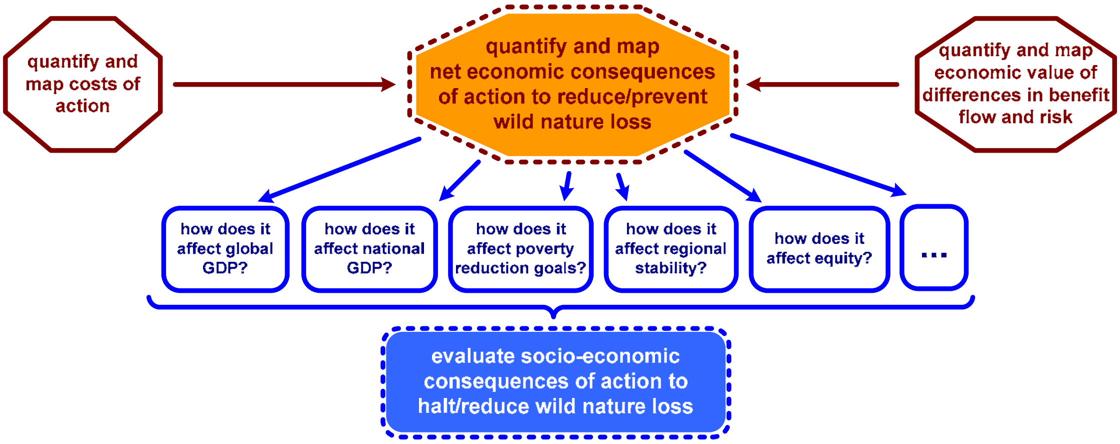
Implementing a policy or set of policies for halting/reducing biodiversity loss and ecosystem degradation will have costs. These include direct costs (e.g. management of marine protected areas) as well as opportunity costs (e.g., compensation for decommissioning part of a fishing fleet). These costs are marginal, because they only exist for the state of the world for which the action is implemented. And they vary across space, and so should be quantified in a spatially-explicit way.

In a few cases, the exact values of those costs will be well inventoried. In most cases, though, the values and geographic spread of the costs will need to be modelled from a collection of data points. For example, data on running costs for a set of protected areas around the world can be used to generate a model for predicting approximate management costs for areas for which no data are available (Balmford et al. 2003, 2004). Creating those models requires the integration of biotic, abiotic and socio-economic data. The separate report "Review of the Costs of Conservation and Priorities for Action", by Bruner, Naidoo and Balmford (also an output of this Scoping the Science project), reviews the state of knowledge for different types of conservation costs.

Conservation costs are initially mapped to the point where they are paid. For example, the running costs of protecting a forested area (e.g., paying forest guards) may be incurred by taxpayers within a given country, while the opportunity costs of not being able to log the forest or convert it to agriculture are more likely to be incurred by populations in the vicinity of the forest (Figure 6f). The value of costs can also be 'back-mapped' onto the area where those costs are generated. In this example, the costs would be mapped onto the forest, being higher in those areas where human pressure is higher, such as near population centres, as this increases opportunity costs (Figure 6g).

Again, an ideal map of the global costs of conserving a given ecosystem service should be more than a static picture of costs at a given point in time, but the overall cost accumulated over time. This would allow for a distinction between areas where initial costs are high and then decline (e.g., one-off costs of establishing a protected area) and areas where costs are likely to increase over time (e.g., where human pressure is increasing and so are pressures on natural habitats).

2.3.7 Quantifying and mapping the net economic consequences of policy action



The net economic consequences of the policy action being evaluated are quantified by comparing the costs of the action with the economic gains from benefits and processes obtained from the additional conservation of biodiversity and ecosystems. It is this comparison that will provide the ultimate answer to the question of the Review: what are the economic consequences of global biodiversity loss?

A key advantage of adopting the current framework to address this question is that it can help explore spatial variation in the answer. Irrespective of whether the net global result is positive or negative, there will always be winners (where there are increases in benefits and/or decreases in costs) and losers (where there are decreases in benefits and/or increases in costs) from the implementation (or lack thereof) of particular actions.

Recognising these patterns is essential for fully understanding the socio-economic implications of particular actions. A diversity of policy-relevant questions can be addressed with this information, besides the global aggregated result. The spatially-explicit nature of the assessment will help answer questions such as:

- How are national economies likely to be affected? Are the poorest countries the ones that benefit/lose the most from a particular action?

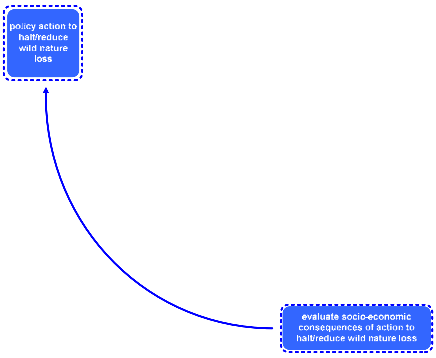
- How does the action (or failure to implement it) affect development goals, such as the Millennium Development Goals? Does it contribute to alleviate poverty or access to water?

- How does the action (or failure to implement it) affect regional stability? Are livelihoods improved in areas currently under great socioeconomic strain or are they likely to worsen?

- How does the action (or failure to implement it) affect equity? Are the winners amongst the richest and the losers amongst the poorest, or vice-versa?

Understanding spatial variation in costs and benefits also allows for the exploration of conservation trade-offs, highlighting for example regions that are conservation bargains (where benefits from conservation largely exceed the costs) and therefore good investments for immediate conservation.

2.3.8 Evaluating the adequacy/desirability of the policy action



From the analyses of this set of policy-relevant questions is then possible to make a better judgement of the overall socio-economic consequences of a particular action or set of actions, and therefore whether the action contributes to improve human wellbeing or not. This in turn informs whether the policy action make sense from ecological, economic, and social perspectives, and therefore whether its implementation is recommended or not.

If the action is implemented, an understanding of the socio-economic impacts provides the opportunity for adding mechanisms that make the action fairer and more effective. For example, market tools such as payments for ecosystem services may be developed to internalise the full costs and benefits of conservation. These can help ensure that conservation takes place where it is needed (e.g. payments for ecosystem services from cities to rural areas) while reducing inequity and social injustice (e.g. when costs of conservation are imposed on, or costs due to loss of biodiversity are incurred by, local populations without adequate compensation).

2.4 Summary of key points from the conceptual framework

In summary, the conceptual framework relies on the *spatial assessment* of the variation in the *marginal* benefits and costs of biodiversity and ecosystems conservation. Marginality is essential because at all times the relevant question is what is the *difference* in benefits and costs from the implementation, or not, of a particular policy package. Being spatially explicit is vital because we need to know how costs and benefits vary across space.

The rationale and key characteristics of the conceptual framework are:

a. A Review of the economic consequences of the loss of biodiversity and degradation of ecosystems (throughout referred to as wild nature loss) is not about quantifying the overall value of wild nature to human wellbeing; such value is infinite, because we cannot live outside of nature. Instead the Review is about quantifying the marginal costs and benefits associated with the loss of ecosystems and their biodiversity.

b. Quantifying marginal costs and benefits requires contrasting two well-defined situations, which we term states of the world. A 'state of the world' is a particular set of biotic and abiotic conditions, including aspects such as land cover, human distribution, human activities, and conservation actions in place. States of the world are the end products of scenarios.

c. The two states of the world being contrasted in each case need to be carefully matched, such that one is the counterfactual of the other, by being equal in everything else except the implementation or not of a set of actions aimed at reducing losses in wild nature losses. Many possible pairs of matching states of the world can be contrasted, each contrast evaluating the economic consequences of a specific set of policies. For example, the contrast between worlds with and without an effective network of marine protected areas would be adequate for quantifying the economic consequences (costs and benefits) of reducing the loss of marine wild nature through protected areas.

d. Given two matching states of the world, the calculation of the economic consequences of losing wild nature by failing to adopt the policies that distinguish the two states requires calculating:

   - The difference in the provisioning of benefits from wild nature (marginal benefits; higher in the biodiversity friendly world; e.g., higher long-term fisheries production in the state of the world with marine reserves);

   - The difference in the costs associated with the biodiversity-friendly policy measures (marginal costs; higher in the biodiversity friendly world; e.g. opportunity and management costs of establishing marine reserves).

   The net consequences of biodiversity loss are obtained by comparing the marginal benefits with the marginal costs.

e. The quantification of the benefits and costs must be spatially explicit. These spatial considerations make the framework a useful tool for addressing key considerations in the evaluation and development of adequate policy measures, by:

   - Allowing for the quantification of costs and benefits to be contextualised more appropriately, thereby shedding light on how different policy options affect development goals.

   - Requiring the explicit statement of the assumptions being made when extrapolating across heterogeneous landscapes using limited data.

   - Enabling an understanding of the mismatches between winners and losers from particular policy actions. This is fundamental for understanding social impacts, equity/fairness issues, and issues of responsibility and governance, as well as for the development of market tools that internalise the full costs and benefits of conservation, ensure that conservation takes place where it is needed and improves the effectiveness of policy actions while reducing inequity and social injustice.

   - Enabling the exploration of trade-offs, highlighting for example regions that are conservation bargains (where benefits from conservation largely exceed the costs) and therefore good investments for immediate conservation.

2.5 Participants

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3 A STRATEGY FOR REVIEWING KNOWLEDGE ON THE LINKS BETWEEN WILD NATURE AND HUMAN WELLBEING

3.1 The need for a strategy

The bulk of this report (sections 3 and 4) covers the most substantial task of the Scoping the Science project, of providing "*a coherent overview of existing scientific knowledge upon which to base the economics of the Review, and to propose a coherent global programme of scientific work, both for Phase 2 (consolidation) and to enable more robust future iterations of the Review beyond 2010*."

This task entails reviewing the vast literature on the many mechanisms through which wild nature contributes to human wellbeing, each typically comprising several mechanisms by which changes in biodiversity and/or in the state of ecosystems have economic consequences. Some of these mechanisms are very direct, for example a decline in fish biomass resulting in the decline in food obtained from fisheries. Others are rather indirect, for example a decline in forests affecting water flow, resulting in increased erosion, resulting in sedimentation of coral reefs, resulting in fish population declines, resulting in a decline in food obtained from fisheries.

Partitioning the vast diversity of links between wild nature and human wellbeing into themes is a critical step for organising the review of those links. Such a stucture needs to be robust, ensuring that it covers the most important links while avoiding overlap (and therefore, economic double-counting).

This section details the rationale and the strategy followed for creating a robust classification of the links between wild nature and human wellbeing, and a coherent partition of our review of ecological knowledge into manageable thematic reviews that provide an adequate platform for the economic evaluation.

3.2 The Millennium Ecosystem Assessment as a starting point

The Millennium Ecosystem Assessment (MEA 2005) firmly established the concept of ecosystem services as an important model for linking the functioning of ecosystems to human welfare benefits.

The MEA framework (Figure 7) was built to demonstrate the importance of ecosystems for the constituents and determinants of human wellbeing and has been very successful in doing so. The MEA thus becomes the groundwork for the evaluation of the economic consequences of biodiversity loss. However, the MA itself was not developed as a valuation exercise, and it has been pointed out that its framework is not directly fit for that purpose (Boyd & Banzhaf 2007; Wallace 2007; Fisher et al. *in press*-a,b).

Here we build from the MEA to propose a classification of the links between wild nature and human wellbeing. As clarified before, the term 'wild nature' is used throughout to refer to biodiversity (both diversity, such as genetic diversity, and amount, such as biomass) and ecosystems. We start by presenting the proposed classification, and then clarify how it relates to the MEA framework. This is, naturally, work in progress: the MEA is still very recent, and much more work will still build from it.

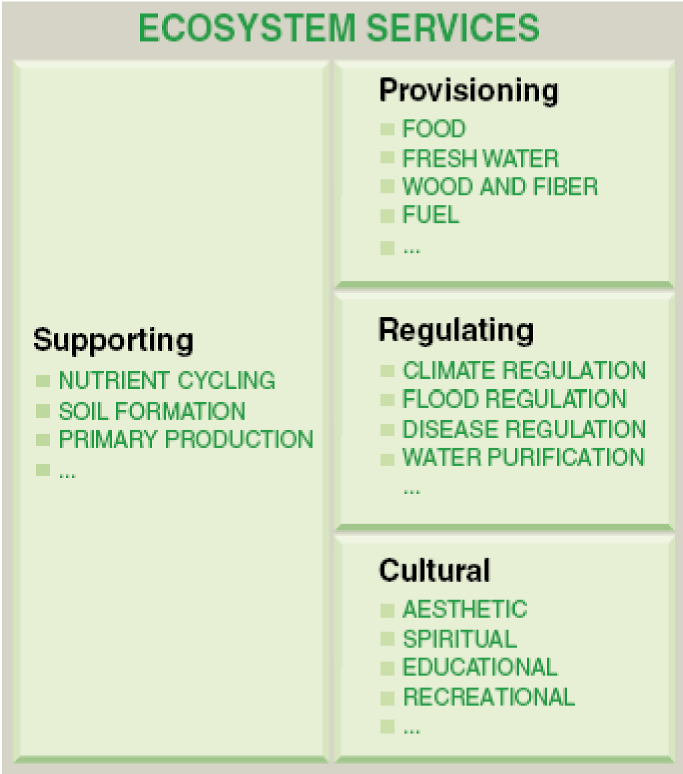

Figure 7. Classification of ecosystem services followed in the Millennium Ecosystem Assessment (MEA 2005).

3.3 The need to avoid double counting

Ecosystem services have been defined in the literature in a diversity of ways (see Fisher et al. *in press*-a for a review), including the "conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life" (Daily 1997), "the benefits human populations derive, directly or indirectly, from ecosystem functions (Costanza et al. 1997), and "the benefits people obtain from ecosystems" (MEA 2005). These definitions reveal a mix between ecological functions (e.g. pollination, water regulation) which are the means for the production of benefits, and the ends (benefits) resulting from these processes (e.g., food from crops, drinking water) (Wallace 2007).

The MEA classification of ecosystem services has proved highly useful as an educational and policy tool. However, as many economists have pointed out since its publication, it is not fit for the purpose of economic evaluation (Boyd & Banzhaf 2007; Wallace 2007; Fisher et al. *in press*-a). By mixing processes (means) and benefits (ends) it is particularly prone to double counting. For example, the regulating service 'water purification' provides added value to the benefits that this process underpins, including drinking water, food (through purification of water used in crop irrigation and in rivers producing freshwater fisheries) and cultural benefits (e.g., aesthetic, spiritual and tourism benefits from a clean river). Valuing separately 'water purification' and 'drinking water', 'food', and 'cultural benefits' results in double counting of the value of water purification. Another example of double-counting in the MA classification is between pollination or pest regulation and food provision, because the value of pollination and crop pest regulation manifests itself through added food production. The relationship between our proposed classification and that followed in the Millennium Ecosystem Assessment is clarified in Section 3.7.

The Total Economic Value framework (Pearce & Turner 1990; Pearce & Moran 1994) also has to be interpreted carefully if it is to be used as a basis for economic valuation, as there is much scope for double-counting particularly between direct and indirect use values. The relationship between our proposed classification and the Total Economic Value framework is clarified in Section 3.8.

3.4 Proposed classification of ecosystem processes and benefits

The classification presented here was developed with the explicit objective of providing an adequate basis for economic evaluation, within the conceptual framework for the broad Review on the Economics of Ecosystems and Biodiversity (Section 2).

We make an explicit distinction between processes and benefits. We consider two types of ecological processes: **'core' ecosystem processes,** the basic ecosystem functions (e.g., nutrient cycling, water cycling) supporting the processes that provide benefits to humankind (corresponding to "intermediate services" in Fisher et al. *in press*-a,b); and **'beneficial' ecosystem processes**, the specific ecosystem processes that directly underpin benefits for humankind (e.g. waste assimilation, water purification; corresponding to "final services" in Fisher et al. *in press*-a,b). The **benefits** are the end products of these beneficial ecosystem processes (e.g., clean drinking water) (Figure 8; Table 1; these are also called benefits in Fisher et al. *in press*-a,b).

Benefits are therefore the discrete products of ecosystem processes that directly impact human wellbeing, ranging from food to spiritual fulfilment. These can, in principle, be valued in monterary terms. In theory, it is possible to predict how each ecosystem process contributes to the production of benefits, by considering the relationships between core and beneficial ecosystem processes, and between the latter and benefits (Figure 9). For example, pollination is a beneficial ecosystem process contributing to the production of some biofuel crops. Pollination in turn is potentially regulated by several ecological processes: evolutionary processes resulting in the diversification of plants and pollinators, and their co-adaptation; animal-plant ecological interactions; production supporting the populations of pollinators and the other species they depend on (e.g. forest trees).

Figure 8. Classification of ecosystem processes and benefits followed in this study (lists of processes and benefits are not exhaustive).

Table 1. Types of core and beneficial ecosystem processes, and of benefits (not an exhaustive list).

CORE ECOSYSTEM PROCESSES: basic ecosystem processes supporting ecosystem services

Production: Production of plant and animal biomass.

Decomposition: Reduction of the body of a formerly living organism into simpler forms of matter.

Nutrient cycling: Cycle by which a chemical element or molecule moves through both biotic and abiotic compartments of ecosystems (e.g. nitrogen cycle, phosphorus cycle, carbon cycle).

Water cycling: Cycle of water through both biotic and abiotic compartments of ecosystems.

Weathering/erosion: Weathering is the decomposition (*in situ*) of rocks, soils and their minerals through direct contact with the atmosphere. Erosion involves the movement and disintegration of rocks and minerals by agents such as water, ice, wind and gravity.

Ecological interactions: Inter- and intra-specific interactions between organisms (e.g., predation, competition, parasitism, and animal-plant interactions such as pollination).

Evolutionary processes: Genetically-based processes by which life forms change and develop over generations (inc. evolution, speciation, adaptation).

BENEFICIAL ECOSYSTEM PROCESSES: ecosystem processes that directly underpin benefits to people

Biomass production: primary: Production of plant biomass.

Biomass production: secondary: Production of animal biomass.

Pollination: Pollen transport (particularly by organisms). [Seed and fruit dispersal may also be considered]

Biological control: Inter- and intra-specific interactions resulting in reduced abundance of species that are pests, diseases or invasives in a particular ecosystem.

Other ecological interactions: Other inter- and intra-specific interactions, for example competition and predation.

Formation of species habitat: Formation of the physical properties of the habitats necessary for the survival of species (e.g., canopy structure in forests).

Species diversification: The production of genetic diversity across species.

Genetic diversification: The production of genetic diversity within species.

Waste assimilation: Removal of contaminants from the soil in an ecosystem (inc. through biological processes such as decomposition or assimilation).

Soil formation: Process by which soil is created (including changes in soil depth, structure and fertility).

Erosion regulation: Control of the processes leading to erosion (e.g. by controlling the effects of water flow, wind or gravity).

Formation of physical barriers: Formation of structures that attenuate the energy of (or block) water or wind flow (e.g., mangroves, dunes, forests).

Formation of pleasant scenery: Formation of landscapes that are attractive to people.

Air quality regulation: Removal of contaminants from air flowing through an ecosystem (inc. through physical processes such as filtration or biological processes such as decomposition or assimilation).

Regional/local climate regulation: Modulation of regional/local climate (e.g., of temperature, or humidity).

Water regulation (timing): Regulation of the timing of water flow through an ecosystem (e.g., attenuation of floods/droughts).

Water purification (quality): Removal of contaminants from water flowing through an ecosystem (inc. through physical processes such as filtration or biological processes such as decomposition or assimilation).

Water provisioning (quantity): Changes in the quantity of water flowing through an ecosystem.

Global climate regulation: Modulation of global climate and ocean acidity through changes in the concentration of greenhouse gases in the atmosphere.

Currently unknown beneficial processes: the possibility that wild nature contributes to our current and/or future welfare in ways we currently do not realise. For example, the contribution of forests to the regulation of global climate has only very recently been realised as a beneficial process.

BENEFITS: the products of ecosystem processes that directly impact human wellbeing (we are specifically interested in understanding the role of wild nature in providing these benefits)

Food:

- From crops (including orchards, mushroom production, cultivated algae);
- From livestock (including poultry);
- From capture fisheries (marine and freshwater);
- From aquaculture;
- Other wild foods (including wild meat, mushrooms, invertebrates, etc);
- ...

Freshwater (for direct consumption; excludes irrigation water, covered in crops):

- Drinking water;
- Water for industry;
- ...

Raw materials:

- Timber (from natural forests and from plantations);
- Fibres from domestic plants (e.g., cotton), or from domestic animals (e.g., wool);
- Fibres from wild plants (e.g., rattan), or from wild animals (e.g., hides);
- Synthetic materials copied from/inspired by natural products;

- ...

Energy:

- Biofuels (e.g., palm oil, algae) from domestic plants;
- Coal/firewood from wild plants;
- Dung from livestock;
- Working animals (e.g. oxen, llama);
- Hydroelectric energy;
- ...

Property:

- Private property value and condition;
- Infrastructure condition (e.g., hospitals, factories);
- ...

Physical health:

- From synthetic medicines copied from/inspired by natural products;
- From cultivated medicines;
- From medicines harvested from wild species;
- By avoiding injury (e.g. from natural hazards);
- By avoiding pollution (e.g., air pollution);
- By avoiding contamination (e.g., contaminated water);
- By stimulating physical exercise (e.g., hiking, diving).
- ...

Psychological wellbeing:

- From tourism;
- From other recreation (e.g., hiking, diving);
- Through spiritual/cultural wellbeing (e.g., sense of wonder from nature);
- Through aesthetic benefits (e.g., pleasure to watch a beautiful landscape);
- From nature watching (e.g., bird watching, coral fish watching);
- From garden plants and pets;
- ...

Knowledge:

- Through research of the natural world;
- Through education about the natural world.
- ...

Currently unknown benefits: the possibility that wild nature provides/will provide benefits currently unknown (e.g., algae now considered a promising biofuel).

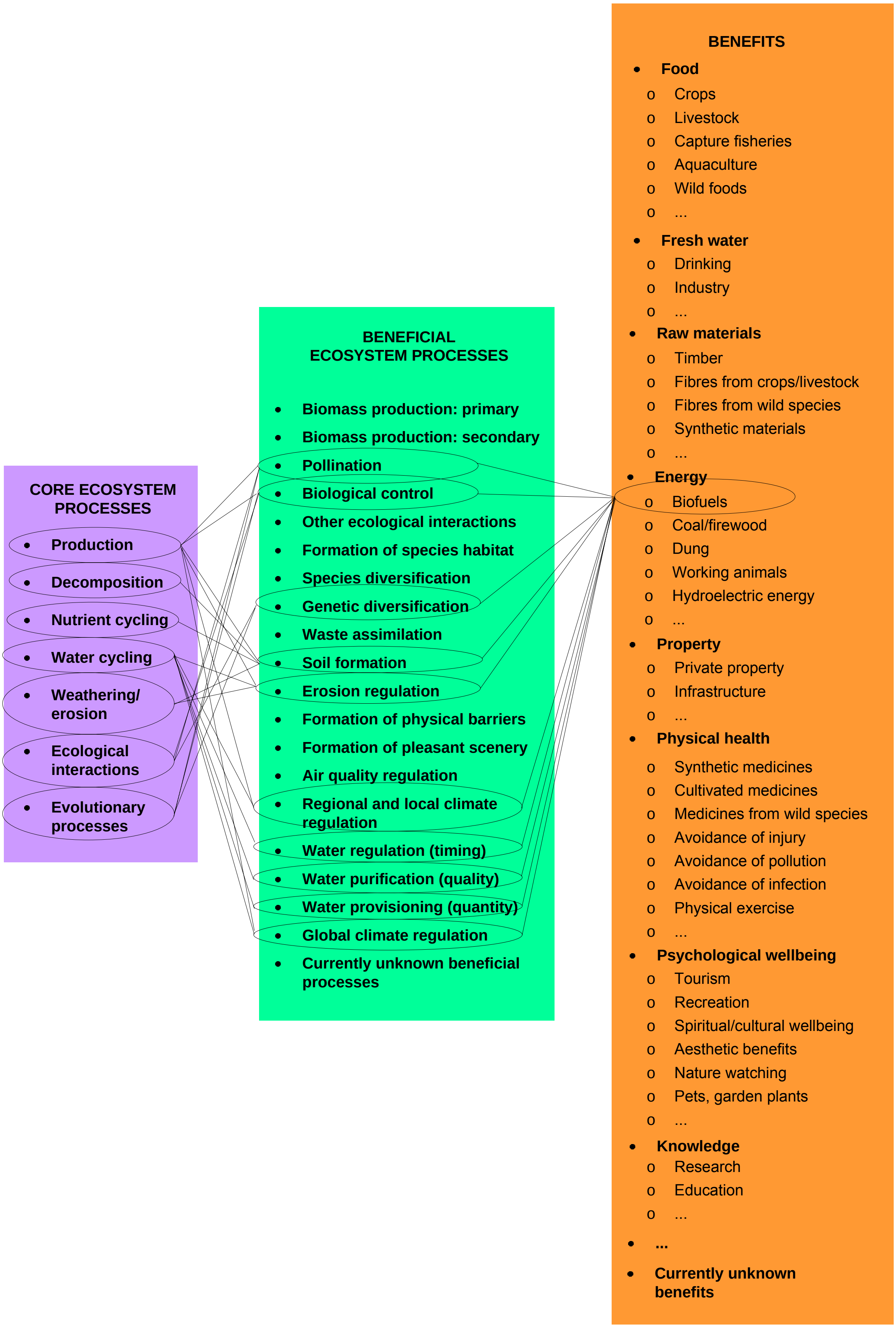

Figure 9. Core and beneficial ecosystem processes underpinning the provisioning of a biofuel crop (lists of processes and benefits are not exhaustive).

3.5 Partitioning the links between wild nature and human wellbeing as a basis for the thematic reviews

Ideally, we would have liked to evaluate the state of knowledge on the links between wild nature and the production of each benefit amongst those listed in Figure 8. In practice, this was not possible within the time frame of this project, and it is also unlikely that it could be done in Phase 2 as well. We have therefore bundled these links into a smaller, but hopefully manageable, number of thematic reviews.

Firstly, we considered general categories of benefits, rather than the more detailed ones in Figure 8 and Table 1. These general categories were defined in terms of the way the benefits are produced (e.g. through cultivation, through fisheries) rather than by type of benefit (food, water, energy, etc.). That is, we grouped benefits that we assumed would have a similar production function (Figure 3). The assumption was that benefits that are similarly affected by ecosystem processes are also likely to be similar in their sensitivity to changes in biodiversity and ecosystems. For example, we grouped all benefits produced through cultivation (e.g., food crops, fibre crops, biofuel crops), because these are likely to be similarly affected by ecosystem processes such as biological control, pollination, water regulation and erosion regulation. If, in contrast, we had grouped benefits by main type (food, water, energy, etc.; Figure 8) we would have had to consider together benefits produced in quite distinct ways (e.g. food produced through crops and through marine fisheries; energy produced through hydroelectric power and through biofuels). The categories of benefits considered (rows in Figure 10) are:

- **Crops**: including for food (e.g., wheat, potatoes), fibres (e.g. cotton, linen), biofuels (e.g. palm oil, sugar cane), timber and paper pulp production (e.g. pine and eucalyptus plantations), ornaments (e.g. flowers), and stimulants (e.g. coffee, cocoa). Benefits are mainly in terms of food, but also raw materials, energy (biofuels), and psychological wellbeing (stimulants and ornamentals).

- **Livestock**: includes all domesticated animals raised for the production of food (including eggs, milk), fibres (e.g., wool), or energy (dung and working animals).

- **Marine fisheries**: includes benefits extracted from animals (including invertebrates such as crustaceans and molluscs) harvested from the seas and oceans, including from mariculture. Benefits are mainly food (including indirectly through fishmeal), but also include psychological wellbeing (recreational fisheries) and possibly physical health (e.g. fish liver oil as a food supplement).

- **Inland fisheries**: includes benefits extracted from animals (including invertebrates) harvested from freshwater systems, including from freshwater aquaculture. Benefits are mainly food, but also psychological wellbeing (recreational fisheries).

- **Wild animal products**: includes benefits extracted from wild harvested animals (including invertebrates), Benefits are mainly food, but also psychological wellbeing (recreational hunting), raw materials (hides, fur) and physical wellbeing (medicinal value).

- **Drinking and industry water**: fresh water that is directly consumed by people or as a raw material in industry.

- **Hydroelectric energy**: the production of energy through the movement of water.

- **Wild plant fibres**: fibres harvested from wild plants, including timber, paper pulp, rattan and bamboo from natural forests, for raw materials and energy (firewood).

- **Wild medicinal plants**: plants with medicinal value harvested from the wild, providing physical health as benefits.

- **Nature-related outdoor activities**: how wild nature contributes to benefits such as human health, psychological wellbeing and education through outdoor activities such as nature tourism, hiking, cycling, diving, camping, species-watching, use of urban parks, and gardening.

- **Avoidance of injury and property loss**: avoidance of personal injury or property loss, for example through the prevention of hurricanes and mudslides. Benefits are mainly in terms of physical health (avoided injury) and property (avoided property losses), but also psychological wellbeing (sense of security).

- **One-time use benefits**: how wild nature contributes to health, psychological wellbeing, physical comfort, and knowledge, by maintaining the biological diversity from where ideas, chemical compounds, and images can be sourced. This only requires one-off use of nature, as the elements are initially obtained from wild nature but subsequently propagated outside of it. They include pharmaceuticals compounds, raw materials and ideas inspired by/copied from nature, as well as photography, films and art based on/inspired by nature.

- **Non-use benefits**: non-material benefits, in which appreciation of nature (species and landscapes) results in improved psychological wellbeing, knowledge and social relations without a direct use of biodiversity. These may translate into cultural diversity and heritage, spiritual and religious values (inc. sacred species and groves), knowledge systems, educational values, aesthetic values, social relations, and sense of place (MA 2005). The benefits they deliver are mainly in terms of psychological wellbeing (e.g. sense of wonder).

- **Unknown benefits:** present and future contribution of wild nature to human wellbeing through benefits that are currently not realised. While these values are difficult or even impossible to measure, they may be very substantial to our wellbeing. By definition, they may contribute to all other types of benefits.

We would have liked to focus on these groups of end benefits as the basis for the economic valuation. Focusing on benefits prevents double counting, as the value of different ecosystem processes would be partitioned according to their contribution to each of the final benefits (e.g., part of the value of water regulation would be attributed to drinking water, part to crop production). In practice, we were unable to follow this strategy. Indeed, while in some cases, the ecological knowledge is aligned with the production of benefits (e.g. marine fisheries; rows in Figure 10), in others the literature focuses on processes (e.g. water regulation; columns in Figure 10). We therefore partitioned our task into thematic reviews that are a mix of benefits and processes (Figure 10). By plotting the links between benefits and processes onto a matrix (Figure 10), we were able to group them in a way that is a compromise between theoretical ideal and practical reality, while explicitly avoiding double-counting.

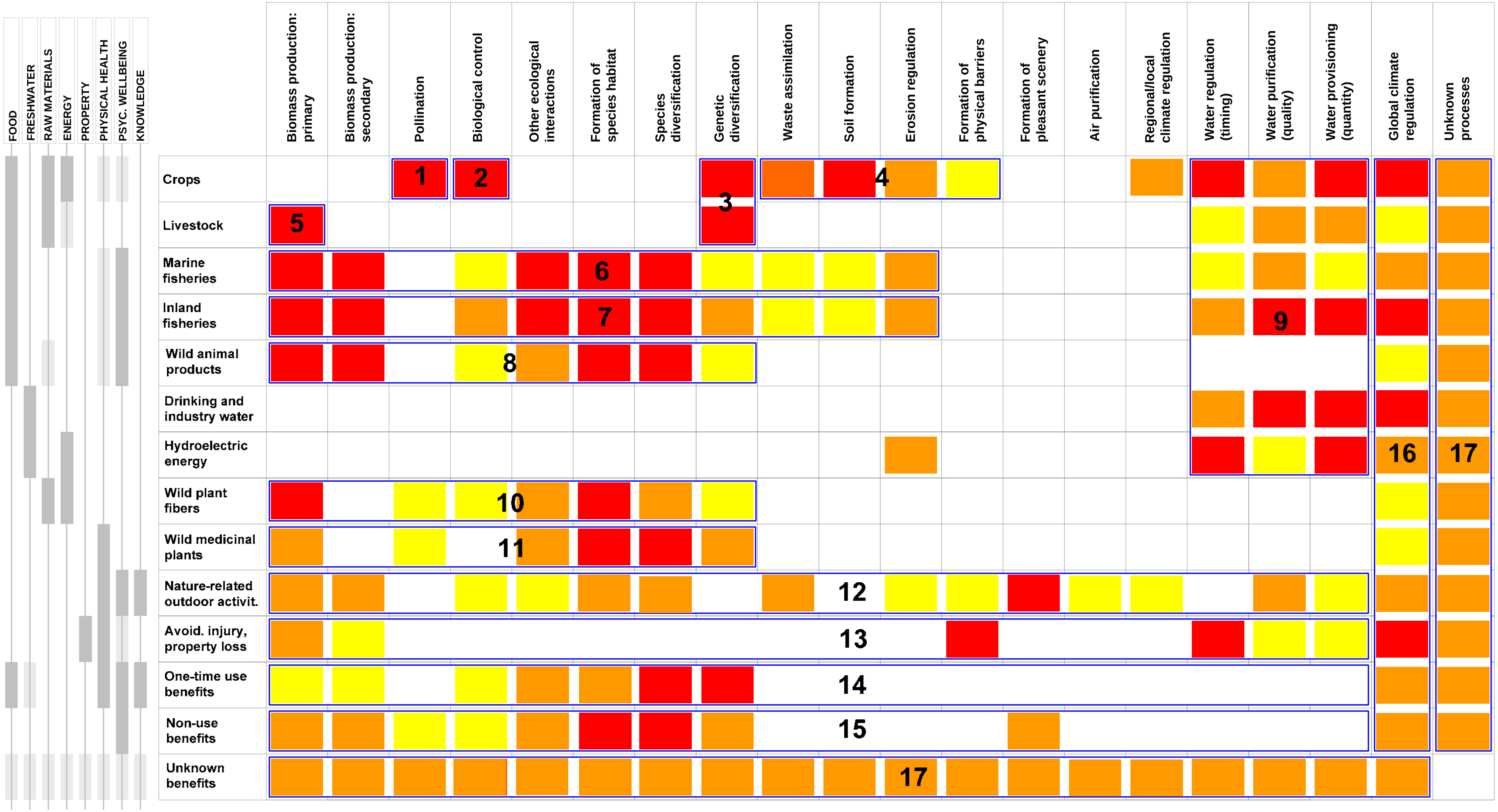

Figure 10. Review themes in this project (1 to 17), and how they relate to beneficial ecosystem processes (columns) and benefits (rows). See text details on each number. Grey bars on the left indicate main types of benefit. Each review theme is represented by a block with a blue border. Colours indicate an estimate of the importance of the link between each ecosystem process and each type of benefit: red – very important; orange – important; yellow – somewhat important; white – relatively unimportant.

For example, we considered global climate regulation (including regulation of ocean acidity) as a single ('vertical') theme, rather than considering it within each of the relevant benefits (e.g. a part of crop production, of avoidance of injury and property loss, of the production of marine fisheries; Figure 10). We decided to do so because the Stern Review (Stern 2007) has, effectively, already valued the consequences of avoiding climate change by considering its effects on each of a diversity of benefits, including food crops, injury and property damage, and marine fisheries. Stern did not include all the possible links between climate change and human wellbeing (e.g., how climate change affects recreation, tourism, research and education, spiritual and cultural wellbeing), but we recommend that it would be a better investment of resources in Phase 2 to build from the Stern Review rather than to repeat their analyses (and invest those resources in any of the other themes, which are much less advanced in terms of valuation). In order to avoid double-counting, the effects of climate change on benefits have been excluded from all the other themes (Figure 10).

3.6 Thematic reviews

Overall, we considered 17 thematic reviews (Figure 10). Some of these have the same titles as the benefit categories listed before, but often the review had a narrower focus/definition, as explained:

- **Wild crop pollination (1)**: how wild nature contributes to crop yields through wild (unmanaged) pollination (Section 4.1).
- **Biological control of crop pests (2)**: how wild nature contributes to crop yields through the control of crop pests and diseases (Section 4.2).
- **Genetic diversity of crops and livestock (3)**: how wild nature contributes to agriculture and livestock production by maintaining a diversity of populations of wild relatives and varieties (Section 4.3).
- **Soil quality for crop production (5)**: how wild nature contributes to crop yields by contributing to soil formation and to preventing erosion regulation (Section 4.4).
- **Livestock (5)**: we focused specifically on the way wild nature contributes to livestock production through the provision of natural grazing and browsing areas (rangelands) (Section 4.5).
- **Marine fisheries (6)**: we focussed specifically on capture fisheries (mainly for food), with some considerations on recreational fisheries (Section 4.6).
- **Inland fisheries (7):** we focussed on capture fisheries and aquaculture (mainly for food), with some considerations on recreational fisheries (Section 4.7).
- **Wild animal products (8)**: we focused on wild meat, with some considerations on recreational hunting (Section 4.8).
- **Fresh water provision and regulation (9)**: how wild nature affects freshwater quantity, timing and quantity (Section 4.9).

- **Wild plant fibres (10)**: we focussed on timber production (Section 4.10).

- **Wild medicinal species (11)**: how wild nature contributes to human health through the provision of a diversity of harvestable medicinal plants (Section 4.11).

- **Nature-related outdoor activities (12)**: how wild nature contributes to benefits such as human health, psychological wellbeing and education through outdoor activities such as nature tourism, hiking, cycling, diving, camping, species-watching, use of urban parks, and gardening (Section 4.12).

- **Natural hazard regulation (13)**: how wild nature contributes to avoid human injury and property loss by avoiding or mitigating the effects of natural hazards, including coastal storms, hurricanes, floods, avalanches, mudslides (Section 4.13).

- **One-time biodiversity use values (14)**: how wild nature contributes to health, psychological wellbeing, physical comfort, and knowledge by maintaining the biological diversity from where ideas, chemical compounds, and images can be sourced. We focused on pharmaceutical compounds (Section 4.14).

- **Non-use values (15)**: non-material benefits, in which appreciation of nature (species and landscapes) results in improved psychological wellbeing, knowledge and social relations without a direct use of biodiversity. These may translate into cultural diversity and heritage, spiritual and religious values (inc. sacred species and groves), knowledge systems, educational values, aesthetic values, social relations, and sense of place (MEA 2005). The benefits they deliver are mainly in terms of psychological wellbeing (e.g. sense of wonder) (Section 4.15).

- **Global climate regulation (16)**: how wild nature contributes to human wellbeing by contributing to climate regulation (concentration of greenhouse gases in the atmosphere). We focused on terrestrial systems (Section 4.16).

- **Unknown benefits or processes (17)**: wild nature may potentially contribute to our future welfare in ways currently not realised, including both through ecosystem processes not currently known or valued (e.g., carbon storage and sequestration has only recently been identified as a ecosystem valuable process) and through benefits currently not predicted (e.g., algae are just starting to be considered as a promising biofuel). While these benefits are difficult or even impossible to measure, they may be very substantial (Section 4.17).

In addition to these main review themes, we considered three cross-cutting themes:

- **Resilience**: review the evidence for how wild nature contributes to the resilience in the provisioning of each of the benefits and services described in points 1 to 17 (this was covered under each main theme).

- **Scenarios**: brief overview of main tools available for generating scenarios that could be useful in Phase 2 (Section 4.18). Under each main theme there is also a brief description of the main type of information that scenarios would need to produce.

- **Prioritisation**: a systematic assessment of the relative priority of the recommended analysis under the main themes for Phase 2 of the review (and beyond) based on a combination of predicted feasibility (given the state of ecological knowledge) and importance (how they are likely to affect the results of the valuation) (Section 4.19.1).

3.7 Relationship between the proposed classification and the Millennium Ecosystem Assessment framework

The MEA followed a classification of ecosystem services into supporting, provisioning, regulating and cultural (Figure 7). These types of services map into our proposed classification as illustrated in Figure 11. Essentially, the MEA's supporting services correspond mainly to our 'core' ecosystem processes, although they also include some 'beneficial' services. The regulating services essentially correspond to our 'beneficial' processes, and provisioning and cultural services tend to correspond to our benefits.

The match is not perfect: the MEA provisioning service 'genetic resources' (underpinned by our processes 'genetic diversification' and 'species diversification') is spread across a diversity of benefits such as crop and livestock genetic diversity contributing to food production, diversity of wild pollinators contributing to crop yields, and diversity of wild species contributing to tourism. The MEA regulating service 'natural hazard regulation' is partially covered by our process 'formation of physical barriers', but also by water, climate and erosion regulation. The benefits provided by these services include the avoidance of personal injury, of damage to property and infrastructure, of damage to crops, and of damage to tourism.

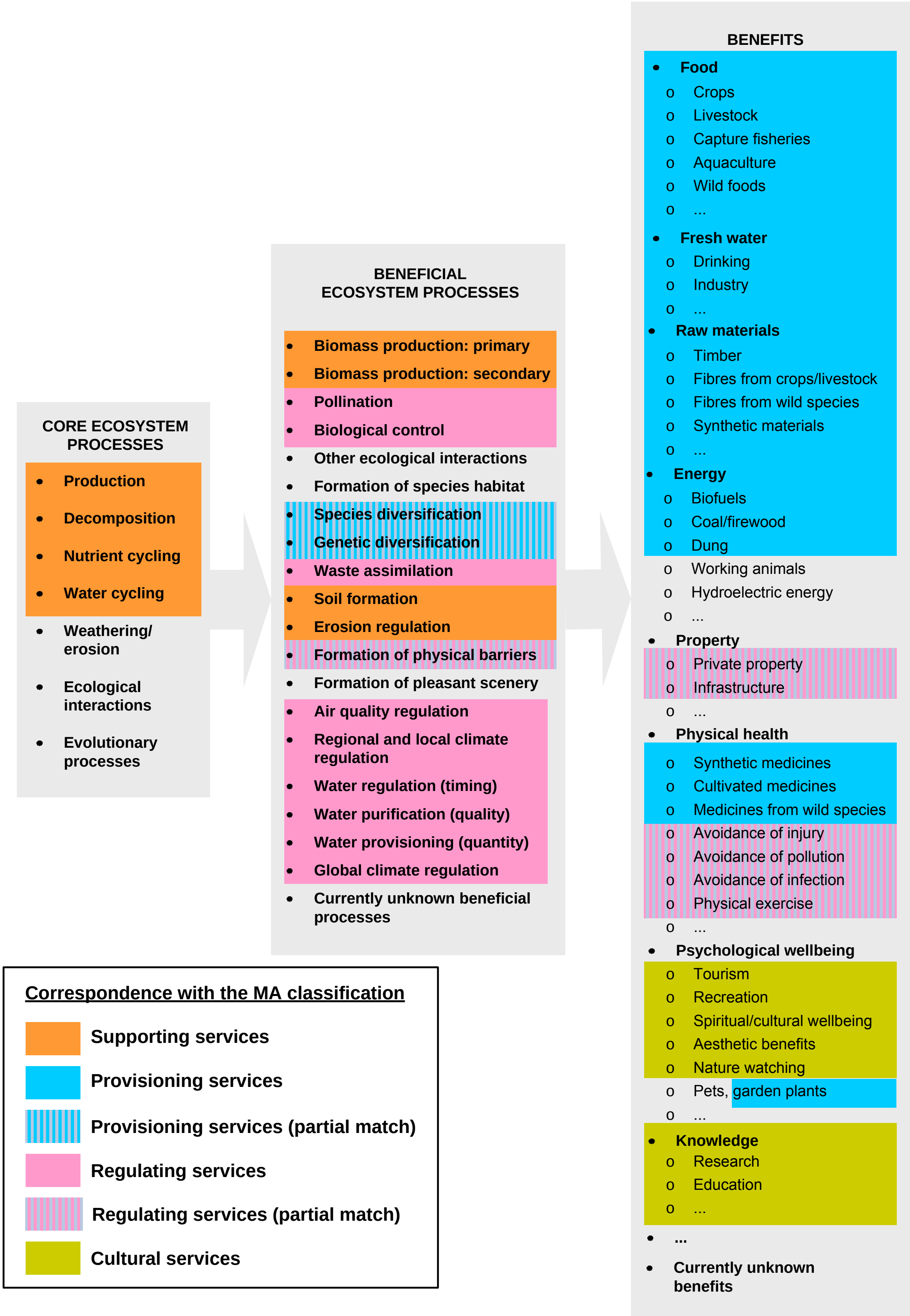

Figure 11. Correspondence between the classification followed in this review (into core ecosystem processes, beneficial ecosystem processes, and benefits) and the classification followed by the Millennium Ecosystem Assessment (into supporting services, provisioning services, regulating services, and cultural services).

3.8 Relationship between the proposed classification and the Total Economic Value framework

While the MEA is still relatively recent, and few economic papers have been published on it, a longer and well-established approach for classifying the value of wild nature is the Total Economic Value (TEV) framework (Pearce & Turner 1990, Pearce & Moran 1994; Figure 12).

Here we clarify the relationships between the review tasks and the TEV framework (Figure 13). Throughout, we use the definitions in Defra (2007) for each type of value.

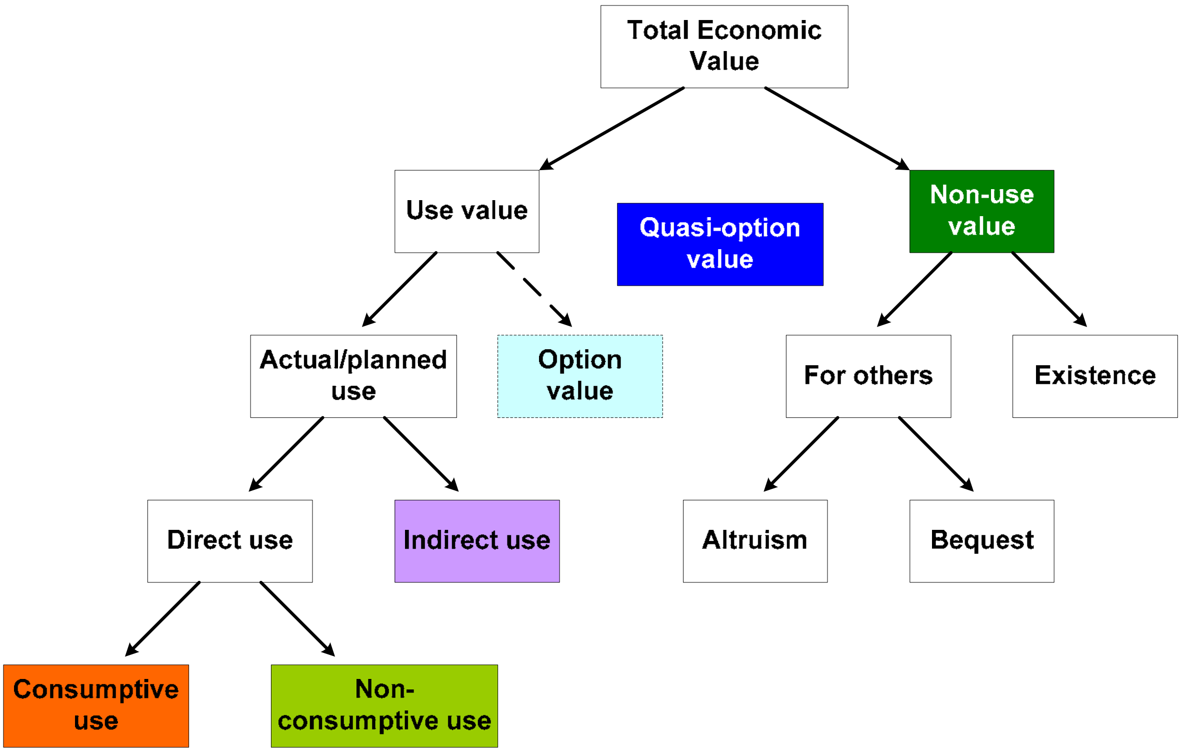

Figure 12. Total Economic Value framework (adapted from Defra 2007).

Direct consumptive use values

Direct consumptive use values are those obtained when individuals make actual or planned use of an ecosystem by extracting resources from it. Amongst the benefits/processes we are reviewing, this corresponds to those where there is direct harvesting from nature: livestock grazing and browsing in natural areas (theme 5), marine and inland fisheries (including recreational fishing; 6 and 7), harvesting of wild animal products (including recreational hunting; 8), harvesting of wild plant fibers (10), and harvesting of medicinal plants (11).

Direct non-consumptive use values

Direct non-consumptive use values are defined as being obtained when individuals make actual or planned use of an ecosystem without extracting any element from it. Nature tourism and other forms of outdoors recreation related to nature are traditionally considered within this category (e.g. Rockel & Kealy 1991), and we followed that approach (theme 12), although it should be acknowledged that these activities may have substantial impacts on the places visited, even if not harvesting resources from them.

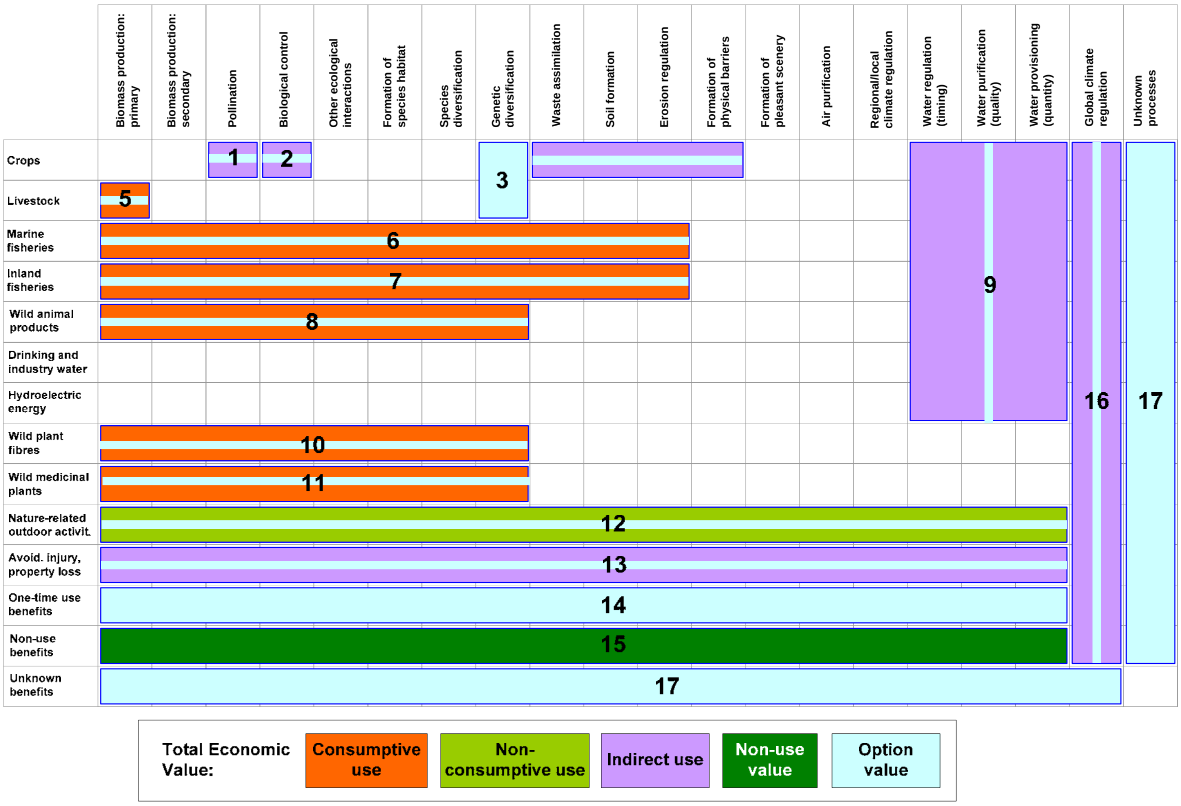

Figure 13. How each main review theme relates to the Total Economic Value framework.

Indirect use values

Indirect use values refer to situations when individuals benefit from ecosystem processes underlying the provision of direct benefits. For example, crop pollination (theme 1) is not used as a benefit per se, but as a process towards the increased provision of crop yields. We also included in this category: biological control of crop pests (2), maintenance of soil quality for crop production (4), fresh water provision and regulation (9), global climate regulation (9), and the regulation of natural hazards (13).

Option values

Option values are the values that people put on the option to use a resource in the future, even if they are not current users. These future uses may be either direct or indirect. Option values can be thought of as a form of insurance, but the economic literature is divided over whether such values are a true component of TEV (Freeman 1993). Strickly speaking, we should treat them as a separate concept related to a precautionary policy stance (theme 17)

Wholly within option values is genetic diversity of crops and livestock (theme 3), which can be considered a form of insurance in the case of future need to add genetic diversity to existing crop varieties or livestock breeds, for example in case of emergence of a new pest or disease. Also within option values are benefits from one-time use (14), the set of the ideas, chemical compounds, and images initially obtained from nature but subsequently propagated outside of it. Indeed, by definition the only way in which wild nature is valuable for these is through predictions of their future value. Data on the economic value of previously extracted ideas, compounds and images can be used to obtain estimates of likely future value, but not for measuring directly the value of existing wild nature (as those past ideas, compounds and images are now fully independent from nature).

All tasks covering use values (direct and indirect) listed previously can potentially have a component of option value. For example, not all known marine fish stocks need to be exploited today (even sustainably). One possibility is that some stocks may be exploited immediately and some left untouched as options for the future. These two types of values are likely to be monetised differently in the subsequent economic valuation.

As an example of option values for indirect use, natural water purification processes have no use value if nobody is using the water, for example in a region with no inhabitants. Nonetheless, we may want to attach value to those processes (and the ecosystems generating them) as providing options for future use.

Option values also apply to non-consumptive uses: we may want to set aside areas as valuable for future tourism, even if currently they are not used as such (e.g. deep sea systems may become attractive for tourism given future technological developments).

All considerations of resilience discussed within each task refer to option value, as they relate to the concern of ensuring future use. For example, the diversity of marine functional groups provides resilience to fisheries production (see Section 4.6), and so there is value in the conservation of such variety as a form of insurance against future changes.

Non-use value

Non-use value is derived simply from the knowledge that the natural environment is maintained. There are three main components:

- Bequest value: where individuals attach value from the fact that a given natural resource is passed on to future generations;

- Altruistic value: where individuals attach values to the availability of a given natural resource to others in the current generation;

- Existence value: where individuals attach value simply from the existence of a resource, even though the individual has no actual or planned use of it.

Theme 15 (non-use values) relates principally to existential non-use values.

Bequest values – which directly address issues of intergenerational equity – are implicit or explicit in other aspects of the Review on the Economics of Ecosystems and Biodiversity.

In particular, we recommend that the comparison between states of the world for a given resource (e.g. fisheries production, wild meat) is based on the capacity of ecosystems for the sustainable production of the resource, rather than based on current flows that may not be sustainable (see Section 2.3.4). In valuation terms, bequest values and the associated concerns about intergenerational equity will be explicitly incorporated in the decision of which discount rate to apply when valuing future resources, as was also the case with the Stern Review on the Economics of Climate Change (Stern 2007). They can also be addressed in the estimation of values (based on contingent valuation methods).

Altruistic values, on the other hand, include (amongst others) concerns for intragenerational equity. The emphasis placed on the spatial distribution of costs and benefits (see Section 2.2.4) is aimed at addressing such concerns, allowing for the redistribution of benefits for example through the establishment of payments for ecosystem services. Some of these are not altruistic, though, as such redistribution may directly benefit the 'donors' by increasing the likelihood of long-term provision of the resource or by reducing the deleterious social effects of inequity (for example in terms of security). Purely altruistic values nonetheless exist and can be integrated in the valuation component of the Review. But the only way to assign monetary values to non-use motivations is via so-called stated preference survey/choice-based methods. It is still an open research question as to how adequately values such as existence value can be meaningfully expressed in monetary terms (Heal 2000; Turner et al. 2003; Barbier 2007). To the extent that non-use values are not captured via economic analysis the TEV and the Total Ecosystem Value (TSV) will diverge with TEV < TSV.

3.9 Participants

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4 THEMATIC REVIEWS

Given time limitations, we were unable to fully develop all the themes listed in the previous section. As a result, themes have been analysed at different levels of depth, from fully developed thematic reviews that received expert support, to very brief overviews. The level of each review is indicated at the start of their respective section.

We tried to answer each of the following questions in each thematic review. However, for the least detailed reviews that was not possible. At ver least, we tried to obtain an answer to the questio: **Can we quantify and map the global provision of this benefit/process and how it might change?** We aimed to evaluate how far scientific knowledge is from being able to produce a spatially-explicit quantification of the production of the benefit (e.g. fisheries production) or of the function of the beneficial process (e.g., added value of wild pollination for crop yields), that can act as a basis for the economic valuation contrasting two states of the world. The information gathered was then used to guide the prioritisation of recommendations for Phase 2 (Section 4.19).

The questions are:

- ***Why is this benefit/process important for human wellbeing?*** A brief review of the information supporting the relevance of each benefit/process for human wellbeing. Any monetization presented here is to support the importance of the benefit to human welfare, rather than to quantify the importance of wild nature for the provision of the benefit. For example, the information that "floods affected 3.5 million people in Cambodia (with associated costs of US$145 million) in 2000" should not be interpreted to mean that conserving wild nature could have prevented those floods and therefore avoided those costs.

- ***What are the overall trends in the provision of this benefit/process?*** Review of state of knowledge on the trends of the provision of the benefit/function.

- ***How is the provision of this benefit/process affected by changes in wild nature?*** Here we describe the links between wild nature and the production of the process/benefit. This section is tailored to each review task, discussing the most relevant aspects of wild nature in each case. These may include 'diversity', such as species richness, and 'amount', such as biomass, as well as ecosystem condition. We specifically looked for information on which each of these components is particularly important for resilience in the provision of the benefit/process. And we specifically tried to understand what the relationship is between habitat area and the provision of the benefit/function, which is important for estimating the likely consequences of habitat loss.

- ***What are the main threats to the provision of this benefit/process?*** Information on the drivers of loss is key to the development of scenarios that are fit-for-purpose (Section 2.3.1). Indeed, these will only be accounted for in the Review on the Economics of Ecosystems and Biodiversity if they are different in business-as-usual and biodiversity-friendly states of the world. The information on these threats is also fundamental to calculating the costs associated it conservation, as costs will vary depending on the type and intensity of threat (see separate report on "Review of the

Costs of Conservation and Priorities for Action", by Bruner, Naidoo and Balmford, also an output of this Scoping the Science project).

- ***Are abrupt changes likely in the provision of this benefit/process?*** Here we were aiming to understand if there is evidence that the provision of this benefit/ process may be subject to thresholds/tipping points in the foreseeable future (in the questionnaire to experts we used 2025 as the foreseeable future data). By thresholds/tipping points we refer to a situation when a small (anthropogenic) change in nature may have disproportionate effects on the provision of the benefit/process. We drew from the information in the previous sections and on the opinion of experts to make a prediction of the likelihood of thresholds or tipping points. The result is a qualitative (and inevitably subjective) assessment of relative risk of abrupt changes.

- ***Can we quantify and map the global provision of this benefit/process and how it might change?*** Here we evaluate how far scientific knowledge is from being able to produce a spatially-explicit quantification of the production of the benefit (e.g. fisheries production) or of the function of the beneficial process (e.g., added value of wild pollination for crop yields), that can act as a basis for the economic valuation contrasting two states of the world. At one extreme, a good model already exists (a well-developed production function) and maps can easily be generated from available data. At the other extreme, no good model and/or good data exist that could form the basis of a global map. In most cases the state of knowledge was somewhere in between, with at least a first cut being possible within one year by building from existing studies. We also tried to obtain information on: the main gaps in data/knowledge; what would be needed from scenarios for the model proposed; who could potentially do the analysis; and what the required effort would be (in researchers-months).

- ***Insights for economic valuation*** The purpose of this section is to maximise the value of our work as a basis for the economic valuation, helping readers to interpret our recommendations by clarifying what our assumptions were on how the ecological modelling and the valuation fit.

- ***Some key resources*** Here we list key resources that we came across in our review and which are likely to prove useful in Phase 2.

- ***Participants*** The list of authors, contributors, reviewers and acknowledgements (see Annex 3).

4.1 Wild crop pollination

This section is a fully developed review, including contributions by experts in the field, who subsequently reviewed the text.

4.1.1 Why is wild crop pollination important for human wellbeing?

Many plant species benefit for their reproduction on animals that transport pollen between flowers. This biotic pollination is typically done by insects (particularly bees; also flies, beetles, moths, butterflies and wasps), but in some species is performed by vertebrates (particularly birds and bats). Pollination is therefore a key ecological process service, upon which both natural and agricultural systems depend (Nabhan & Buchmann 1997). While the extent to which staple food crops depend on pollinator services has been questioned by some (Ghazoul 2005), Klein and colleagues (2007) found that fruit or seed numbers or quality of 87 out of 115 leading global crops (representing up to 35% of the global food supply) were increased upon animal pollination.

In many agricultural systems, pollination is actively managed through the establishment of populations of domesticated pollinators, particularly the honeybee *Apis mellifera*. However, the importance of wild (i.e. unmanaged) pollinators for agricultural production is being increasingly recognised (e.g. Westerkamp & Gottsberger 2000; Kremen et al. 2007; Kremen & Chaplin-Kramer 2007). For a diversity of crops, it has been found that wild pollination increases the size and quality of harvests (Klein et al. 2007). Wild pollinators may also interact synergistically with managed bees to increase crop yields (Greenleaf & Kremen 2006). Furthermore, a diverse assemblage of native pollinators provides insurance against year-to-year population variability or loss of specific pollinator species (Kremen et al. 2002; Rickets 2004; Tscharntke et al. 2005), and might better serve flowers because of pollinator-specific spatial preferences to a flowering plant or crop field (Klein et al. 2008). Given current declines in populations of managed honeybees (Colony Collapse Disorder [Johnson 2007], and abandonment of beekeeping in regions affected by 'Africanization' of honeybees [Brosi et al. 2007]), the importance of wild pollination is likely to increase.

Estimating economic value is difficult and controversial, but the global value of wild and domestic pollination has been estimated at $120 billion per year (Costanza et al. 1997), while Losey & Vaughan (2006) estimated that wild pollinators alone are responsible for about $3 billion of fruits and vegetables produced in the United States.

4.1.2 What are the overall trends in the provision of wild crop pollination?

The Millennium Assessment indicated a low to medium certainty that pollination ecosystem services are declining (Duraiappah et al. 2005). Direct evidence for the decline in pollination services (i.e., evidence that global crop yields are being affected by a reduction in wild pollination) does not seem to exist, and would be difficult to obtain given that many other aspects of agricultural practices are changing simultaneously. Some indirect evidence exists based on reported declines in the abundance or area of occupancy of some wild pollinators, for example in North America (NRC 2007), and in Europe (Biesmeijer et al. 2006; Goulson et al. 2008). On the other hand, some species have extended their ranges (Ghazoul 2005). A continent-level assessment of pollinator declines is still lacking (Díaz et al. 2005). Most of the evidence for a decline in wild pollinator services is inferred from changes in land use known to affect pollinator communities, particularly declines in the extent and condition of available natural and semi-natural habitats, and the effects of agricultural intensification (e.g. Kremen et al. 2002). On the other hand, an increase in organic agriculture (Willer et al. 2008) may be increasing provisioning of pollination services (Holzschuh et al. 2008) in some parts of the world.

4.1.3 How is the provision of wild crop pollination affected by changes in wild nature?

Many crops are self-compatible to different degrees or wind pollinated, and therefore receive only small benefits from wild pollination services (Klein et al. 2007). For crops that rely heavily on biotic pollination, there are several ways in which changes in biodiversity can affect yields.

Wild pollinators often depend on natural or semi-natural habitats for the provisioning of nesting (e.g. tree cavities, suitable soil substrates) and floral resources that cannot be found within crop fields (Kremen et al. 2004). Consequently, the available area of natural habitat has a significant influence on pollinator species richness (Steffan-Dewenter 2003), abundance (Heard et al. 2007; Morandin et al. 2007), and pollinator community composition (Steffan-Dewenter et al. 2002; Brosi et al. 2007). Accordingly, habitat area in the neighbourhood of crop fields has been found to be strongly related to a direct measure of the pollination service measured here in terms of pollen deposition provided by bees (Kremen et al. 2004).

Besides area, the quality of the habitat, both in natural systems and in croplands, seems to be important for pollinator services, particularly the extent to which they provide nesting and floral resources (Klein et al. 2003a; Goulson et al. 2005; Potts et al. 2005). Not all pollinators are dependent on 'natural' habitats (Ricketts et al. 2004) and some are able to use resources within agricultural fields themselves (reviewed in Kremen et al. 2007) and therefore can even profit from agricultural management (Klein et al. 2002, Westphal et al. 2003).

There is clear evidence that wild pollination is strongly related to proximity to natural or semi-natural habitats. A recent quantitative review of 23 studies (Ricketts et al. 2008) found an exponential decay in pollinator richness and native pollinator visitation rate with distance to natural or semi-natural habitats. A decline in yield was less clear, possibly because few studies measured it directly. Visitation rate declined more steeply (dropping to half at 0.6 km) than richness (1.5 km). Despite the general exponential relationship found by Ricketts et al. (2008), it may be that slightly different relationships apply to different crops (e.g. linear for coffee, Klein et al. 2003a,b; log-linear for watermelon, Kremen et al. 2004) and to different types of pollinators (e.g. of different sizes; Klein et al. 2008).

Density of wild pollinators at the crop site is considered a good proxy of visitation rates (Kremen et al. 2007; Ricketts et al. 2008) and therefore key for crop production (Vázquez et al. 2005). This is confirmed by a few experimental case-studies (Roubik 2002), although results are potentially confounded by a positive relationship between numbers of individual visitors and diversity of pollinator species (Kremen & Chaplin-Kramer 2007). Interestingly, Klein et al. (2003b) found that numbers of individual visiting bees did not explain fruit set but that diversity of visitor species did. Three potential mechanisms may lead to a relationship between pollinator diversity and pollinaiton services, '*Sampling effect*' a greater chance of having a pollinator species that perfectly fits the flower morphology leading to more reliable pollination; (2) '*Niche complementarity'* all receptive flowers over an extended blooming period receive optimal pollination service; (3) '*Functional facilitation'* synergistic interspecific interactions with domestic honey bees (Greenleaf & Kremen 2006, Klein et al. 2008).

The identity of the pollinator species matters, as different crops benefit from pollination by different species (Klein et al. 2007). Some crops have specific pollinator requirements, either certain guilds or certain flowering times, and so abundance of pollinator individuals may not translate into high pollination services if they belong to the wrong guilds (so they visit flowers but are inefficient at actually pollinating them), if their abundance is out of synchrony with crop mass flowering (e.g. almonds in California typically flower before many wild bees have built up significant populations) (Ricketts et al. 2006; Kremen et al. 2008; Klein & Kremen unpublished), or if honey bees are attracted to another flowering crop adjacent to the target crop field but other pollinating species still prefer to forage at the target crop e.g. for alfalfa pollination.Accordingly, crops with a narrow range of specialised pollinators are more likely to experience pollinator shortage when grown in highly modified landscapes (Klein et al. 2007). Overall, functional diversity (diversity of functional traits) may be more important to crop yield than either pure abundance or species richness (Klein et al. 2008). Consequently, the order in which species are lost is likely to affect pollination services differentially (Larsen et al. 2005). Unfortunately, large-bodied bee and beetle species seem to be both most extinction-prone and most functionally efficient, their loss contributing to rapid functional loss (Larsen et al. 2005). A current NCEAS analysis into body size, nesting behaviour, trophic specialisation and sociality will shed more light on this issue.

Pollinator diversity has been found to improve resilience of crop production, by buffering pollination against asynchronous fluctuations of bee abundances between years, including temporal variation in the relative abundance within native species (Kremen et al. 2002) and a sharp decline in domestic honeybees (Ricketts 2004, Winfree et al. 2007b). Given that different species are differentially effective as pollinators (both within and among crops), managing for bee diversity could meet the pollination requirements of a greater number of crops, provide insurance in the event of shortages of any specific pollinator (managed or unmanaged), and provide options for new or alternative crops (Kremen et al. 2002). Furthermore, different taxa are likely to respond differently to landscape isolation and habitat characteristics (e.g. Steffan-Dewenter et al. 2002; Klein et al. 2003a,b), so a diversity of pollinator taxa may help to reduce the effects of land use change on pollination services (Ricketts et al. 2008). The current precipitous declines in managed honeybees in the United States due to Colony Collapse Disorder are making clear the dangers of relying on a single pollinator species (Johnson 2007; NRC 2007).

Relationship between habitat area and pollination

Most services from wild pollinators take place near the interface between natural/semi-natural habitats and crop fields (Ricketts et al. 2008), and so landscape patterns have a strong influence on pollination services.

There is strong evidence that both richness of pollinator species and rates of visitation by native pollinators at crop sites decline quickly with distance to natural/semi-natural habitat (Ricketts et al. 2008; Figure 14a) and so crop yields are also expected to decline with distance.

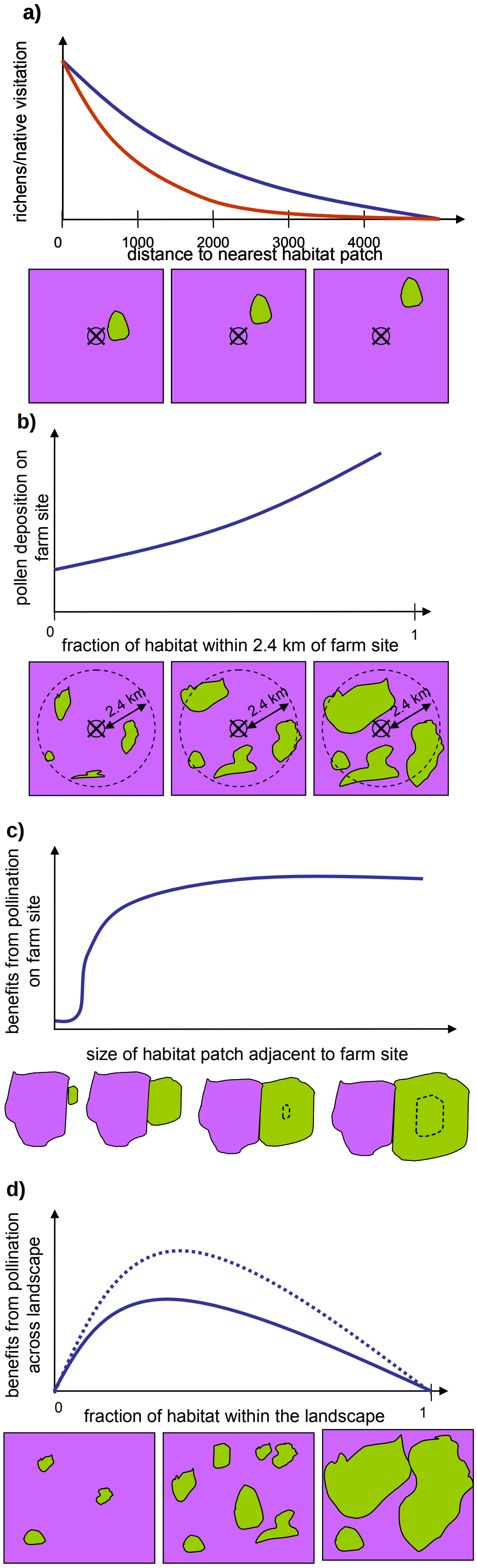

Figure 14. Four aspects of the spatial relationship between habitat area and provisioning of pollination services.

a) Both species richness (blue) and the rates of visitation by native pollinators (red) have been found to decline with distance to nearest habitat patch (Ricketts et al. 2008), and so presumably the benefits from pollination (in terms of crop yields) should also show a strong relationship with distance.

b) Given a particular farm site (cross), pollination increases with the fraction of habitat (green) within 2.4 km of the site's neighbourhood (after Kremen et al. 2004).

c) Our prediction for the relationship between benefits obtained from pollination at the farm and the size of the habitat patch (green) adjacent to a particular farm site (purple): very small habitat patches (smaller than a minimum viable area for pollinators) may not have any effect; afterwards, pollination is predicted to increase rapidly with area, but then to stabilise as further increases in habitat happen correspond only to "core" area (dashed).

d) Predicted relationship between the fraction of overall habitat area (green) within a landscape (square) and overall pollination services for the crop (purple; Morandin & Winston 2006): pollination values are zero if there is zero habitat for pollinators; pollination then increases as habitat area increases, but declines again as habitat expansion results in a decline in crop land at the interface with the habitat; for 100% natural habitat, pollination for crops has zero value. For a given overall area of natural/semi-natural habitat, pollination benefits may be higher (dashed line) or lower (solid line) depending on the spatial arrangement of the habitat patches.

The fraction of natural/semi natural habitat area in the neighbourhood of crop fields has been found to be strongly related to a direct measure of the pollination service, pollen deposition (Kremen et al. 2004; Figure 14b). Concrete studies on species–area relationships for pollinators (particularly in tropical forest fragments; Priess et al. 2007) are lacking, but patch area of natural habitat is likely to influence species richness and abundance of pollinator communities, as larger habitat fragments in many cases show higher species richness and density (Fahrig 2003).

We therefore predict that, after a minimum threshold of habitat patch size, pollination at a given crop site increases with the size of the adjacent habitat patch, both because the patch itself becomes larger (and potentially has higher pollinator diversity and abundance), and because a larger fraction of the site's neighbourhood becomes natural/semi-natural habitat (Figure 14c). This increase, however, is only expected up to a certain point, followed by stabilisation, as additional area increases take place too far from crop fields to produce pollination services (Figure 14c; however, Brosi et al. [2007] found non-significant relationships between bee abundance or diversity and size of neighbouring forest patch). If so, the overall benefits from pollination within a given landscape are expected to peak at intermediate levels of natural/semi-natural habitat cover, at the point where the spatial interface with crops is maximised (Morandin & Winston 2006; Figure 14d). The spatial arrangement of the habitat patches within the landscape is likely to matter; modelling (Keitt, in review) suggests that pollination benefits are optimised for maximized by providing islands of nesting habitat where inter-island distance matches mean foraging and dispersal distances of wild pollinators. Hence, strategically creating new patches of habitat in areas of intensive agriculture (for the crops that benefit from wild pollination) can increase services from wild pollination significantly. The size of these islands is likely to be important, though, as pollinator communities may become unviable in very small fragments (Keanrs et al. 1998).

For some pollinator species, agricultural areas (the crop fields themselves) are also important habitats, providing nesting and floral resources. In this case, even small natural or semi-natural habitats (too small to support pollinator populations by themselves), can bolster populations of important pollinators.

4.1.4 What are the main threats to the provisioning of wild crop pollination?

<u>Loss of suitable habitat</u> is recognised as a key driver of declines in wild pollination (e.g. Steffan-Dewenter 2003; Ricketts et al. 2004; Morandin & Winston 2006; Priess et al. 2007). <u>Habitat degradation</u>, for example through agricultural intensification, leads to scarcity in critical floral and nesting resources for many species. The <u>use of chemicals</u> (such as insecticides, herbicides and fertilisers) in conventional agriculture has been found to reduce populations of pollinators (Kearns et al. 1998). <u>Climate change</u> is emerging as a potentially new threat, as phenological shifts may result in the disruption of plant-pollinator interactions (Memmott et al. 2007). <u>Invasive non-native species</u> (including plant, mammals and insects) pose additional threats (Cole et al. 1992; Kearns et at. 1998).

4.1.5 Are abrupt changes likely in the provision of wild crop pollination?

It is possible that a threshold in pollinator species/functional diversity, exists below which pollination services become too scarce or too unstable (Klein et al. 2007). Such a tipping point might occur when, at a landscape context, sufficient habitat is destroyed that the next marginal change causes a population crash in multiple pollinators. Modelling supports this prediction (Morandin & Winston 2006; also Keitt in review).

Alternatively, a threshold in habitat loss may lead to the collapse of particularly important pollinators, leading to a broader collapse in pollination services (pollinator keystones). Supporting this prediction, Larsen et al. (2005) found that large-bodied pollinators tended to be both most extinction-prone and most functionally efficient, contributing to rapid functional loss with habitat loss.

Empirical data on changes in the provision of pollination services to agriculture are still very sparse. However, studies in California clearly indicate that there is indeed a tipping point in the provision of such services from wild pollinators (Kremen et al. 2002, 2004; Larsen et al. 2005; Greenleaf & Kremen 2006; Klein et al in prep; Chaplin-Kramer et al. in prep). From this work we can extrapolate that in most areas of California's agricultural region such threshold has already been surpassed and there is now little wild pollination. This makes crop production substantially to entirely reliant on managed honey bees, whose numbers show strong within-year variation and an overall declining trend due to diseases and other factors (NRC 2006). The studies in California are illuminating because the region has very strong gradients, ranging from small farms surrounded by natural habitat to some of the most intensively managed agricultural landscapes in the world. Given global trends in agricultural intensification, the California region is therefore very informative of the possible future changes in the provision of pollination services worldwide.

A model currently being developed by the NCEAS working group could be used in the future to explore such thresholds.

Overall, we predict that there is a medium to high probability that the provisioning of wild pollination services is likely to be subject to thresholds/tipping points in the foreseeable future (by 2025), with a very high probability that such thresholds will happen in regions of very intensively managed agriculture.

What this will mean for actual crop productivity is less clear as there are substitutes for wild pollinators and for pollinator-dependent crops.

4.1.6 Can we quantify and map the global provision of this wild crop pollination and how it might change?

State of knowledge and data availability

Wild pollination is a valuable ecosystem process because it can increase the yield of economically important crops. Phase II of the Review on the Economics of Ecosystems and Biodiversity will attempt to assess how such services are affected by biodiversity loss (e.g. by deforestation leading to declines in habitat for wild pollinators). An ideal answer to this would involve producing a global map of pollination services (measured in units reflecting the contribution to increased yields per ha per year). Such a map would be generated for different scenarios of possible global changes (e.g. what if all remaining Andean cloud forest is lost?), and differences in pollination services would be contrasted to evaluate the extent to which pollination is affected by those changes. The economic valuation would be done on top of this ‘biophysical model’, accounting for local/regional variation in crop yields as well as for global aspects such as changes in commodity market prices.

Although a biophysical model of delivery of pollination services is not fully in use as yet, two major efforts are currently being reconciled with the aim to have such a model in operation by October 2008:

- Kremen et al. (2007) have created a conceptual model of ‘mobile agent-based ecosystem services’, which forms the basis of work carried out by the NCEAS Working Group on pollination services. The model includes interactions and feedbacks among policies affecting land use, market forces, and the biology of organisms involved. The group is conducting quantitative syntheses of the key relationships within this model, which has already been done for the dependence of world crops on pollinators by Klein et al. (2007) and for the relationship between distance from natural habitat and pollination services by Ricketts et al. (2008), and is underway for impacts of disturbance on bee abundance and diversity (led by Rae Winfree), and pollinator functional traits (led by Neal Williams). This component considers the sensitivity of pollinator species based on life history and other traits (e.g., body size, trophic specialization, nesting habit) and explores the consistency of resulting functional group responses among different types of disturbance. From this, a quantitative model will be built.

- The Natural Capital Project is developing tools for modelling and mapping the delivery, distribution, and economic value of ecosystem services and biodiversity. The Project’s pollination module is developing a biophysical model for pollinator abundance on crops in a landscape. It uses information on key pollinators, the availability and location of their nesting and floral resources on the landscape, and their flight ranges, to predict their abundances on crops. The result is a map of relative pollinator abundance in agricultural fields or pixels. Predicted effects on yield or quality of crops have not yet been calculated, but the group plan to do so.

Recently, these two efforts have merged, with key leaders of the NCEAS group helping to devise the Natural Capital model. By October 2008, they plan to have the model formulated and validated on three landscapes by comparing model predictions against empirical data.

In a related study, Priess and colleagues (2007) used empirical relationships on the effects of forest distance on both pollinator diversity and fruit set of coffee (based on the results of field experiments carried out by Klein et al. 2003a,b,c) to estimate future changes in pollination services for different land use scenarios in Sulawesi, Indonesia.

These efforts indicate that it is possible to develop a landscape-scale spatially-explicit model based on an empirical understanding of how different crops benefit from wild pollination (Klein et al. 2007) and how pollination decays with distance from natural/semi-natural habitat (Ricketts et al. 2008), ideally complemented with more detailed data on the identity/guild of the pollinators, the specific pollinator needs of crops, and the value of modified habitats (including the agricultural lands themselves) for nesting and floral resources. Currently available data are far from perfect, but allow for broad generalisations, and expert opinion could be used to fill some data gaps.

The key challenge would reside in extrapolating these models for application at a global scale. Indeed, global maps would be needed not only on the distribution of the relevant natural/semi-natural habitats, but also on the distribution of the relevant crops (that benefit from pollination). Furthermore, these maps would need to have sufficient spatial detail to be relevant to the fine-scale spatial dynamics that governs pollination, as foraging ranges of pollinators have been empirically shown to be typically less than 1 km. Such maps currently do not exist. The main limiting factor is probably the unavailability of a detailed map of global crops distribution, with information for individual crops and with sufficiently fine spatial resolution. But the production of fine scale maps of natural/semi-natural habitat (with information on their relative value for pollination services) is also not trivial. Such maps can be generated for particular regions (e.g. Natural Capital model now being applied to the Valuing the Arc project in Tanzania) but certainly not for the world within the time frame of the Review.

Two possibilities exist in the short-term for the development of a first-cut pollination model at the global scale:

- A probabilistic approach, whereby the probability of each pixel being under a crop is estimated from data on the pixel's suitability for crops (Fisher et al. 2000) as well as broader crop maps (Cassman et al. 2005; Ramankutty et al. 2008). The expected benefits from pollination on each pixel would then be estimated using a generalised biophysical model.
- A sampling approach, in which detailed models are developed for each of a set of representative landscapes, applied using fine-scale maps of crops and habitats for those landscapes. Expected global benefits from pollination, and predicted changes for different scenarios, would then be obtained by extrapolation using global crop data (Cassman et al. 2005; Ramankutty et al. 2008).

We believe the second approach is the most feasible as well as the most informative.The probabilistic approach would allow for the consideration of uncertainty of what the relevant crops are in each area, but the spatial resolution would not be sufficiently fine for the adequate incorporation of the relevant spatial dynamics. The sampling approach would benefit from finer spatial data on the distribution of both crops and natural/semi-natural habitats, therefore resulting in substantially more reliable predictions. A choice of adequate sample landscapes (including temperate and tropical regions, containing strong gradients in agricultural intensification) would allow for a first-cut extrapolation to the global scale. We therefore recommend that a global evaluation of the effects of biodiversity loss and ecosystem change on pollination services builds from the work currently being done by the Natural Capital Project, which is focusing on four landscapes: California, Tanzania. China and Hawaii.

Even at the finer landscape scale, it is important to keep in mind the limitations of any model that can possibly be developed and applied in the near future. Smaller plots of agricultural land in close proximity to natural biodiversity, which are important for pollination services, are unfortunately likely to be lost due to coarse resolution of the data. Small field sizes and mixed cropping systems make Africa particularly vulnerable to this. A model in the lines currently being developed and proposed here would be particularly informative for the effects of changes in wild pollination services associated with changes in land cover, an less useful to predict variation in other factors. Effects of variation in pesticide use can somehow be incorporated in the classification of crop types (as intensive crops tend to make more use of pesticides) but only up to a certain extent (as, within any given crop type, level of pesticide use depends on individual farmer decisions). The model would be unlikely to capture changes dues to invasive species, unless the value of invasive floral resources for pollinators is know along with habitat associations of these plant species (one example for which this could be known is *Centaurea solstitialis,* yellow star thistle).

Adequacy of scenarios

A probabilistic approach would require the generation of predicted global maps of land cover, including predictions for the distributions of particular sets of crops, at a fine spatial scale. Approaches such as those developed by the ATEAM Project (Advanced Terrestrial Ecosystem Analysis and Modelling) for Europe (Schröter et al. 2004; Rounsevell et al. 2006) could potentially be expanded to the global scale (within 1 year?)

A sampling approach would require the generation of detailed maps of predicted land cover for a set of representative regions, for each of the chosen scenarios. While these are not trivial to generate either, results would be more realistic and a finer scale than for when produced at a global scale. Approaches such as those used by Priess et al. (2007) seem to have potential. They used ‘generalized cellular automata’ to allocate future land use based on biophysical suitability (climate, soils, topography), allocation factors (e.g. distance to the next river, land use on adjacent cells, preferred walking distance to field), demography (population growth rate, migration), and the land use strategies of farmers (e.g. moderate or high intensity agriculture, forest use) and other restrictions like the protection status of the area. The extrapolation from study landscapes to the global scale would require linking global coarse scenarios for changes in land use distribution with the finer landscape scenarios. This could potentially draw on the methods and approaches that have been developed for downscaling land use scenarios (e.g. Verburg et al. 2006). Model validation is fundamental, by matching modelled predictions to exisiting data from targeted areas. A potential approach would be to select existing data sets, while another would be to prioritize study of pollination service at particular target areas for which other data are known.

A fundamental variable in the scenarios is the extent to which managed pollinators (particularly honeybees) are made available in crops. The scenarios being contrasted would need to consider that this is the same in both cases. Possible extreme scenarios include, on the one hand, a situation in which wild pollination becomes the only available mechanism for crops that depend on biotic pollination (in the remote possibility that Colony Collapse Disorder eliminates all managed honeybees) and, on the other hand, a situation in which managed pollination is provided whenever useful and the value of wild pollination is simply by complementing (or interacting with) managed pollinators. Reality is somewhere in between, and so scenarios could also consider an intermediate situation in which particular crops, from particular regions, benefit from managed pollination and others do not. The key consideration would be that any pair of scenarios being compared would need to consider matching levels of managed pollination. The same applies for levels of pesticide use.

What can be done in Phase II? At what cost? By whom?

We predict that it will be possible to produce within one year a first-cut global model to evaluate how global crop pollination services are affected by biodiversity loss (changes in land cover). We recommend building from ongoing landscape-scale assessments currently being developed by the Natural Capital project. In our opinion, the research group better positioned to lead these efforts is the ongoing collaboration between the NCEAS Working Group on pollination (which includes the lead of the ALARM project) and the Natural Capital Project[1].

A dedicated post-doc would be required (12 researcher-months), plus resources to involve other experts, perhaps through two NCEAS-style workshops at the beginning and middle of the project.

4.1.7 Insights for economic valuation

Appropriate scenarios can generate maps of different states of the world, varying in their spatial patterns of distribution of crops and natural/semi-natural habitat. A model along the lines of those described above can then be used to create surfaces of pollination services (measured in changes in the yields of particular crops) for each state of the world. The contrast between these two maps will indicate areas where crop yields are expected to change (either increase or decrease). Data on the market value of these crops (adjusted for possible variations in global crop prices arising from differences in the scenarios) could then be used to monetise the map of variation in crop yields.

4.1.8 Some key resources

- The National Center for Ecological Analysis and Synthesis (NCEAS) Working Group on "Restoring an ecosystem service to degraded landscapes: native bees and crop pollination" is synthesising and analysing data on bee populations, pollinator communities and pollination services across agro-natural landscapes, including the development of models for the management of pollination services. The Group is led by Claire Kremen and Neal Williams and composed of c. 20 experts, including amongst others Alexandra-Maria Klein, Ingolf Steffan-Dewenter, Rachael Winfree and Taylor Ricketts.
- The Natural Capital Project, a joint venture among The Woods Institute for the Environment at Stanford University, The Nature Conservancy, and the World Wildlife Fund, is developing tools for modelling and mapping the delivery, distribution, and economic value of ecosystem services and biodiversity. It includes a pollination module (led by Taylor Ricketts), which is developing a biophysical model for pollinator abundance on crops in a landscape.
- Modelling for the NCEAS Working Group and the Natural Capital pollination module is being led by Eric Lonsdorf.

[1] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

- The EU-funded ALARM (Assessing Large-scale Risks for Biodiversity with Tested Methods) project, led by Ingolf Steffan-Dewenter, includes a research module on pollination investigating the risks resulting from, and rates and extent of, loss of pollinators. With a particular emphasis on Europe, this 5-year (2004-2009) project is developing predictive models for pollinator loss and consequent risks.
- The Millennium Assessment (Cassman et al. 2005) used data on rain-fed and irrigation croplands obtained from a cropland-focused reinterpretation of GLCCDv2, based on methods described in Wood et al. (2000).
- Ramankutty et al. (2008) developed a new global data set of croplands and pastures circa 2000, at 5 min (~10 km) resolution, by combining agricultural inventory data and satellite-derived (MODIS-derived and GLC2000) land cover data.
- The Global Agro-Ecological Zones (GAEZ) assessment created a global database of estimated yields for various crops by matching the soil type, terrain, and climate of grid cells with productivity levels for 154 land-use types documented in places with similar characteristics (Fisher et al. 2002). The 5-arc-minute grid generated by the GAEZ assessment contains measures of agricultural suitability, assigned to eight categories based on estimated productivity levels as a percentage of maximum observed yields for a given crop.
- The International Pollinators Initiative, established by the Fifth Conference of Parties to the Convention on Biological Diversity in 2000, declared an 'urgent need to address the issue of worldwide decline of pollinator diversity'.

4.1.9 Participants

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4.2 Biological control of crop pests

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.2.1 Why is biological control of crop pests important for human wellbeing?

Biological control is the process by which one organism reduces the population density of another organism, for example through predation or parasitism. In nature, most organisms are consumed by other organisms, which in many cases leads to drastic reductions in the population of the prey species; in biological control of crop pests, man exploits this 'natural control' to suppress the numbers of pest species (Bale et al. 2008).

Here we focus on the value of biological control for reducing the effects of pests in agricultural systems, and therefore contributing to increases (or preventing decreases) in crop yields. We recommend a focus on biological control of crop pests mainly because of the availability of data (e.g. Losey & Vaughan 2006) and the huge value of crop plants to human well-being.

Agricultural pests cause significant economic losses worldwide. Globally, more than 40% of food production is being lost to insect pests, plant pathogens, and weeds, despite the application of more than 3 billion kilograms of pesticides to crops, plus other means of control (Pimentel 2008). In the US alone, it is estimated that more than US$18 billion are lost due to insect damage (including more than US$ 3 billion spent on insecticides), of which about 40% ate attributed to native species and the remaining to exotic pests (Losey & Vaughan 2006). These values, however, would be much higher if biological control was not in place. Losey & Vaughan (2006) estimate that 65% of the financial cost of potential damage by pest species is being suppressed in the US, with a total value of pest control by native ecosystems around US$13.6 billion/y. Through a predator removal experiment, Östman et al. (2003) showed that the presence of natural enemies increased barley yields 303 kg/ha, preventing 52% of yield loss due to aphids.

Biological control of crop pests may be totally natural, without direct intervention from man, or it may be enhanced through biological control interventions. Frequently, the term biological control refers to the latter only, when an organism is used *by man* to reduce the population density of another organism (Bale et al. 2008). Here, however, we are interested in the contribution of 'wild nature' to both of these mechanisms, and particularly to the natural control that flows from unmanaged ecosystems.

Natural control of plant pests is provided by generalist and specialist predators and parasitoids, including birds, bats, spiders, ladybugs, mantis, flies, and wasps, as well as entomopathogenic fungi. In the short-term, this process suppresses pest damage and improves yields, while in the long-term maintains an ecological equilibrium that prevents herbivore insects from reaching pest status (Zhang et al. 2007 and references therein).

In biological control interventions, pest control is enhanced through a diversity of approaches (Bale et al. 2008 and references therein):

- Inoculative control is used mainly against 'exotic' pests that have become established in new countries or regions of the world and involves the collection and release of natural enemies from the country or region of origin of the pest. This works better with perennial crops (fruit plantations and forests), where the persistence of the agro-ecosystem enables the interactions between pest and natural enemy to become fully established over a period of time.

- Augmentative control involves periodically reintroducing natural enemies that are usually commercially produced for that purpose. It is particularly indicated for short-term annual crops, as populations of natural enemy species may not persist between the crop production cycles. This includes both 'inundation' (the mass production and release of large numbers of the control agent) and 'seasonal augmentation' (where the natural enemy population is built-up through successive releases during the same growing season).

- Conservation control (see Landis et al. 2000 for a review) refers to the manipulation of the environment to increase the effectiveness of indigenous predators and parasitoids, usually against native pests. Various measures are implemented to enhance the abundance or activity of the natural enemies, including manipulation of the crop microclimate, creation of overwintering refuges (like 'beetle banks'), increasing the availability of alternative hosts and prey, and providing essential food resources such as flowers for adult parasitoids and aphidophagous hoverflies.

Biological control reduces rather than eradicates pests, such that the pest and natural enemy remain in the agro-ecosystem at low densities. A number of important pests can be kept at a low population density by biological control over long periods of time. In other cases, populations of pests are significantly reduced by natural enemies, but repeated additional methods are needed to achieve an adequate level of control (e.g., resistant plants, cultural techniques, physical barriers, semiochemicals and, as a last resort, the use of selective chemicals) – this is the fundamental philosophy of integrated pest management (Bale et al. 2008).

Here we are particularly interested in the value of wild nature to natural (unmanaged) pest control and to (managed) conservation control. However, wild nature also contributes to inoculative and augmentative pest control through the provision of natural enemies in the regions of origin of exotic pests (see Section 4.14).

4.2.2 What are the overall trends in the biological control of crop pests?

The Millennium Assessment indicated a medium to high certainty that pest regulation services are declining (MEA et al. 2005).

Agricultural intensification (with associated reduction in the landscape complexity, increase in pesticide use) and loss of crop genetic diversity increase the exposure of crops to major disease outbreaks and therefore the importance of biological control.

At the same time, improvement in artificial pest control (through the use of pesticides) and the development of pest-resistant crops reduce the need for biological control. However, pesticide use has an associated array of problems (Pimentel 2008): cost – pesticides are expensive and need to be applied frequently; public health and environmental impacts, with more than 26 million people poisoned annually from pesticides, resulting in more than

220,000 deaths; and environmental impacts such as losses to non-target species, including natural enemies of crop pests. Indeed, pesticides can do be more damage to natural enemy populations than to pests, thereby exacerbating the very problem they were designed to solve (Kremen & Chaplin-Kramer 2007). Furthermore, pesticide-resistant organisms evolve rapidly (e.g. insects usually evolve resistance within about a decade of the introduction of an insecticide; Palumbi 2001), while there are no known cases of evolved resistance of natural enemies (Bale et al. 2008). The value of natural control should therefore not be underestimated or presumed to be declining, even less so as demand for organic products is increasing (Willer & Yussefi 2008).

Data on the populations of biological control agents are scarce but the trends are presumed to be negative due to habitat transformation associated with agricultural intensification (agricultural expansion, enlargement of field size, removal of non-crop habitat resulting in a loss of natural landscape features required for maintaining their populations) and increasing pesticide use. On the other hand, the increase in organic farming worldwide may help to reverse this trend (Bengtsson et al. 2005; Willer et al. 2008).

4.2.3 How is the provision of biological control of crop pests affected by changes in wild nature?

Biological control seems to be greatly determined by the abundance of natural enemies, and indeed this is often used as a measure of the intensity of biological control (e.g. studies reviewed in Bianchi et al. 2006 and Kremen & Chaplin-Kramer 2007). Several studies have demonstrated increases in predation rate with predator abundance, while for parasitoids the majority of evidence for increased densities affecting pest control comes from exotic introduction examples (see references in Kremen & Chaplin-Kramer 2007). However, enhanced enemy abundance or richness does not necessarily guarantee improved pest control (see below).

The diversity of natural enemies also seems to improve biological control, through a variety of mechanisms, including: species complementarity, when more than one type of predator or parasitoid adds to the control of a pest species; the sampling effect, whereby a particularly effective natural enemy is more likely by chance alone to occur when more species are present; redundancy, where more species will buffer against disturbance or ecosystem change; and idiosyncrasy, when the whole is greater than the sum of the parts due to interactions among species (Tscharntke et al. 2005; and see Kremen & Chaplin-Kramer 2007 for a review). However, these expected benefits of diversity to pest control may be countered by antagonistic effects between different natural enemy species, such as competition and intraguild predation (Kremen & Chaplin-Kramer 2007). On the other hand, intraguild predation could help supplement natural enemy diets when pest densities are low, potentially maintaining a higher natural enemy population size more capable of controlling sudden pest outbreaks. The overall effect seems to be context-dependent, with it being important to consider both the diversity and the composition of natural enemy assemblages (Wilby & Thomas 2002; Kremen 2005), as well as the characteristics and ecology of predators, and the identity of their prey species (Kean et al. 2003; Wilby et al. 2005). While species richness will be most important for the control of some pests, the presence of a particular predator or parasite will be of utmost importance for others (Wilby & Thomas 2002). The provision of biological control is crucially determined by the identity of biological control organisms, which may even have a stronger effect than diversity (e.g. Straub & Snyder 2006).

Non-crop habitats are fundamental for providing habitat where many biological control agents (predators, parasitoids) mate, reproduce, and overwinter (Zhang et al. 2007) as well as additional, complementary, food (Kremen & Chaplin-Kramer 2007). The quality of the habitat can make a significant difference (e.g. Thies et al. 2005; Tylianakis et al. 2007) and this can be improved through habitat management (Landis et al. 2000).

Landscape diversity or complexity is generally positively associated with the abundance and species richness of natural enemies and may be the most crucial factor driving biological control services (for reviews see Bianchi et al. 2006; Kremen & Chaplin-Kramer 2007; Tscharntke et al. 2007). Landscape complexity can be quantified by a variety of measures, including the proportional area of natural habitat, distance to natural habitat or perimeter to area ratio of the field (Kremen & Chaplin-Kramer 2007), and beta diversity (species turnover) across natural habitat patches (Tscharntke et al. 1998). Landscape connectivity (Tscharntke et al. 2007) and age of natural habitat (Tscharntke et al. 1998) also seem to increase biological control. While there is evidence that increased landscape complexity is correlated with diversity and abundance of natural enemy populations, this does not necessarily translate into improved pest control, as pest densities may also respond positively to landscape complexity (Thies et al. 2005). For example, while 75% of the studies in Bianchi et al. (2006) found increased abundance of natural enemies in more complex landscapes, only 45% showed lower pest pressure. Very few studies measured actual predation rates or yields through controlled experiments (e.g. Östman et al. 2003; Williams-Guillen et al. 2008).

The proximity to semi-natural habitats greatly influences biological control by increasing natural enemy abundance, diversity, predator/prey ratios and importantly predation risk or parasitism for crop pests (Tscharntke et al. 2007; Bianchi et al. 2006). The distance of habitats over which parasitism is enhanced varies from tens to hundreds of metres, depending on parasitoid species (Tscharntke et al. 2005). Although there are many examples of parasitism rates being higher at the edges of fields than the middle, they are also frequently found to be unrelated, so that one cannot be sure whether the number of natural enemies or their movement is limiting biological control (Tscharntke et al. 2007). Unfortunately, there are few experimental studies (e.g. Tscharntke et al. 1998) quantifying the relationship between distance from semi-natural habitat and biological control, and no work has investigated the shape of a general relationship, as has been done for pollination (Ricketts et al. 2008).

Habitat fragmentation may also affect biological control (With et al. 2002), with a greater effect on specialist parasitoids than generalist predators (Tscharntke et al. 2007), such that generalist and mobile enemies may even profit from the high primary productivity of crops at the landscape scale, their abundance compensating for losses in diversity.

Natural habitat may actually have negative effects (‘ecosystem disservices’) on biological control, particularly as a source of pests (Thies et al. 2005; Kremen & Chaplin-Kremer 2007). On the other hand, some studies have shown a greater effect of habitat loss on the predators or parasitoids than on their herbivore prey and so the disservice of pest provision may be negated by the benefits of natural enemies in large and well-connected habitats (Kremen & Chaplin-Kremer 2007 and references therein).

Crop diversity can reduce the incidence of pests and diseases, through both the increase in the genetic diversity of individual crops (e.g. Zhu et al. 2000) and the increase in the diversity of crop species in multi- or poly-cropping systems (e.g. Sutherland & Samu 2000).

Relationship between habitat area and biological control

Work quantifying the effect of the area of natural habitat on the provision of natural enemies and biological control services is lacking (Kremen & Chaplin-Kramer 2007), although results suggest that the effect may be greater for parasitoids/predators than their prey (e.g. Kruess & Tscharntke 2000). The question is complicated by the different dispersal abilities of insect species: while a large uncultivated area may provide a large source of natural enemy individuals and species, the same area distributed in smaller fragments among cultivated fields would allow greater dispersal (Tscharntke et al. 2007). The optimal resolution to this tradeoff will also depend on the attributes of the pests and the crops, as ephemeral annual crops will benefit most from natural enemies moving quickly into the area (Bianchi et al. 2006) whilst a larger refuge of diverse natural enemies might be more suitable for perennial crops. Equally, highly mobile pests may be best controlled by natural enemies in a landscape of well-connected natural habitats (Tscharntke et al. 2007).

4.2.4 What are the main threats to the provision of biological control of crop pests?

The main threat to the provision of biological control seems to be habitat loss and degradation, particularly associated with agricultural intensification. This results in landscape simplification, and natural habitat fragmentation leading to losses in the diversity and abundance of natural enemies. Intensive agricultural management practices threaten natural enemy populations (Naylor & Ehrlich 1997). Ironically, pesticide use has had severe detrimental effects on natural enemies, often greater than on the target pests, and may even result in the emergence of new pests (Kremen & Chaplin-Kramer 2007). Natural enemy species will also be under threat from invasion of foreign species and the disruptive effects of climate change.

4.2.5 Are abrupt changes likely in the provision of this benefit/process?

The relationship between densities of natural enemies and the biological control services they provide is unlikely to be linear (Losey & Vaughan 2006). From the complexity of interactions and non-linearity of the relationship between diversity and function mentioned above, it seems likely that a function may decline disproportionately when a tipping point in natural enemy diversity is passed. However, there does not seem to be empirical evidence in support of this. Thresholds in landscape structure (interpatch distance) have been shown to impact the aggregative response of predators, thereby generating a similar threshold in pest populations (With et al. 2002). The importance of natural enemy assemblage composition in some instances of biological control (Shennan 2008) indicates that changes in composition can lead to disproportionately large, irreversible and often negative shifts in ecosystem services (Díaz et al. 2005).

Overall, we predict that there is a medium to high probability that the provisioning of biological control is subject to thresholds/tipping points in the foreseeable future (by 2025), with a very high probability that such thresholds will happen in regions of very intensively managed agriculture.

4.2.6 Can we quantify and map the global provision of biological control of crop pests, and how it might change?

Making assumptions about the proportion of insects that cause any damage, the proportion of native pest species in the US, and the relative amount of pest suppression that is due to insects rather than other causes, Losey and Vaughan (2006) illustrate how to use the ratio of cost of damage due insect pests to estimate the value of native pest control services. However, such a calculation is based on location-specific assumptions, and it is not clear that it could be scaled up to a global calculation. It would also not be fine-scale enough to quantify the effects of changes in biological control associated with changes in states of the world due to specific policy measures.

A model for biological control services has not been created, although Kremen et al. (2007) created a conceptual framework for the effects of land-use change on 'mobile-agent-based ecosystem services', including pest control. Initially developed for pollination (see chapter), the model includes interactions and feedbacks among policies affecting land use, market forces, and the biology of organisms involved.

Unlike for pollination (Section 4.1), syntheses of the literature to quantify the key relationships within this framework have not yet been carried out for biological control. These include:

- The relative value of biological control to crop productivity, that is, the vulnerability of crops to a shortage of natural enemies. This can be identified by the intensity of pesticide applications to those crops (Kremen & Chaplin-Kramer 2007).

- The relationship between the provision of biological control and distance to natural habitat patch (related to the dispersal and foraging movements of natural enemies).

- The relationship between the provision of biological control and area and quality of habitat patch.

- The functional significance of species diversity and the extent to which this varies across systems. This is needed to understand service provision by assemblages of natural enemies.

These syntheses need to build from field data and (ideally) experiments. We are not sure if the current body of knowledge is sufficient for such syntheses to be produced from published information, or if additional field work would be required. A complication in relation to pollination (in which natural habitat is always expected to have a positive effect), is that natural habitat may simultaneously result in the increase in the populations of natural enemies *and* of pests. Hence, the measure of enemy abundance on its own is not an adequate surrogate for biological control and instead information on the variation in crop yields should be used.

Adequacy of scenarios

We suspect that the same variables required for modelling pollination (Section 4.1) would be needed for modelling biological control.

What can be done in Phase 2? At what cost? By whom?

We suspect that there is not enough empirical data on which to base a global model to evaluate how biological control services are affected by changes in wild nature. This model could be obtained following the same lines as for pollination (and probably led by the same group) but we suspect this would not be possible within Phase 2.

We nonetheless recommend that further advice is obtained from experts on the feasibility of this particular task.

4.2.7 Insights for economic valuation

See pollination section (Section 4.1)

4.2.8 Some key resources

This section was based on a (quick) overview of the literature, and we suggest it is further developed by contacting experts in this field. We recommend the following as first points of contact:

- Claire Kremen at the Department of Environmental Science, Policy, and Management, University of California, Berkeley (USA) who has led development of a model on 'mobile-agent-based ecosystem services', including pest control (Kremen et al. 2007).
- Teja Tscharntke in the Department of Crop Science, University of Göttingen, (Germany), whose group also includes Carsten Thies.
- David Pimentel at the Department of Entomology, College of Agriculture and Life Sciences, Cornell University (USA).
- Felix Bianchi at the Functional and Spatial Ecology group in CSIRO Entomology (Australia).

4.2.9 Participants

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4.3 Genetic diversity of crops and livestock

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.3.1 Why is the genetic diversity of crops and livestock important for human wellbeing?

Food production and security depend on the wise use and conservation of agricultural and livestock biodiversity and genetic resources. Crops and their wild relatives (plant genetic resources for food and agriculture; FAO 1997) and livestock and their wild relatives (animal genetic resources for food and agriculture; FAO 2007a) have the genetic variability that provides the raw material for breeding new crop varieties, through classical breeding and biotechnological techniques. This diversity is fundamental for responding to environmental and demographic changes (Esquinas-Alcázar 2005). Note that this section only refers to species that are relatives of known crop species, and which are therefore of potential value to their future productivity or resilience. Species that have not yet been domesticated, but have the potential to become so, are covered in Section 4.14.

While thousands of plant and animal species have been domesticated for human use, currently, barely more than 150 plant species are cultivated and most of mankind lives off no more than 12 plant species (Esquinas-Alcázar 2005) with 30 crops supplying 90% of the global calorie intake (Wood et al. 2005). Domestic animals provide 30% of food globally but just 14 species represent 90% of the global production (Wood et al. 2005). Global population growth and corresponding rising demand for food have favoured the adoption of highly productive varieties. The introduction of modern farm machinery, marketing and transport methods that require uniform crop and livestock characteristics have required the introduction of standard, homogeneous plants and animals. This trend, which peaked during the so-called 'Green Revolution', has made it possible to boost food production, but has also led to the loss of innumerable heterogeneous (and often locally adapted) traditional farmers' varieties (Esquinas-Alcázar 2005).

The loss of local species and varieties usually results in irreversible loss of the genetic diversity they contain, known as genetic erosion. This has dangerously shrunk the genetic pool that is available for natural selection, and for selection by farmers and plant and livestock breeders, and has consequently increased the vulnerability of agricultural crops and livestock production to sudden changes in climate, and to the appearance of new pests and diseases (Esquinas-Alcázar 2005). For example, in the United States in 1970, the fungus *Helminthosporium maydis* destroyed more than half the standing maize crop in the southern part of the country. The crop had been grown from seeds that have a narrow genetic base and are susceptible to this disease. In this case and others, the problem was resolved by breeding resistant varieties using genetic resources that were obtained from other parts of the world (Esquinas-Alcázar 2005).

The conservation of animal and plant genetic resources for food and agriculture relies on the preservation of both the variety of domesticated species and their wild relatives. This can take place *ex-situ*, particularly for plants, through the development of seed banks and germplasm collections (FAO 1997; e.g. the recently inaugurated Svalbard Global Seed

Vault, already with 100 million seeds). Here, we are particularly interested in the *in-situ* conservation of genetic diversity, through the protection of the natural habitats of wild relatives or through the maintenance of traditional agricultural landscapes (Maxted 2003; Maxted et. al 2007). Note that with the exception of the wild boar (*Sus scrofa*) the ancestors and wild relatives of major livestock species are either extinct or highly endangered and that therefore domestic livestock are the only repositories of the now largely vanished diversity of the wild ancestors. This is a major difference from crop species, for many of which the wild ancestors are commonly found at the centres of origin and represent an important source of variation and adaptive traits for future breeding programmes (FAO 2007a).

The term 'crop biodiversity' is sometimes used to refer to the diversity of crops species within one region (e.g. wheat and barley rather than just wheat; e.g. Di Falco & Perrings 2005). We are not addressing the effects of crop diversity in that sense; we are concerned with intraspecific diversity of crop species, and particularly with the wild varieties.

Despite the widespread use of modern varieties, landraces (locally adapted, traditional varieties) are still grown in many parts of the world. This is particularly the case in marginal areas where such varieties (locally adapted, e.g., to specific soil and water regimes) thrive better. Landraces may also allow labour requirements to be spread (through differences in maturing time of the varieties), may be adequate for multiple usages (e.g., differences in stalks can make them better for use as fencing material or for feeding to cattle), and may reduce the risk of outbreaks of diseases or pests (Cassman et al. 2005).

4.3.2 What are the overall trends in the genetic diversity of crops and livestock?

The genetic diversity of crops and livestock has been declining markedly, but overall rates of loss are not easy to quantify (FAO 1997). For example, in China nearly 10,000 wheat varieties were in use in 1949, but only 1,000 were still in use by the 1970s. In the US, 86% of the apple varieties, 95% of the cabbage, 91% of the field maize, 94% of the pea, and 81% of the tomato varieties documented as having been in use between 1804 and 1904 apparently no longer exist (FAO 1997). Of the 7,616 livestock breeds listed by FAO's Global Databank for Animal Genetic Resources for Food and Agriculture, around 20% are classified as at risk and 62 breeds became extinct during the 2001-2007 period – amounting to the loss of almost one breed per month (FAO 2007a).

The three main crops – rice, maize and wheat – that provide over half the world population's requirement for protein and calories (Bioversity International 2008) are increasing reliant on a small number of modern varieties. Indeed 80% of the wheat area sown in developing countries is modern semi-dwarf varieties and over 75% of all rice in Asia is improved semi-dwarf varieties (Cassman et al. 2005)

4.3.3 How is the genetic diversity of crops and livestock affected by changes in wild nature?

The genetic diversity of wild relativess of crops (and to a lesser extent livestock) species is a direct result of the <u>species and population diversity</u> amongst these species. They are therefore affected by changes that lead to species/populations extinctions, such as <u>loss and degradation of natural habitats</u>. As a response, it has been proposed that networks of protected areas are established for the protection of wild crop relatives in the UK and Europe (Maxted 2003; Maxted et al. 2007).

Amongst domesticated relatives, loss of varieties and diversity within varieties is expected from the loss of area under traditional agricultural practices and associated local breeds, such as when these are either abandoned or converted to intensive agriculture.

Relationship between habitat area and genetic diversity of crops and livestock

Losses in genetic diversity species with area are likely to follow a non-linear relationship (linear in log-log space), similar in shape to the species-area relationship (but perhaps steeper, as it responds not only to the loss of species but also to the loss of intra-variety (intra-specific) genetic diversity.

4.3.4 What are the main threats to the genetic diversity of crops and livestock?

The main threat to the genetic diversity of crops and livestock is the marginalization of traditional production systems and the associated local breeds, driven mainly by the rapid spread of intensive agriculture and livestock production, which typically uses a narrow range of crop varieties and livestock breeds (FAO 1997, 2007). The intensification process has been driven by rising demand for agricultural and animal products, and has been facilitated by the ease with which genetic material, production technologies and inputs can now be moved around the world (Esquinas-Alcázar 2005; FAO 2007a).

Acute threats such as major disease epidemics and disasters of various kinds (droughts, floods, military conflicts, etc.) are also a concern – particularly in the case of small, geographically concentrated varieties and breed populations (FAO 2007a).

The loss of cultural diversity is also a factor in the loss of genetic diversity as cultural knowledge about production of traditional varieties and extraction of wild relatives is also lost (FAO 1997).

Wild varieties are threatened by habitat loss and degradation, and overharvesting. We presume that climate change may also affect not only wild relatives but also domestic breeds and varieties, if it reduces the degree to which they are adapted to their local environments.

4.3.5 Are abrupt changes likely in the genetic diversity of crops and livestock?

Given the likely non-linear relationship between area and genetic diversity, we predict that in some cases a small change in area (of natural habitat, or of traditional agricultural lands) may result in a disproportionate loss in genetic diversity of crops and/or livestock. This is probably more likely in areas that have already suffered extensive habitat loss and land conversion, where the remaining populations of particular varieties and breeds are quite small.

Climate change may also have non-linear effects on genetic diversity of crops and livestock.

Overall, we suspect that abrupt changes are likely in the genetic diversity of the wild relatives of crops and livestock in the foreseeable future (by 2025), particularly in regions subject to extensive habitat loss and land conversion.

4.3.6 Can we quantify and map the global diversity of crops and livestock, and how it might change?

We predict that it would be possible to create a global, spatially-explicit, model of crop and livestock diversity, its values, and how these may change, through the following steps:

1) Map the distribution of wild relatives and of local varieties and breeds. There seem to be a few databases on which this could be based, including for example the Domestic Animal Genetic Resources Information System (DAGRIS), and the databases under the Germplasm Resources Information Network (GRIN). However, we are not sure if these are geographically and taxonomically comprehensive and if the distributional information (which seems to be at the country level) is sufficient.

2) Obtain an estimate of the economic value of each breed/variety as insurance against pest and disease outbreaks. This would need to be estimated from statistics on past outbreaks to predict the risk of future outbreaks. The economic consequences of those outbreaks would be estimated from the predictions of damage and of crop/livestock prices. The economic value of each breed/variety would be estimated as the likely damage avoided by using its genetic diversity to prevent outbreaks. This would in principle follow similar principles to the calculation of an insurance premium given the predicted probability of relatively infrequent (but economically costly) events. We do not know if the data required for these calculations exist.

3) The comparison between states of the world (with different maps of landuse, including the replacement of areas of extensive with intensive agriculture) would allow for an estimate of which breeds/varieties would be more likely to be lost given their spatial distributions. Particularly relevant would be to understand which of these breeds/varieties are restricted to specific areas (and are not in seed banks, for example). The economic value of those losses would be calculated from the estimate of the insurance value of each breed/variety.

We suspect that this model would be best for calculating changes in crop, rather than livestock, genetic diversity, given that the latter are less likely to be directly related to changes in landuse. For breeds/varieties conserved ex-situ (e.g. in seed banks) the relevant comparison would be between one scenario in which the creation and maintenance of these banks is ensured versus one where it is not – the costs of creation/maintenance would then be compared with the benefits (estimated avoidance of costs of disease and pest outbreaks).

What can be done in Phase 2? At what cost? By whom?

We believe that conceptually it is possible to assess the economic consequences of different policy strategies (different states of the world) on the genetic diversity of crops (and perhaps livestock). However, we are not sure if the adequate data exist on which to base at least a first-cut model. If the data are available in a centralised way, then this calculation should be possible within Phase 2, otherwise it is unlikely that such task can be undertaken in such a short time frame.

We recommend that further advice is obtained from experts on the feasibility of this particular task.

Adequacy of scenarios

We predict that the key data that scenarios would have to produce is information on landuse change, including on the intensification of agriculture. Scenarios could also specifically assess the costs and benefits of maintaining breeds and varieties *ex-situ*.

4.3.7 Insights for economic valuation

The general model proposed would directly provide results in economic terms.

4.3.8 Some key resources

- Food and Agriculture Organization of the United Nations (FAO), Commission on Genetic Resources for Food and Agriculture. Has published both *The State of the World's Plant Genetic Resources for Food and Agriculture* (1997) and *The State of the World's Animal genetic Resources for Food and Agriculture* (2007).
- International Treaty on Plant Genetic Resources for Food and Agriculture (ITPGRFA)
- Global crop diversity trust
- Convention on Biological Diversity (CBD) – Agricultural Biodiversity
- IUCN SSC Crop Wild Relative Specialist Group, a network of crop wild relative experts around the world dedicated to working jointly to promote the conservation and use of crop wild relatives.
- International Food Policy Research Institute (IFPRI)
- International Livestock Research Institute (ILRI), that has build the database Domestic Animal Genetic Resources Information System (DAGRIS), containing research-based information on the distribution, characteristics and status of 871 breeds of cattle, sheep, goats, pigs and chickens indigenous to Africa and Asia.
- Consultative Group on International Agricultural Research (CGIAR) is a strategic alliance of countries, international and regional organizations, and private foundations supporting 15 independent international agricultural research centers. Bioversity is one of these CGIAR-supported Centres. Bioversity International (formerly IPGRI) is one of those centes.

- Germplasm Resources Information Network (GRIN) within the U.S. Department of Agriculture's Agricultural Research Service.

Possible experts to contact include:

- Dr Nigel Maxted, Senior Lecturer of Plant Gentic Conservation at the University of Birmingham and Chair of the IUCN Species Survival Commission Crop Wild Relative Specialist Group. His research focuses on the conservation of genetic diversity in plants, including the development of approaches to the *in situ* conservation and management of crop wild relatives.
- Melinda Smale, Senior Research Fellow at IFPRI. Her research emphasizes the development of methods to assess the value of crop biodiversity and the identification of policies to enhance the utilization and management of crop genetic resource, particularly in developing economies.
- Jan Engels, Genetic Resources Science and Technology Group, at Bioversity International.
- José Esquinas-Alcázar, FAO Commission on Genetic Resources for Food and Agriculture.

4.3.9 Participants

Authors

Hannah Peck, Andrew Balmford and Ana Rodrigues (University of Cambridge, UK)

Acknowledgements

John Hopkins (Natural England, UK) provided useful comments and references.

4.4 Soil quality for crop production

This section is based on a quick literature review. It greatly benefited from expert contributions (see below) but it has not been reviewed by experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.4.1 Contributions of wild nature to soil quality and crop production

There seem to be four main ways in which wild nature improves or maintains the quality of farmland soils and thereby contributes to the benefits that we obtain from agricultural crops (e.g. food, fibres, fuels) by improving or retaining farmland soil quality. We called these: internal effects (soil biota); conversion effects (when non-agricultural land is converted to agriculture); neighbourhood effects (when neighbouring, non-agricultural, systems contribute to soil quality in croplands); and wild fertilisers (e.g. guano).

Note that the importance of soil for water regulation is covered separately (Section 4.9).

Internal effects

The soil fauna and flora, and the communities occupying the agricultural land, affect soil properties and therefore its quality for agriculture. These 'internal effects' seem to be the main way in which wild nature affects soil quality, by affecting: nutrient fixation (e.g. nitrogen fixation by bacteria); nutrient cycling (e.g. organic matter decomposition,by fungi, bacteria, dung beetles...); soil structure (e.g. by plant roots, termites...); water regulation, particularly water holding capacity and drainage (e.g., by plant roots, termites, micro-organisms and earthworms...); uptake of water and nutrients (mycorrhizal fungi); erosion regulation (e.g. by vegetation cover and leaf litter). The soil biota may also suppress pests and diseases (e.g. mycorrhizal fungi), but these are considered under biological control (Section 4.2).

More than species diversity, it seems that functional diversity (and its influence on trophic interactions) is key to the decomposition, nutrient cycling, and stability of soil processes. Soil diversity is important for resilience to stress and disturbance (Griffiths et al. 2000; Brussaard et al. 2007).

As agricultural intensification occurs, the regulation of functions through soil biodiversity is progressively replaced by regulation through chemical and mechanical inputs (Giller et al. 1997; Swift et al. 2004). The type of agricultural management is the primary determinant of the composition, diversity and functioning of the soil food web. Reaping the rewards of soil biodiversity in farming systems therefore requires identifying those practices that best harness the beneficial roles of the soil food web. One idea commonly proposed is that management regimes that encourage a soil community that bears the closest resemblance to natural ecosystems are more likely to require fewer inputs because of greater reliance on ecosystem self-regulation (e.g. Bardgett & McAlister 1999; Mäder et al. 2002). In tropical agricultural systems undergoing intensification, large numbers of farmers have limited access to inputs, and therefore the maintenance and enhancement of soil biodiversity may be particularly relevant (Giller et al. 1997). One key feature of natural ecosystems is a soil community that is dominated by fungal-based pathways of decomposition, whereas intensive agriculture tends to promote more bacterial-based pathways of decomposition. Therefore, farming systems that encourage fungi and their consumers (such as non-tillage systems, lowering of fertiliser inputs, and organic farming) might perform best in terms of nutrient cycling, carbon storage (e.g. Chivenge et al. 2007), and nutrient retention under stress (e.g. Gordon et al. 2008).

We predict that the importance of 'internal effects' is increasing as agricultural land expands and appropriate soil management becomes more important to ensure long-term crop production (particularly when the inputs of chemical fertilizers and pesticides are restricted for environmental reasons), and as organic agriculture expands (Mäder et al. 2002).

We do not know the extent to which the economic consequences of different agricultural management regimes on crop yields (specifically through their effect on soil quality) have been quantified. We get the impression that while a fair amount of work seems to have been done in comparing organic and conventional farming systems, this has mainly focused on temperate regions. Even if information is available on how crop yields are affected by changes in the soil biota resulting from different management regimes, it would still be difficult to assess these effects globally. Indeed, the scenarios being produced to compare a business-as-usual world with a more biodiversity-friendly world would need detailed spatial information not only on the specific crops but also on the management regime for each crop. We suspect this would not be possible for Phase 2.

Other relevant references:

- Cassman et al. (2005) – section on cultivated systems of the *Millennium Ecosystem Assessment* (volume *Ecosystems and Human Well-being: Current States and Trends*).
- Bloem et al. (2004) – Measuring soil biodiversity: experiences, impediments and research needs.
- Bloem et al. (1997) – Food webs and nutrient cycling in agro-ecosystems
- Bloem et al. (2006) – Microbiological methods for assessing soil quality
- Schjønning et al. (2004) – edited volume on *Managing Soil Quality: Challenges in Modern Agriculture*, particularly section by Brussaard on *Biological soil quality from biomass to biodiversity – Importance and resilience to management stress and disturbance*.
- FAO (1999) – Sustaining Agricultural Biodiversity and Agro-ecosystem Functions.
- Brussaard et al. (2007) – Soil biodiversity for agricultural sustainability
- Wood et al. (2000) – Pilot analysis of global ecosystems: Agroecosystems

Conversion effects

These are inputs from wild nature, when non-agricultural land is converted into agricultural land. They may take place as one-off events as agriculture expands into areas occupied by natural ecosystems (e.g. the Amazonian frontier), or cyclically through shifting agriculture.

Here, crops benefit from the improvements in soil quality that were accumulated over time in the previous form of land use (e.g. long-term soil improvement in forest systems).

Agricultural expansion into new areas often occupies terrains that are nor particularly suitable for agriculture, and soil fertility may decline very quickly as crops effectively mine the soil nutrients (Alfsen et al. 2001; Carr et al. 2006). Soil fertility may however be retained under appropriate agricultural practices that ensure adequate replacement of the nutrients used by the crops (Barber & Díaz 1994).

Shifting cultivation, also called ‘swidden’ agriculture or ‘slash-and-burn’ agriculture, is one of the oldest forms of farming and consists of cropping on cleared plots of land, alternated with lengthy fallow periods. These systems are the dominant form of agriculture in tropical humid and subhumid upland regions and are typically associated with tropical rain forests. Shifting cultivation is practiced on about 22% of all agricultural land in the tropics and is the primary source of food and income for some 40 million people (Giller and Palm 2004), including many of the world’s poorest. While the contribution to global food security is negligible, given the low yields and general lack of infrastructure in areas where shifting cultivation predominates, this method of cultivation has a potentially large impact on regional and global ecosystem services through its effects on biodiversity, greenhouse gas emissions, and soil nutrients. Although these systems are generally associated with soils of low fertility, they are highly sustainable and resource-conserving in areas with low population density (Kleinman et al. 1995). However, high population density increases the pressure on available land and resources, reducing the time available for a regenerative fallow between cropping cycles (Cassman et al. 2005), and effectively converting the region into an agricultural frontier.

We predict that the importance of conversion effects is at present increasing as the agricultural frontier expands in regions such as the Amazon. Over the longer term, however, we expect that conversion effects will become less significant as the availability of unused land that is suitable for farming decreases, and as as population growth renders shifting agriculture non-viable.

In comparing different states of the world (created by different scenarios) it should be possible to account for the economic value of soil fertility for crops in agricultural frontiers, quantified as one-off subsidies in the conversion from natural ecosystems to agriculture.

As for shifting agriculture, by knowing the time needed for soil recovery and the number of people that can be sustained per unit of area of cultivated plots it should be possible to map potential for sustainable shifting cultivation worldwide and the maximum population densities allowed for long-term sustainability. This could then be crossed with data on demand (regions where populations are more likely to depend on this type of agriculture) as a basis for calculating the potential economic value of a sustainable input from natural ecosystems to farmland soil through shifting agriculture. Given current human population densities, it is predicted that shifting cultivation can only provide a minor contribution to food production (from a global perspective), although as mentioned poor people in some regions will depend on this type of agricultural practice. The time needed for fields to recover is likely to be variable and will depend on soil type and climate.

Other relevant references:

- Moreira et al. (2006) – book *Soil Biodiversity in Amazonian and Other Brazilian Ecosystems*.
- Noordwijk (1995) – book *Below-ground Interactions in Tropical Agroecosystems*.

Neighbourhood effects

This occurs when neighbouring, non-agricultural, systems contribute to improve or maintain soil quality in croplands. We envisage that this could include, for example, coastal sand dunes sheltering inland agricultural fields from sea salt spray, or uphill forests protecting downslope croplands from erosion (by reducing water runoff) or from landslides. Under some circumstances there may be positive neighbourhood effects of ecosystem degradation – for instance if upstream deforestation and resulting erosion and downstream siltation boosts agricultural yield (e.g. the annual replenishment of soil fertility in the Nile Valley by soil from Ethiopia, prior to the construction of the Aswan High Dam). However, such gains may not be long-lasting, if upstream erosion is sustained, or if siltation has other negative effects downstream.

The importance of neighbourhood effects may be increasing as agriculture expands from the most favourable lowlands into slopes, or declining as the size of fields and the overall size of farmed areas expands and therefore reduces the contact with other ecosystems.

Assuming that these effects are essentially related to the physical protection of farmland soil by neighbouring ecosystems, it seems conceptually possible to model them, such that, for example, one would be able to quantify the changes in farmland soil quality in a watershed having either more or less coverage of upslope forest. However, we found no evidence for the existence of such models.

Wild fertilisers

These are products obtained from wild nature, which are used as crop fertilisers, such as guano, seaweed, peat and fish. These subsidies to soil quality have been very important historically (e.g. guano was the basis of the Peruvian economy in the mid 19th Century) but their importance has declined substantially with the development of artificial fertilisers. Wild fertilisers may still be important in areas where farmers cannot afford artificial fertilisers, though.

We suspect there is not a centralised dataset on current use of wild fertilisers that could be used as the basis of for a spatially explicit economic evaluation.

4.4.2 Can we quantify and map the contribution of wild nature for the provision of soil quality for crops, and how it might change?

What can be done in Phase 2? By whom?

Internal effects We predict that it is unlikely that even a first-cut quantification could be possible of the value of wild nature for agriculture through internal effects. The type of agricultural management is the primary factor in this evaluation, and so modelling the economic consequences of different policy options on soil fertility (and crop yields) would require detailed information not only on how specific crops vary across space but also on management practices in each plot. We suspect this is not a feasible task for Phase 2.

Conversion effects We predict that it should be possible to obtain a first-cut model for quantifying the value of wild nature for agriculture through conversion effects, for both frontier and shifting cultivation. We suggest further expert consultation (see below) to evaluate the feasibility of this particular task and understand who the appropriate experts would be.

Neighbourhood effects While conceptually possible to quantify, we found no evidence for researchers thinking along these lines and therefore suspect that this task would not be feasible in the short-term.

Wildlife fertilisers We suspect there is no centralised dataset on current use of wild fertilisers that could be used as the basis of for a spatially explicit economic evaluation in Phase 2.

4.4.3 Some key resources

- FAO Land and Water Development Division, particularly its section on soil health for food security. They maintain the *Soil Biodiversity Portal*: *The role of soil organisms in agriculture for better land management decision.*
- ISRIC – World Soil Information, including David Dent (Director) and Zhanguo Bai (global assessment of land degradation and improvement).
- Alterra, Wageningen University, The Netherlands, including: Peter de Ruiter, Jaap Bloem and K.B. Zwart
- Kenneth G. Cassman and John W. Doran at the University of Nebraska-Lincoln, USA.
- Stanley Wood at the International Food Policy Research Institute, Washington DC, USA.
- Rattan Lal at the School of Natural Resources, College of Food, Agricultural, and Environmental Science, Ohio State University, USA.

- Peter Kleinman at the Department of Crop and Soil Sciences, Pennsylvania State University, USA

- Paul Mäder at the Research Institute of Organic Agriculture, Switzerland

- Richard D. Bardgett Soil and Ecosystem Ecology, Lancaster University, United Kingdom

- Sara Scherr at Ecoagriculture Partners, USA

4.4.4 Participants

Authors

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Contributors

The following experts provided very valuable insights on which this section was based. We apologise to them that time constraints prevented us from circulating the text for revision.

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Jaap Bloem (Alterra, Wageningen University, The Netherlands)

Richard D. Bardgett (Lancaster University, UK)

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Stanley Wood (IFPRI, Washington DC, USA) provided useful references.

4.5 Livestock

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.5.1 Why is the production of livestock important for human wellbeing?

Livestock and livestock products (e.g. milk, eggs) are estimated to make up over half of the total value of gross agricultural output in industrial countries, and about a third of the total in developing countries (Wood et al. 2005). Globally, they account for 40% of agricultural gross domestic product, provide one-third of humanity's protein intake, employ 1.3 billion people and create livelihoods for one billion of the world's poor (Steinfeld et al. 2006). Livestock are also a significant source of income and consumption in developing countries, often providing a supplementary source of income and income stability, and are frequently one of the few assets available to the poorest.

Livestock production also affects human wellbeing negatively by: causing land degradation and desertification, habitat loss, climate change (18% of greenhouse gas emissions), and water pollution (see Steinfeld et al. 2006 for a very comprehensive review).

4.5.2 What are the overall trends in the production of livestock?

The global importance of livestock and their products is increasing with population growth, changes in food preferences (associated with rising incomes and urbanisation), and expanding global trade. Global production of meat and milk are projected to double between 2000 and 2050 (Steinfeld et al. 2006).

4.5.3 How is the production of livestock affected by changes in wild nature?

Livestock production is the single greatest anthropogenic land use, with direct grazing and browsing accounting for 26% of the ice-free terrestrial land surface, and production of of fodder and feedgrains accounting for 33% of total arable land. In all, livestock production occupies around 70% of all agricultural land and 30% of the land surface of the planet (Steinfeld et al. 2006).

An increasing fraction of livestock production is intensive, with crops provided as feed to animals kept at high densities. Globally, it is estimated that 35% of all cereals produced are fed to animals. In this case, the contribution of wild nature for livestock production is indirect, via the contribution of nature to crop production, including through pollination (Section 4.1), biological control (Section 4.2), soil quality (Section 4.4) and water regulation (Section 4.9) for crops, and the regulation of natural hazards (Section 4.13).

An important fraction of livestock production takes place in natural grasslands, and that is the focus of this section. Here livestock production is being directly subsidised by the primary productivity of natural ecosystems, and the species and ecosystem processes that sustain it.

Wild nature also subsidises livestock production when pasturelands expand into other ecosystems. Expansion of livestock production is a main driver of deforestation, particularly in Latin America – 70% of the previously forest landed area in the Amazon is occupied by pastures, and feedcrops occupy much of the rest (Steinfeld et al. 2006). Here, wild nature is contributing by providing soil fertility, as discussed under "Conversion effects" in Section 4.4.

The contribution of genetic diversity to livestock production is discussed in Section 4.3.

Relationship between habitat area and livestock production

We expect this relationship to be linear.

4.5.4 What are the main threats to livestock production in natural grasslands?

Overexploitation of natural grasslands, by keeping livestock at a higher density than what is sustainable, leads to degradation of the grasslands that is not easily reversed. For example, about 20% of the world's pastures, rising to 73% in dry areas, have been degraded to some extent, mostly through overgrazing, compactation, and soil erosion created by livestock action. Drylands tend to be particularly affected as they have lower natural productivity and livestock is often the only source of people's livelihoods in such areas (Steinfeld et al. 2006). Invasive plants may also reduce pasture productivity; for example, *Lantana camara* considred one of the world's ten worst invasive species, is toxic to livestock and furthermore can greatly alter natural fire regimes (Steinfeld et al. 2006).

Climate change is likely to affect livestock production in due course, both by affecting natural rangelands (particularly in regions where predictions are of increased aridity) and by affecting feed crop production.

4.5.5 Are abrupt changes likely in the provision of livestock in natural grasslands?

We found no evidence for future abrupt and disproportionate changes in the capacity of natural systems (grasslands) to support livestock production.

4.5.6 Can we quantify and map the contribution of wild nature for livestock production, and how it might change?

A first-cut map of the contribution of wild nature for livestock production already exists, as a map of the annual production of livestock derived – at least in part – from grazing on unimproved natural grasslands (Naidoo et al. in press). This was obtained by combining global data on livestock distributions (Wint & Robinson 2006), producer prices, and current and potential vegetation (Ramankutty & Foley 1999; Bartholome & Belward 2005)

To map livestock production on natural pastures, Naidoo et al. (in press) used recently-developed 5' resolution global maps of livestock distributions (Wint & Robinson 2006), which used regression-based methods to estimate the expected density of cattle, sheep, goats, pigs, poultry, and buffalo across the earth's surface. For each livestock type, they used these density estimates and data on the mass of edible meat per animal (estimated by country from FAOSTAT data) to estimate the tons of meat produced in each cell. A global producer price was used to weight different livestock types; using these weights, an aggregate index of livestock production was obtained by summing the weighted livestock meat weights. Naidoo et al. (in press) then constructed a global map of natural pastures by combining a 5' resolution potential vegetation data set of savanna, grassland/steppe, and shrubland biome types (Ramankutty & Foley 1999) and then masking out all known human-altered landscapes using the Global Land Cover 2000 remote sensing data (Bartholome & Belward 2005). They intersected the resulting map of potential natural pastures with the livestock production index to produce a global map of livestock production on natural pastures.

Limitations of these data include the difficulties of mapping pastures from remotely sensed imagery and the lack of spatially explicit weightings that reflect differences in the economic values of livestock species in different regions of the world (Naidoo et al. in press). The new pastureland maps by Ramankutty et al. (2008) provide more updated information than Ramankutty & Foley (1999).

The map by Naidoo et al. (in press) provides an estimate of current production in grasslands. However, this may overestimate the natural sustainable production rates. On the one hand, existing livestock may in some areas receive other food supplements, and therefore not depend completely on the natural grassland production. On the other hand, livestock may be kept at densities that are unsustainable over the long term (Steinfeld et al. 2006). Hence the Naidoo et al. (in press) map is not ideal, but we believe it is a very useful first cut to quantifying the value of wild nature for livestock production in natural areas. In the longer-term, it would be helpful to calibrate this map to account for the maximum natural productivity of land.

What can be done in Phase 2? At what cost? By whom?

A first-cut global quantification of livestock production on natural pastures has already been produced by Robin Naidoo and colleagues. This could be used directly or updated with the new pasture map by Ramankutty et al. (2008).

Adequacy of scenarios

Appropriate scenarios would need to predict landuse and demand for livestock products.

4.5.7 Insights for economic valuation

The map produced by Naidoo et al. (in press) has as units an aggregate index of livestock production, but it could have equally have been produced in terms of estimated monetary value of the livestock produced in each cell.

4.5.8 Some key resources

- Wint & Robinson (2006) – *Gridded Livestock of the World*, by the FAO.
- Ramankutty et al. (2008) present newly derived global maps of agricultural lands, including pasturelands.
- Steinfeld et al. (2006) present an assessment of the full impact of the livestock sector on development and the environment, produced by the Livestock, Environment and Development (LEAD) Initiative of the FAO.

4.5.9 Participants

Authors

Ana Rodrigues, Andrew Balmford (University of Cambridge, UK)

4.6 Marine fisheries

This section is a fully developed review, including contributions by experts in the field, some of whom subsequently reviewed the text.

4.6.1 Why are marine fisheries important for human wellbeing?

This section focuses on marine capture fisheries, not covering marine aquaculture.

Marine capture fisheries are a globally important source of food: fish is highly nutritious, rich in micronutrients, minerals, essential fatty acids and proteins, and represents a valuable supplement to diets otherwise lacking essential vitamins and minerals. Globally fish provides more than 2.8 billion people with at least 20 percent of their average per capita animal protein intake (FAO 2006b). Marine capture fisheries contribute 56% of the total amount of fish available for human consumption (FAO 2006b). Marine fisheries also affect the provisioning of food indirectly, via the use of fishmeal and fish oil in aquaculture and in poultry and pig production (Arthurton et al. 2007). Demand for fish is increasing with population growth, rising wealth and changing food preferences as a result of the marketing of fish in developed countries as part of a healthy diet (Arthurton et al. 2007).

Accordingly, marine capture fisheries are an important source of economic benefits, with an estimated first-sale value of US$84.9 billion (FAO 2006b), and important for income generation, with an estimated 38 million people employed directly by fishing, and many more in the processing stages (Arthurton et al. 2007).

Marine fisheries are increasingly valuable for recreation, particularly in developed countries. In the US alone, in 2006 nearly 13 million anglers made more than 89 million marine recreational fishing trips on the Atlantic, Gulf and Pacific coasts, capturing almost 476 million fish, of which 55 percent were released alive (Van Voorhees & Pritchard 2007). In the European Union (EU 15), an estimated 8 million recreational sea anglers (Stevenson 2007) spend an estimated €25 billion a year, compared to a €20 billion value for commercial landings in 1998 (Pawson et al. 2004).

4.6.2 What are the overall trends in the provision of marine fisheries?

The global trends in marine fisheries have been reviewed in the Millennium Ecosystem Assessment (Pauly et al. 2005). Catches increased substantially during the twentieth century as fishing fleets worldwide expanded. These trends continued until the 1980s, when global marine landings reached slightly over 80 million tons per year; then they either stagnated or began to slowly decline. However, regional landings peaked at different times throughout the world, which in part masked the decline of many fisheries. This global decline is despite an increase in fleet size, effort, and technology, in conjunction with an expansion into previously unfished areas, depths and stocks (Pauly et al. 2005; Morato et al. 2006).

It is estimated that in 2005 around one-quarter of the stock groups monitored by FAO were underexploited or moderately exploited (3% and 20%, respectively) and could perhaps produce more. About half of the stocks (52%) were fully exploited and therefore producing catches that were at or close to their maximum sustainable limits, with no room for further expansion. The other one-quarter were either overexploited, depleted or recovering from depletion (17%, 7% and 1%, respectively) and thus were yielding less than their maximum potential owing to excess fishing pressure exerted in the past, with no possibilities in the short or medium term of further expansion and with an increased risk of further declines and need for rebuilding (FAO 2006b). The real extent of collapses in marine systems is likely to be more dramatic than realised, as massive declines that took place before systematic statistics began are likely to go unnoticed as perceptions are adjusted to what a natural marine system is (the 'shifting baseline syndrome'; Carlton et al. 1999; Jackson et al. 2001b). A meta-analysis of collapsed fisheries has indicated minimal recovery even after fishing moratoria are imposed (Hutchings 2000, Hutchings and Reynolds 2004). Loss of production has mean loss of earnings which has been disportionately absorbed by developing countries whose resources were the last to be fished down by multinational fleets (Srinivasan et al. 2008).

4.6.3 How is the provisioning of marine fisheries affected by changes in wild nature?

Biomass of the exploited species is the key parameter determining catches in marine fisheries, at least in the short-term. Global declines in exploited biomass are inferred from declines in catches despite increased effort. In the North Atlantic, for example, current overall biomass of high-trophic level fishes is estimated to be one third of what it was in 1950 (Christensen et al. 2003). Fishing pressure has reduced the biomass of some species to less than 10% of the pre-exploitation level within a few decades, particularly species with vulnerable life history traits such as large predatory fishes (Myers and Worm, 2003), including sharks and relatives (Myers et al. 2007), and deep sea species (Devine et al. 2006). While commercially exploited species have been particularly affected (e.g. Cod, Cook et al. 1997; bluefin tuna, Fromentin & Powers 2005), non-target species have also suffered depletions as by-catch (e.g. leatherback turtles, Spotila et al. 2000).

Fishing efforts tend to be concentrated on high-trophic species first. As these become depleted, the biomass of lower-trophic species may increase due to predator release (Scheffer et al. 2005). Fishing efforts then move down the food web, and landings from global fisheries have accordingly shifted in the last few decades from large piscivorous fishes toward smaller invertebrates and planktivorous fishes, especially in the Northern Hemisphere (Pauly et al. 1998). This may at first lead to increased catches, but overfishing of lower-trophic species will also result in their declines, and so fishing down the food web is unsustainable in the long-term (Pauly et al. 2005).

Provisioning of fishmeal for aquaculture and for meat production can be done using lower trophic levels species but recreational fisheries benefit specifically from large, long-lived, high-trophic fish species.

There is evidence that species diversity is important for marine fisheries, both in the short-term, by increasing productivity, and in the long-term, by increasing resilience. A recent review of fisheries trends in large marine ecosystems (Worm et al. 2006) found that the proportion of collapsed fisheries decayed exponentially with increasing species richness. Furthermore, the average catches of non-collapsed fisheries were higher in species-rich systems. Diversity also seemed to increase robustness to overexploitation, with rates of recovery positively correlated with fish diversity. This increased stability and productivity was attributed to a portfolio effect (Tilman et al. 2006), whereby a more diverse array of species provides a larger number of ecological functions and economic opportunities, leading to a more stable trajectory and better performance over time (Worm et al. 2006).

The diversity of genetically distinct populations, adapted to particular regions, is also likely to be important, especially in terms of the sustainability and resilience of wild stocks to longer-term natural and anthropogenic change (Hilborn et al. 2003a; Worm et al. 2006; Hiddink et al. in press). Maintaining the genetic diversity of marine fisheries keeps also options open for future farming of marine species.

Given that marine fisheries are sustained by lower-trophic species, they depend directly on the biomass of other, non-targeted, species, including phytoplankton and zooplankton. The diversity of non-target species, such as prey species, may also be important: experiments have shown the importance of diverse food sources for secondary production and for the stability of populations across trophic levels in marine ecosystems (Worm et al. 2006).

The removal (or depletion below a certain level) of populations of particular species or functional groups has been shown to have dramatic effects on some marine ecosystems and the associated fisheries. Predators in particular ('top-down control') seem to be very influential in shaping and maintaining various habitat states or population levels (Myers et al. 2007). Many examples of regime shifts associated with biodiversity losses have been documented (reviews in: Jackson et al. 2001b; Bellwood et al. 2004; Folke et al. 2004; Agardy et al. 2005) and include: kelp forest losses due to urchin population explosions after declines in large predatory fishes and sea otters; die-offs of turtlegrass beds following turtle population collapse; sudden catastrophic coral mortality due to overgrowth in algae after grazers (herbivorous fishes, urchins) have been lost (Mumby et al. 2007); trophic cascades following the collapse of cod and other large-bodied predators in the northwest Atlantic, transforming a highly productive system dominated by benthic fish into systems dominated by pelagic fish and macroinvertebrates with poor fisheries productivity (Frank et al. 2005); and two successive regime shifts in the Black Sea triggered by overfishing and compounded by euthrophication, firstly after depletion of top predators and secondly after depletion of planktivorous fish, resulting in a jellyfish-dominated, fisheries-poor system (Daskalov et al. 2007). Recovery following change may be slow and may follow a different trajectory from the one observed during the decline (hysteresis; Hughes et al. 2005).

Fisheries records provide compelling evidence of dramatic declines in the stocks of many exploited species. Substantially less is known about trends in non-targeted species. Marine species have long been assumed to be resilient to extinction, and therefore until recently been neglected in extinction risk assessments (Baillie et al. 2004). This is now changing, with ongoing assessments demonstrating that as human activities take over the world's oceans (Halpern et al. 2008) marine systems are suffering serious declines in species diversity, both locally (particularly in coastal habitats; Jackson et al. 2001b; Worm et al. 2006) and globally (e.g. predatory fish communities; Myers and Worm 2003). The largest ongoing assessment of species conservation status is the Global Marine Assessment (due to assess 20,000 marine species by 2010), which is assessing the threat status of every marine vertebrate species, and of selected habitat-forming invertebrates and plants.

In addition, experimental evidence suggests that a loss of biodiversity increases vulnerability to the establishment of invasive species (Stachowicz et al 1999, 2007; Worm et al. 2006).

Marine fisheries are vulnerable to the decline in the extent or quality of particular marine habitats that play important roles in the provisioning of key resources (e.g. food, shelter) for targeted species. These include, amongst others:

- Coral reefs, which directly support fisheries that constitute 9–12% of the world's total fisheries (up to 25% in some parts of the Indo-Pacific), providing livelihoods for millions of people in tropical coastal regions. A large number of offshore fisheries also rely on the supporting services of reefs as breeding, nursery or feeding grounds (Moberg and Folke 1999; Agardi et al. 2005). Cold water reefs are also important to fisheries (Fosså et al. 2002; Freiwald et al. 2004), although much less is known about these. Coral reefs are one of the world's most threatened habitats, particularly in the Caribbean (Gardner et al. 2003). Coral reefs are suffering a worldwide decline in quality and extent (Pandolfi et al. 2003; Bellwood et al. 2004, Newton et al. 2007), with some regions particularly degraded (e.g. Caribbean, Gardner et al. 2003; SE Asia, Middle East and Indian Ocean; Wilkinson 2006). They are also one of the habitats most at risk from climate change (Hoegh-Guldberg et al. 2008).

- Seamounts are very poorly known systems, but thought to be important for the provision of food, as sites for spawning aggregations, and nurseries/refuges for juvenile open ocean fish (Moore & Jennings 2000; Stocks 2004). Out of an estimated 100,000 seamounts (>1 km high), only about 350 have been sampled biologically (many more have no doubt been fished out), and less than 100 of these have been studied in any detail; the ones that were revealed extraordinary levels of species endemism (EuroCoML 2008). Seamounts are among 'newly' targeted ecosystems that have been intensively fished since the second half of the 20th century, particularly through highly destructive bottom trawling (Watson & Morato 2004). However, their long-lived, slow-growing, species are particularly vulnerable to over-exploitation, and so reports of drastic collapses are accumulating (Devine et al. 2006). Whole populations have been known to be depleted in just a few years (Devine et al. 2006) while few deep-sea fisheries have recovered from bottom trawling even two or three decades after fishing ceased (Morato et al. 2004; Watson & Morato 2004).

- Seagrass meadows are highly productive systems, thought to have a fundamental role in maintaining populations of commercially exploited fish and invertebrate species, in both tropical and temperate regions, by providing a permanent habitat, a temporary nursery area for the development of the juvenile stages, a feeding area for various life-history stages, and/or a refuge from predation (McArthur et al. 2003; Agardy et al. 2005). They are also thought to maintain fisheries indirectly by providing organic matter that is incorporated into coastal nutrient cycles (Jackson et al. 2001a; Gullstrom et al. 2002). Major losses of seagrass habitat have been reported from the Mediterranean, Florida Bay, and Australia (Duarte 2002). Present losses are expected to accelerate, especially in Southeast Asia and the Caribbean (Burke et al. 2001; Duarte 2002), as eutrophication increases, algal grazers are overfished, and coastal development increases.

- Kelp forests are highly productive temperate systems, with a complex biological structure organized around large brown algae, supporting a high diversity of species and species interactions. Kelp support fisheries of a variety of invertebrate and finfish species, particularly by providing nursery and shelter habitats, and the kelp itself is harvested for food and additives. Overfishing of predator species has led to regime shifts in many regions towards systems that are totally dominated by urchins and of substantially less value for fisheries (Agardy et al. 2005).

Marine fisheries are also vulnerable to the declines in the extent or quality of coastal habitats, including:

- Mangroves are found in intertidal zones and estuarine margins in tropical and subtropical regions, where they can be important to fisheries as refuge areas from predators or physical disturbance, or via food input (Agardy et al. 2005). Mangroves are particularly important for supporting artisanal fisheries in some regions; in Bangladesh, for example, the artisanal fishery (contributing 85–95% of the total coastal and marine catch) is highly influenced by mangroves (Islam & Haque 2004). Furthermore, mangroves may be ecologically linked to seagrass beds and coral reefs, with synergistic effects for fish productivity (Mumby et al. 2004). Much of the coastal population of the tropics and sub-tropics resides near mangroves, and it has been estimated that 35% of mangrove forests have been loss in the recent past principally due to conversion to agriculture, aquaculture, salt pans, and urban and industrial development (Valiela et al. 2001; Agardy et al. 2005).
- Estuaries and coastal wetlands play a key role in marine fisheries provision, particularly as nursery areas for fisheries (Beck et al. 2001), but also by providing habitat for commercially important molluscs and crustaceans, and in regulating water quality (Agardy et al. 2005). The loss of these habitats has been linked to the collapse of fisheries in North America, North Africa and elsewhere, although further quantitative studies are required (Agardy et al. 2005). Estuaries and coastal wetlands have been degraded, altered, or eliminated in many areas, particularly through reduction of the estuaries watershed through agricultural conversion, eutrophication, pollution by pesticides and herbicides, overfishing, and biological invasions (Agardy et al. 2005).

Marine fisheries are furthermore affected by changes in inland ecosystems that affect the quality, volume and timing of water inputs, as well as erosion regimes:

- Forests, wetlands and riparian habitats play an important part removing excess nutrients from freshwater runoffs (see Section 4.9). This role is becoming increasingly important as agricultural intensification is increasing nutrient loads, leading in some regions to eutrophication responsible for extensive ‘dead-zones’ that can impact seriously on the production of fisheries (Rabalais et al. 2002; Agardy et al. 2005).
- The volume and timing of freshwater inputs to marine systems affect the salinity and nutrient loads in estuarine and other coastal environments, in turn affecting the fisheries that depend on them. Inland changes in the regulation of the timing, volume and nutrient load of freshwater input (e.g., through the establishment of dams; Syvitski et al. 2005) are therefore likely to affect the breeding and recruitment of fisheries species dependent these environments. A dramatic example of fisheries declines associated with changes in freshwater regime can be found at the mouth of the Colorado River (Kowalewski et al. 2000; Rowell et al. 2005).
- Coral reefs, seagrass meadows, mangroves and estuaries can be degraded or even destroyed by increasing sedimentation caused by landuse change, and so inland erosion regulation is important for the conservation of these habitats and of the species that rely on them (Agardy et al. 2005). For example, a decline in the tropical fisheries of the Caribbean and Pacific has been linked to the degradation of coral reefs, seagrass beds, and mangroves from sedimentation (Rogers 1990).

Relationship between habitat area and fisheries provisioning

This question can be addressed in at least three ways:

i) Are larger areas of suitable habitat (e.g. larger coral reefs, larger mangroves) more productive in terms of fisheries *per unit of area* than smaller ones?

ii) How does fisheries productivity in a region vary with the fraction of area reserved?

iii) Are larger reserves more effective at improving fisheries than smaller ones?

Answering the first question is important to understanding whether the effects on fisheries yields are proportional to area change when habitat declines (e.g., when part of a coral reef is dredged). We found only one study that directly investigated what a fisheries-area relationship looks like (Watson et al. 1993) but there are likely to be more. We suggest that the relationship is likely to be linear at least for larger areas, such that doubling habitat area doubles overall yields (Figure 15a). However, we also predict that there is likely to be an area threshold (Roberts et al. 2003), below which yields decline disproportionately with habitat loss. Whether the threshold corresponds to a large (purple line) or a small area (blue line) is likely to depend on the extent to which there is low or high species redundancy for fisheries production, and the extent to which the system has top-down regulation. For high redundancy and bottom-up regulation, fisheries per unit area are likely to depend essentially on the primary productivity of the habitat, and will likely remain unaffected by habitat loss up to very small areas, when area finally becomes too small to retain viable populations of any exploited species. For systems with low redundancy and/or strong top-down community regulation, a threshold effect is likely to take place as the area tips below the minimum viable area needed to support the most sensitive targeted species and/or keystone species. As discussed above, there is evidence that marine systems are at least in some cases affected by strong top-down regulation (e.g., by predators, large herbivores; Jackson et al. 2001b; Folke et al. 2004). There is also evidence that increasing species diversity increases fisheries productivity (Worm et al. 2006), indicating low functional redundancy (Micheli & Halpern 2005). This suggests that there may be a relatively high threshold in the relationship between area and fisheries provisioning (a: purple line). These thresholds are likely to become even higher when considering the long-term resilience of fisheries productivity.

We found no empirical tests investigating how the fraction of a region that is reserved (in no-take zones) affects overall fishing productivity in the region. Theoretical metapopulation models (Botsford et al. 2003) suggest that the effect will vary with the overall fishing effort. For relatively low fishing effort, high catches are obtained even in the absence of reserves; they can be improved by a low level of reservation, but beyond that an increase in reserve area reduces catches as it reduces the area available for fishing. For high fishing efforts, though, low levels of reservation may be unable to avert population collapses, and therefore null catches (this is because the movement of animals and cheating of fishers around boundaries depletes stocks near the boundaries even within a reserve, and so if the reserve is too small there is no protection rendered); in this case, it may pay to set aside a substantial fraction of the region as reserves.

We did not find studies that investigated directly how the size of individual marine reserves (no-take zones) affects fishing yields in the surrounding areas. There have however been several studies that measured differences in fish communities (diversity, biomass and/or density) inside and outside protected areas. It is assumed that positive differences result in higher spill-over effects resulting in improved fisheries in the surrounding areas (Roberts et al. 2001; Halpern 2003). Recent reviews have investigated whether these differences vary with reserve size: most found no relationship (Cotê et al. 2001; Halpern 2003; Guidetti & Sala 2007) but one found a positive relationship (Claudet et al. 2008). If there is no relationship with area, then many small reserves are predicted to be a better option for maximising fisheries within a region (as they provide higher edge:area ratios and therefore more gain from spills over for the same area), although larger reserves are likely to be needed to ensure long-term viability (Roberts et al. 2003). Even if in general there is no relationship between reserve effect and area (Figure 15b, blue line), there is likely a threshold below which the effect declines very quickly with area (see discussion above) below which the effect declines very quickly with area, for the same reasons as described above. The size of such threshold has not been determined (Roberts et al. 2003). Reserve effect may also plausibly increase with reserve size, as overall species richness increases (Neigel 2003), but it is likely that it does so with diminishing returns (Figure 15b, purple line).

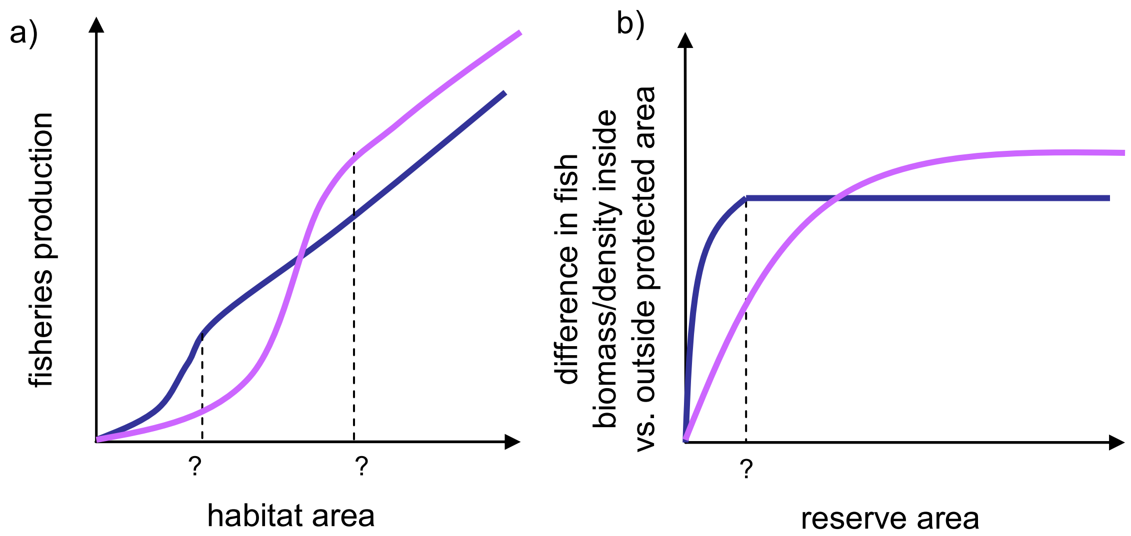

Figure 15. Hypothetical relationship between habitat area and fisheries production, represented in three ways. a) Relationship between area of a given habitat and fisheries production, for lower (blue) and higher (purple) thresholds. b) Relationship between fraction of a region reserved and fisheries production across the entire region, for low (blue) and high (purple) fishing effort. c) Relationship between the size an individual reserve and reserve effect (difference between fish biomass/density inside and outside the reserve, assumed to be an indicator of fisheries catches), assuming no relationship (blue) but an area threshold, or a positive relationship (purple).

4.6.4 What are the main threats to the provision of marine fisheries?

Human activities have now affected all oceans to various degrees (Halpern et al. 2008). Marine fisheries are under threat from different, but often interdependent and self-reinforcing, pressures (Pandolfi et al. 2005).

Overfishing is the most widespread and the dominant direct impact on marine fisheries (Pauly et al. 2005). Fishing pressure is increasing as technological advances allow industrial fleets to fish with greater efficiency, farther offshore, and in deeper waters to meet the global demand for fish. Overfishing is in many parts of the world stimulated by subsidies which either reduce the costs of fishing or increase the net revenue fishers obtain (Pauly et al. 2005). Illegal, unregulated, and unreported (IUU) fishing is estimated to constitute a substantial fraction of the overall catches, undermining efforts for sustainable fisheries management (Pitcher et al. 2002).

Habitat destruction affects the provisioning of fisheries by marine systems. Most is caused by highly destructive fishing techniques, including bottom trawling and coral reef dynamiting. Expansion of coastal infrastructure may affect marine habitats directly (e.g., dredging during port construction) or indirectly (e.g. by increasing sedimentation). Underwater mining activities (minerals, gas and oil) pose an increasing threat. Some systems may recover fairly quickly, but others, such as cold-water corals and seamounts, may take hundreds of years to recover (Pauly et al. 2005).

Pollution is particularly prevalent in coastal habitats, including through sewage discharge and agricultural runoff. In some regions, the extremely high nutrient loads trigger hypoxia conditions known as dead zones (Malakoff 1998; Rabalais et al. 2002). The open seas can also be affected by pollution, for example oil spills. Past waste dumping, particularly radioactive waste, poses a significant threat to deep-sea ecosystems should the containers leak, which seems likely over the long term (Pauly et al. 2005). There is still poor understanding of the effects of persistent organic and inorganic pollutants on marine fisheries (Pauly et al. 2005).

Climate change is predicted to have both direct and indirect impacts on marine fisheries. Direct effects act on physiology and behaviour and alter growth, development, reproductive capacity, mortality, and distribution of targeted species. Indirect effects alter the productivity, structure, and composition of the ecosystems on which fish depend for food and shelter. Some of the changes are expected to have positive consequences for fish production, but in other cases negative impacts are predicted (Brander 2007). Ocean acidification is predicted to have dramatic impacts for calcifying organisms, including coral reefs (Hoegh-Guldberg et al. 2007) as well as some types of zooplankton, particularly at high latitudes (Orr et al. 2005).

Other current threats include biological invasions, such as by competitors (e.g. Daskalov et al. 2007) or pathogens (e.g. Mumby et al. 2007), while future threats may include larger-scale projects for carbon sequestration through iron enrichment (Pauly et al. 2005).

4.6.5 Are abrupt changes likely in the provision of marine fisheries?

Fisheries yields of individual species are well-known to be subject to sudden collapse following overexploitation, with stocks of slow-growing, slow-maturing species in particular often failing to recover to former levels of abundance (e.g., Hutchings 2000; Watson & Morato 2004).

There are many documented examples of recent sudden regime shifts in marine systems (e.g. Jackson et al. 2001b; Bellwood et al 2004; Frank et al. 2005; Daskalov et al. 2007), with implications for fisheries provisioning. Such shifts seem to be particularly likely in ecosystems that are or have been under intense fishing effort, and which have been simplified by the loss of one or more higher-trophic functional groups. Many apparently stable systems have very little resilience, with shifting between stages possibly depending on small changes in the abundances of one or a very few species (e.g., Mumby et al. 2007).

While the collapse of entire fisheries has been observed across relatively large areas (e.g. Black Sea; Daskalov et al. 2007) more often the collapse of a particular species or set of species results in a shift in fishing effort towards other species (often further down in the food web; Pauly et al. 1998) or towards other regions/ecosystems (e.g. towards increasing depths; Pauly et al. 2005). These shifts mask the underlying sequential collapses from ocean-level or global fisheries statistics. Under current knowledge, it is therefore unlikely that a synchronised global collapse will be observed by 2025, but it is very likely that the steady decline that has been observed since the mid-1980s will continue, with some local- or regional-sclae collapses (Pauly et al. 2005).

Climate change (Brander 2007) and related ocean acidification (Orr et al. 2005) are the greatest sources of uncertainty in predictions of marine fisheries, and potentially could be responsible for sudden, large-scale, changes in the foreseeable future.

Overall, we predict that it is almost certain that fisheries collapses will continue to occur amongst overexploited species, and that local regime shifts will take place in particular systems that have already suffered extensive overfishing.

We predict that there is a high likelihood that such changes will be observed at regional scales.

It is unlikely that a synchronised collapse in marine fisheries will take place at the global scale. However, climate change and ocean acidification have the potential to trigger sudden, large-scale effects on marine fisheries.

4.6.6 Can we quantify and map the global provision of marine fisheries and how it might change?

State of knowledge and data availability

Fisheries management has traditionally relied on single-species models aimed at guiding harvesting decision for particular species of interest (Hilborn et al. 2003b). In the last two decades, there has been a shift towards ecosystem-based fisheries management, stimulated by the development of ecosystem modelling (Christensen & Walters 2005). These are particularly relevant as tools for evaluating scenarios and trade-offs associated with particular policy options, and as such particularly relevant for the Review on the Economics of Ecosystems and Biodiversity. A diversity of models have been developed and these are reviewed and compared in Plaganyi (2007).

One of the most widely used models is the Ecopath with Ecosim (EwE) modeling approach (Christensen & Walters 2005), with development spearheaded by the University of British Columbia's Fishery Centre (www.ecopath.org). EwE combines software for ecosystem trophic mass balance analysis (Ecopath), with a dynamic modeling capability (Ecosim) for exploring past and future impacts of fishing and environmental disturbances as well as for exploring optimal fishing policies. Recent versions of the software have brought Ecosim much closer to traditional single-species stock assessment, by allowing age-structured representation of particular, important populations and by allowing users to 'fit' the model to data. Ecosim models can be replicated over a spatial map grid (Ecospace) to allow exploration of policies such as marine protected areas, while accounting for spatial dispersal/advection effects and migration (Christensen & Walters 2004). EwE has already been applied for regional-scale policy exploration aiming to achieve economic, social and ecological sustainability objectives (Pitcher & Cochrane 2002).

Based on the EwE approach and software, the University of British Columbia's Fishery Centre has just developed a global marine model to explore scenarios for the world's oceans (Alder et al. 2007). EcoOcean is spatially defined by the 19 FAO fishing areas covering the world's oceans and driven by effort of five fleets. The model was constructed using 43 functional groups, selected with special consideration for exploited fish species but intended to include all major groups in the oceans. The model relies on global fisheries datasets available through the Sea Around Us Project (SAUP) and FAO websites, in particular fish catches, ex-vessel prices, and fleet statistics. The model output from EcoOcean can be used to describe how landings, profits, the marine trophic index and depletions may change under different policy scenarios in different areas of the world. For the first time, EcoOcean provides a common reporting platform so that the outcomes of the different scenarios can be compared within and between geographic areas, as well as for fleets and fisheries.

Fishing effort is the most important driver for the ecosystem model simulations, and so EcoOcean can be applied to investigate the consequences for fisheries of policy actions under different scenarios, such as a variation in the area covered by marine reserves, or the implementation or not of fishing quotas for particular groups of species. It also tracks changes in community structure, for example in the Mean Trophic Index (Pauly et al. 1998) However, currently the finest spatial resolution possible for the scenario comparison is that of the 19 FAO fishing areas used to develop the model. The model is therefore not appropriate for investigating the consequences of finer-scale changes, such as the implementation of particular marine reserves, the loss of a coral reef, or local changes in water quality as a result of terrestrial runoff, for example leading to the creation of dead zones. The next version of EcoOcean has a planned spatial resolution of half-degree cells (Alder et al. 2007), which will allow greater sensitivity to these changes. Further ongoing developments in the quality of the dataset by the SAUP will improve the accuracy and reduce the uncertainty associated with the models.

These limitations notwithstanding, EcoOcean seems a good candidate for evaluating the consequences of policy measures towards reducing biodiversity loss on the provisioning of marine fisheries. Indeed, EcoOcean has already been applied to explore the scenarios posed in the Global Environment Outlook 4 (UNEP 2007) and the International Assessment for Agricultural Science, Technology and Development, demonstrating its usefulness as a policy tool. The outputs from these two assessments have provided policy makers with plausible results on which to base future decisions regarding management of fisheries and marine ecosystems.

For other models of marine ecosystem management, see Plaganyi (2007).

Adequacy of scenarios

Adequate scenarios for application of marine ecosystem management models must specify fishing effort across space, and for each functional group.

What can be done in Phase 2? At what cost? By whom?

EcoOcean can be used in its current state to compare how the provisioning of marine fisheries (landings) is expected to differ for scenarios varying in fishing effort, across FAO area, and across functional groups.

In our opinion, the research group better positioned to lead these efforts is the University of British Columbia's Fishery Centre, developers of EwE and of the EcoOcean model[2].

4.6.7 Insights for economic valuation

Appropriate scenarios will generate different states of the world, with corresponding predictions of fisheries intensity across the world's oceans, for different functional groups. EcoOcean can be used to investigate how these translate into long-term trends in fisheries yields for each state of the world. Data on the market value of those fisheries, and appropriate discount rates, can be used to monetise the difference in fisheries yields between different states of the world. This market value should reflect predicted variations in demand for different regions/functional groups. For example, increasing demand in fishmeal for aquaculture may result in increasing prices for small, pelagic fish. An increase in recreational fisheries, on the other hand, may lead to increasing value in the stocks of large, high-trophic species.

4.6.8 Some key resources

- Fishery Centre at the University of British Columbia, Vancouver, Canada, is a leading centre of research on global marine systems, including the development of modelling tools such as Ecopath with Ecosim (EwE), a freely available ecological/ecosystem modelling software suite. The Fishery Centre is also the home of the Sea Around Us Project (SAUP) that actively collates and analyses data on global fisheries catches, fishing effort and costs of fishing to investigate the impact of fisheries on the world's marine ecosystems to support the development of sustainable, ecosystem-based fisheries policies. SAUP is led by Daniel Pauly and includes amongst other leading experts Jackie Alder, Villy Christensen, and Reg Watson.
- The Food and Agriculture Organisation (FAO) Fisheries and Aquaculture Department collates and distributes global data on catches, value, consumption, fleets and trade for marine fisheries.
- United Nations Environment Programme (UNEP) Global Marine Assessment: following from a call at the World Summit on Sustainable Development (WSSD) for the "establishment by 2004 and under United Nations of a regular process for global reporting and assessment of the state of marine environment, including socio-economic aspects, both current and foreseeable, building on existing regional assessments", and subsequent approval by the United Nations General Assembly, effort is underway to evaluate all the recent assessments at regional and national

[2] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

scales for fisheries as well as other marine resources/issues. A first report Survey of Global and Regional Marine Environmental Assessments and Related Scientific Activities has been completed by UNEP-WCMC.

- A Global Environmental Facility funded Large Marine Ecosystems Strategy is underway for the assessment and management of international coastal waters, as a joint effort between the IUCN – The World Conservation Union, the Intergovernmental Oceanographic Commission of UNESCO (IOC), other United Nations agencies, and the US National Oceanic and Atmospheric Administration (NOAA). This programme is led by Ken Sherman at NOAA.
- The Global Marine Species Assessment is based at Old Dominion University, VA, USA, coordinated by Kent Carpenter. A partnership between the IUCN Species Survival Commission and Conservation International's Centre for Applied Biodiversity Science, this project will be the first global review of the conservation status of every marine vertebrate species, and of selected invertebrates and plants. The project involves a range of partners in compiling and analyzing all existing data on approximately 20,000 marine species, and will determine the risk of extinction according to the IUCN Red List Categories and Criteria. The goal is to complete Red List assessments of approximately 15,000 marine species by the year 2010.
- The Census of Marine Life a global network of researchers in more than 80 nations engaged in a 10-year scientific initiative to assess and explain the diversity, distribution, and abundance of life in the oceans. The world's first comprehensive Census of Marine Life-past, present, and future-will be released in 2010.
- FAO Fisheries and Aquaculture Department provides statistics on fisheries commodities and trade, global production, fishing fleets, fishers, and consumption of fish and fisheries products.
- SAUP is actively collating and making freely available data on global fisheries catches, fishing effort and costs of fishing.
- The Stock Recruitment Database consisting of maps, plots, and numerical data relating to over 600 fish populations (over 100 species) from all over the world, collated by the late Ransom Myers at the Department of Biology at Dalhousie University in Halifax, Nova Scotia, Canada.
- FishBase is was developed at the WorldFish Center in collaboration with the FAO and other partners. FishBase is a relational database with information on the ecology, distribution, and conservation of ~28,500 marine and freshwater species (practically all fish species known to science).

4.6.9 Participants

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4.7 Inland fisheries and aquaculture

This section is a fully developed review, including contributions by experts in the field, some of whom subsequently reviewed the text.

4.7.1 Why are inland fisheries and aquaculture important for human wellbeing?

Inland capture fisheries and aquaculture provide globally important sources of food and income. They account for 27% of global fisheries production: 9.6 million tonnes (7%) from capture fisheries and 28.9 million tonnes (20%) from aquaculture. Ninety percent of production is in Africa and Asia, with China alone accounting for 70% of inland aquaculture worldwide FAO 2006b). Inland capture fisheries and aquaculture employ more than 50 million people directly and many more indirectly (Finlayson et al. 2005). These fisheries are almost entirely finfish, with some regional exploitation of crustaceans and molluscs (Wood et al. 2005). Three types of freshwater fisheries are covered in this section:

Subsistence fisheries are important for their roles in nutrition, food security, income generation and informal employment. They are particularly important in certain regions, such as SE Asia and Africa, where they can be the only source of animal protein (Finlayson et al. 2005). These fisheries rely on a broad diversity of species and corresponding diversity of fishing gear (Coates et al. 2003). In Asia, many people practise integrated agriculture-aquaculture, maintaining and fishing the fish fauna of rice paddies as an additional income or food source (Dugan et al. 2006). There are very few reliable official statistics available, and the available figures are thought to be large underestimates (Finlayson et al. 2005; FAO 2006b). For example, Coates et al. (2003) estimated the total fisheries production of the Lower Mekong Basin to be between 2.6 and 21 times higher than official statistics. Similarly, in Africa the lack of quality information has meant that FAO has had to estimate the catch for over half of the countries where inland fisheries were known to occur (FAO 2006b).

Commercial fisheries, including artisanal commercial fisheries, are traded for their economic value. They are mainly centred on the Great Lakes of North America, Africa, and Asia (Baikal remains a substantial fishery; the large lakes of China are intensively fished) (Finlayson et al. 2005; Coates et al. 2003). For example, the Great Lakes of North America have supported one of the largest freshwater fisheries for over a century, valued (along with sport fishing) at $4 billion annually (Finlayson et al. 2005). In Cambodia, annual harvest at the Tonle Sap Great Lake exceeds that of all five North American Great Lakes (Campbell et al. 2006). The world's major river systems, such as the Amazon, also support important commercial fisheries, possibly rivaling those in large lake, but statistics are worse as effort is more diffuse. The global value of tropical river fisheries in Latin America, Africa and Asia is estimated at US$ 5.58 billion (gross market value), equivalent to 19% of the current (2004) value of annual fish exports from developing countries (US$ 29 billion) (Neiland and Béné 2008). Many freshwater commercial fisheries are mostly traded extensively, with a few species (e.g. salmon, Nile perch) making it to the international market.

An increasing proportion of the world's freshwater fish production is coming from aquaculture-based systems. These use feeding and restocking activities to produce a higher yield ('agriculturalisation' of freshwater systems; Wood et al. 2005). Often, there is a focus on a few productive species, rather than the ecosystem as a whole. For example, the Great

Lakes fisheries in North America consist of a mix of native and introduced species, which are maintained by regular stocking (Finlayson et al. 2005). The Lake Victoria fishery, which was once based on over 500 species, now relies on the exotic Nile Perch and Nile Tilapia for ~90% of its catch (Njiru et al. 2008). There is a continuum of increasing levels of human input in freshwater systems - from purely capture fisheries, to stocking (release of large numbers of young fish to feed and grow in the ecosystem), to true aquaculture (rearing and feeding of fish until these acheive a marketably large size). As a result, these two types of fisheries are often difficult to separate in practice.

Recreational fisheries are dominant in northern temperate countries, where they contribute significantly to the economy. In Europe, for example, the inland recreational fishing industry has been valued at $25 billion/year. This move towards recreational fisheries is also increasing in developing economies (e.g. Brazil, India, Argentina, and several states of the Zambezi River basin), often to boost the local tourism economy (Dugan 2006). These fisheries tend to focus on a few native and/or introduced prize species, with a substantial proportion being catch-release. In some areas, there is an increasing trend to manage freshwater systems (including through stocking, feeding, and habitat management) to favour recreational species, particularly in reservoirs (Finlayson et al. 2005). Note that other types of (non-extractive) nature-related recreation activities are dicussed in Section 4.12.

4.7.2 What are the overall trends in the provision of inland fisheries and aquaculture?

Catches from inland waters, have shown a slowly but steadily increasing trend (about 2%/year) since 1950, owing in part to stock enhancement practices. However, there are different trends across regions. Catches are declining in Europe, Canada and the United States; for example, Europe has seen a 30% decline in catches since 1999 (FAO 2006b). In some cases, increases are due to better recording of subsistence fisheries, for example in the Lower Mekong Basin (Dugan et al. 2006).

Aquaculture continues to grow more rapidly than all other animal food-producing sectors, with an average annual growth rate for the world of 8.8 percent per year since 1970. However, there are signs that the rate of growth for global aquaculture may have peaked, although high growth rates may continue for some regions and species (FAO 2006b).

Recreational fisheries are increasing as a result of urbanisation, increasing affluence and larger populations, in both developed and developing economies. A significant proportion of the catch from inland waters in developed countries is from recreational fisheries (FAO 2006b), and they too have the potential for dramatic impacts on inland fish stocks (Cooke & Cowx 2006).

4.7.3 How is the provision of inland fisheries and aquaculture affected by changes in wild nature?

The production of inland fisheries ultimately depends on the productivity of targeted species. In many systems, fish growth is controlled 'bottom-up' by the productivity of photosynthetic algae in the water column and on the bottom, which in turn are largely influenced by physicochemical factors such as the availability of nutrients and light (some deep lake demersal fisheries also depend on detrital fallout, also a result of primary production in the surface waters). 'Top-down' control also occurs when predators regulate their prey populations and thereby govern primary productivity through cascading effects on lower trophic levels (Carpenter 1987; Carpenter & Kitchell 1988).

The overall increases in inland fisheries catches mask the underlying decline in the biomass of many individual stocks as most inland fisheries in the developing world are heavily overexploited (Finlayson et al. 2005; Wood et al. 2005; Dugan et al. 2006). As with marine fisheries, there has been a decline in the mean trophic level of inland fish catches over the last few decades ('fishing down the food web'; Pauly et al. 1998; Welcomme 1999; Wood et al. 2005). This tends to result in the replacement of larger, slower-reproducing species with smaller, fast-reproducing species (including invertebrates), whilst the total productivity may remain constant or even increase initially as predators disappear (Welcomme 1999). Hence individual species declines are masked as fisheries switch to other species, as well as by the use of enhancements (such as stocking or feeding) which are increasingly required to maintain yield, and by the general rising production of all types of aquaculture (Finlayson et al. 2005; FAO 2006b). Though there is no evidence for a general collapse of inland fisheries, there have been instances of local collapses related to modifications of freshwater systems, such as dam building (e.g. the loss of migratory guilds after the building of a dam on the Mekong; Amornsakchai et al. 2000).

There is little evidence for an overall relationship between species diversity and freshwater fisheries yields. Indeed, increases in productivity have been observed in some systems despite dramatic declines in diversity; for example, catches from Lake Victoria increased from 30,000 mt to an average of 500,000 mt since the 1970s, despite the loss or near-loss of about half the native species, with the introduced Nile Perch now making around 90% of the landing's volume and value (Njiru et al. 2008). However, it is likely that species diversity (both of targeted species and others) improves the resilience of capture fisheries and aquaculture to anthropogenic and environmental changes, as seems to be the case with marine fisheries (Worm et al. 2006; Section 4.6). Indeed, many inland fisheries are being simplified to depend on a few species that do well in the modified habitats that we are creating. While this seems to be working well for fisheries in the short-term, it may be risky in the long-term if conditions change (e.g. climate warming).

In the transition to more intensively managed freshwater fisheries, the tendency is to focus on a few more desirable and/or more productive species. However, the genetic diversity of managed populations remains very important, both as a source of variation for breeding in desirable traits and to increase the resilience of farmed fish to disease. Also, a broad set of freshwater species are being domesticated/selected for aquaculture production, indicating a greater reliance on species diversity than in other food production sectors (FAO 2006b).

A key peril of eroding freshwater biodiversity is the risk of losing species that play critical functional roles in the ecosystem (e.g. keystone species). For instance, the loss of anadromous salmon from rivers results in a substantial reduction in nutrient inputs (Allan et al. 2005), and declines of heavily-fished species can compromise nutrient recycling rates that support ecosystem productivity (McIntyre et al. 2007). Beavers and some detritivorous fish substantially change the physical structure of freshwater habitats (Rosell et al. 2005; Collen & Gibson 2001; Taylor et al. 2006 Science), thereby playing an important role as ecosystem engineers.

Also, there is still a very broad set of freshwater species being domesticated/selected for aquaculture production, and so there is a greater reliance on species diversity than in other food production sectors (FAO 2006b; this is covered in Section 4.14). A diversity of species is also important for the colonisation of new habitats (such as reservoirs) created by freshwater modification. Often, species from the previous habitats (e.g. flowing water environments) are unable to establish themselves, and in this case other species are frequently introduced, such as cichlids, bass, and carps (Finlayson et al. 2005).

In the transition to more intensively managed freshwater fisheries, the tendency is to focus on a few more desirable and/or more productive species. However, the genetic diversity of managed populations remains very important, both as a source of desirable traits and to increase the resilience of farmed fish to disease.

Freshwater biodiversity is highly threatened, including for example 56% of endemic fish in the Mediterranean basin (Smith & Darwall 2006), and 54% of endemic fish in Madagascar (Darwall et al. 2005).

Freshwater communities are affected by loss, degradation, and transformation of freshwater habitats (Finlayson et al. 2005). Furthermore, many freshwater species use different habitats for breeding, for hunting and for nurseries (e.g. hundreds of species migrate from large rivers/lakes to streams in order to breed). This renders them particularly vulnerable, as they can be affected by the degradation of any of those habitats, as well as by loss of connectivity between habitats, such as through the construction of dams or separation of floodplains from the main river channel. Communities in lotic systems (rivers, streams) are frequently dependent on particular hydrological regimes, which can be disturbed by land use changes (e.g. logging), water abstraction, and dam building (Finlayson et al. 2005; Welcomme & Halls 2004). These systems are also affected by river canalisation, separation from the floodplain, and water pollution. Lentic systems (lakes, reservoirs, wetlands) are particularly vulnerable to eutrophication (e.g., in Lake Victoria a doubling of nutrient input led to an 6-8–fold increase in algal biomass; Balirwa 2007), sedimentation (e.g. forest loss around Lake Malawi resulted in increased sedimentation, and reduced quality habitat for cichlid species; Rusuwa et al. 2006), the introduction of invasive species (e.g. Nile perch in Lake Victoria; Njiru et al. 2008; Balirwa 2007), water abstraction (e.g., Aral Sea; Finlayson et al. 2005), and loss/decline of area due to land use change (e.g. Lake Victoria has lost wetland habitats at a current rate of ~3.8% annually; Njiru et al. 2008). Floodplains are key habitats for many species, particularly for breeding, and they are being threatened by disconnection from main river channel and by habitat loss as they become appropriated for agriculture and human settlements.

Changes in the habitats within the water catchment area of freshwater systems (including riparian habitats) can have important and widespread effects on the quality, extent and composition of freshwater ecosystems (Finlayson et al. 2005). Terrestrial systems can contribute to the regulation of energy and material transfer between the riparian and aquatic ecosystems, particularly in small water bodies (Pusey & Arthington 2003). Thus, changing land use affects water quality (e.g., via energy, nutrient, and toxin inputs; Allan & Castillo 2007), turbidity and sedimentation (e.g. erosion following deforestation and road building; Alin et al. 1999), and hydrologic regimes (e.g., changes in quantity and timing of water flow following deforestation; Allan & Castillo 2007; Welcomme & Halls 2004). Terrestrial habitats can also provide nutrition to freshwater species (e.g. fruit eating fish; Araujo-Lima & Goulding 1997) and contribute to the physical structure of freshwater habitats (e.g. providing woody debris; Wright & Flecker 2004).

While the negative effects of habitat changes on freshwater communities are well-established, the consequences for freshwater fisheries are less studied. In some cases, the creation of new habitats favours particular species, which in turn can be exploited in fisheries. Examples include reservoirs following dam construction, and rice fields following the conversion of floodplain habitat (Dugan et al. in press). In many cases, a few species benefit whilst most others lose out, and so the advantages/disadvantages for fisheries depend on both the fishery value of thriving species and the consequences of altered habitat structure and species assemblages for ecosystem stability and functioning.

Relationship between habitat area and the provision of inland fisherie and aquacultures

In principle, freshwater fisheries might be expected to decline in direct proportion ot loss of the area of relevant habitat (e.g. floodplain, rivers). However, reality is likely to be much more complex, given the intrinsic interconnectivity of freshwater systems. Hence, fish production in a stretch of river may not be maintained if upstream reaches are degraded, and massive floodplain fisheries depend on the connection to and exchange with the main river channel.

4.7.4 What are the main threats to the provision of inland fisheries and aquaculture?

Water abstraction (particularly for agriculture) is among the greatest global threats to inland fisheries and aquaculture, altering flowing regimes and reducing river flow in riverine systems (sometimes completely, as in sections of the Colorado River), reducing the area of available habitat in lentic systems (e.g. Aral Sea, where water volume has been reduced by 75% since 1960; Finlayson et al. 2005), and allowing saline water intrusion in coastal freshwater systems (Finlayson et al. 2005). Water translocation is also an increasingly important consideration, as water is diverted from one river basin to another (e.g. Yellow River in China). Fisheries are directly impacted as the flood cycle and water availability are disrupted, and there is an increased threat from invasive alien species as river basins are linked.

River canalisation and separation from the floodplain changes hydrological regimes substantially, affecting communities in both river systems and floodplains. Floodplains are also directly threatened by habitat loss, as the land is appropriated for agriculture and human settlements. Indeed, floodplain systems are one of the most threatened habitats in the world; Europe, for example, has lost up to 95% of its riverine floodplains, and those that are left are often decoupled from the hydromorphological dynamics of the river (Tockner & Stanford 2002).

Dam building destroys habitats upstream, causes major disruption to hydrological and sedimentation flows downstream, creates a barrier to species migrations, and fragments resident species populations. Fisheries based on migratory species may be totally compromised (e.g. sturgeon fisheries). However, fisheries may improve in the reservoir area when there is restocking with suitable (frequently alien) species (Finlayson et al. 2005).

Overexploitation leads to population reductions and even extinction of the targeted species. A FAO major assessment of inland fisheries (1999, in Finlayson et al. 2005) reported that most inland capture fisheries that rely on natural reproduction of the stocks are overfished or are being fished at their biological limit.

Water pollution and eutrophication (particularly due to agriculture, but also urban and industrial run-off) significantly affect freshwater communities. Eutrophication, caused by increased nutrient loads (particularly due to fertilisers in agricultural runoff) is the most widespread problem in lakes and reservoirs, and also one of the most difficult to abate. Toxic contaminants (e.g. heavy metals and organochlorine compounds) may render fish unusable for human consumption (Finlayson et al. 2005)

<u>Landuse changes in terrestrial habitats</u> (e.g. deforestation) affect water and sediment regime flows and water quality.

Invasive species may affect fisheries directly, by predating, competing with, or parasitizing the targeted species, or indirectly, by co-introduction of disease organisms, or by affecting habitat characteristics (e.g. water hyacinth) and water regimes (e.g. mimosa; Finlayson et al. 2005).

<u>Climate change</u> is likely to affect inland fisheries and aquaculture by altering river flows, lake levels, and the internal mixing dynamics of large lakes. For example, O'Reilly and colleagues (2003) found that global warming reduced productivity of Lake Tanganyika, with strong implications for a 100-200 mtonnes fishery.

Many aquaculture practices also contribute to the degradation of inland water systems, via <u>pollution</u>, the <u>spread of diseases</u> and the <u>introduction of alien species</u> (Finlayson et al. 2005).

It is important to keep in mind that some anthropogenic changes are also resulting in increased fishery yields, due to stocking and feeding (particularly of small lakes and reservoirs) and management of habitat structure for particular target species (Finlayson et al. 2005; Njiru 2008).

4.7.5 Are abrupt changes likely in the provisioning of freshwater fisheries and aquaculture?

As discussed above, there is little evidence of widespread collapse in freshwater fisheries at the community level as a result of general overharvesting. However, collapses and local extinctions of individual species have taken place, and overfishing has clearly impacted the species composition, body size, and total biomass of fishery catches in many individual lakes and rivers, leading to a general assessment that inland waters are overfished (Allan et al. 2005). For instance, commercial fisheries in the North American Great Lakes peaked in the late 19^{th} century, and a combination of overfishing, species introductions, pollution, and other factors has yielded lower, fluctuating yields since that time. Similarly, Lake Baikal was overfished in the early 20^{th} century, and catches have never recovered. It is difficult to predict the impact of the 'fishing down the food web' phenomenon, which has led to collapses in some marine fisheries (Section 4.6), in freshwaters because many lakes and rivers are intensively managed through stocking, feeding, and habitat manipulation. These practices prop up the populations of predatory fish that are the target of most freshwater fisheries.

Human-induced eutrophication can trigger sudden shifts in lakes and reservoirs from clear to turbid water, due to algal blooms. These blooms may include toxic cyanobacteria and affect freshwater fisheries and recreational use of the water bodies. Reduction of nutrient concentrations is often insufficient to restore the original state, with restoration requiring substantially lower nutrient levels than those at which the regime shift occurred (Scheffer et al. 2001).

Species invasions can also cause abrupt disruption to fisheries and aquaculture. Lake Victoria offers the most dramatic case study; the traditional artisinal fishery for native cichlids was transformed into a massive industrial fishery by the explosion of the exotic Nile Perch in the 1970s-90s. In the North American Great Lakes, the loss of native lake trout fisheries is believed to have resulted from the proliferation of exotic, parasitic sea lampreys in the 20th century.

Climate change has the potential to cause widespread disruption to inland fisheries and aquaculture. At present, there is little direct evidence for climate change effects on fisheries. However, records from Lake Tanganyika have been interpreted as evidence of climate forcing of algal productivity, and may help to explain declining catches in that globally-significant fishery (O'Reilly et al. 2003).

Aquaculture is particularly prone to abrupt changes, particularly through the spread of diseases. Several cases have occurred in the past, for example with the brackish water shrimp fisheries of Asia and S. America or salmonid culture in Europe. The establishment and spread of diseases is frequently linked to environmental stresses.

Overall, we predict that there is a high probability of localised disruption to inland fisheries and aquaculture as a result of eutrophication, and species invasions. Large-scale changes may result from climate change.

4.7.6 Can we quantify and map the global provision of inland fisheries and aquaculture, and how it might change?

There is currently no overall global model for the productivity of freshwater fisheries and aquaculture. Different models exist for different systems in different areas, including for example:

- *Tropical river floodplain* – models exist that investigate the impact of hydrological changes on fish yield/productivity, combining a hydrological and population model to simulate production changes with different water regimes (Halls & Welcomme 2004). Others use Bayesian networks to examine the impacts of wide ecosystem changes on fish production (e.g. Baran et al 2007)

- *Northern temperate lakes* – the Morphoedaphic Index (Ryder 1965) has been extensively used as a predictor of fisheries yields based on total dissolved solids and lake depth. However its accuracy is contested, and there are many subtle variations in analysis.

- *African water bodies* – Morpho-edaphic models were used extensively to estimate the potential of Africa lakes and Reservoirs and models which predict fisheries production have been generated by the Africa Water Resources Database. These models use physico-chemistry and climatological maps to predict production based (mainly) on surface area.

It is likely that different freshwater systems will need to be modelled separately. As with marine fisheries (Section 4.6), it is key that models reflect sustainable production, rather than current catches, as the latter are likely to be unsustainable in the long term.

Except for some well-studied systems, the state of knowledge regarding freshwater fisheries is very poor. FAO has historically managed global statistics, but there is widespread recognition that meaningful data need to be collected and analyzed at local, national and basin scales. (e.g. Mekong River Commission Fisheries Programme).

What can be done in Phase 2? At what cost? By whom[3]?

The experts we consulted were of the opinion that a reasonable model, or at least a first estimate, could be generated within one year, by reviewing existing studies on sustainable freshwater fisheries production across the world and extrapolating from those. Two possible approaches were proposed:

a) Engage a group of 12-15 experts covering different regions, including people with good skills in GIS and statistical modelling (see suggested names under key resources), and based on existing models and statistics ask them to provide a predictive collective opinion of how different states of the world would differ in inland fisheries and aquaculture production. One or several collective meetings (NCEAS-style workshops) would be needed.

b) As in a), but preceded by regional meetings to collect and collate existing (but currently dispersed) regional information on which to base finer estimates for productivity change.

It is estimated that it would take around 36-48 researcher-months for a suitable team to compile a reasonable global estimate from existing knowledge (route a). Resources would also need to be allocated for meetings under either approach.

Adequacy of scenarios

In order to be useful in predicting the impacts of changes in wild nature on the provision of freshwater fisheries and aquaculture, the scenarios would ideally provide information on:

- The availability of water (temporally and spatially), particularly affected by abstraction for agriculture and hydroelectric projects.
- In the extent of natural floodplain habitats, closely linked to the expansion of agriculture and urban development.
- The connectivity of water systems, particularly the construction of dams.

[3] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

- The canalisation of rivers.
- The levels of harvesting and demand.

4.7.7 Insights for economic valuation

The subsequent economic evaluation, based on the maps of biophysical production, would need to carefully consider what the end use of each type of fisheries is. Different valuation methods would be necessary in each case:

- For commercial fisheries, most data can be taken from the market in fish and fish products (African, Asian, and North American Great Lakes; large rivers).
- For commercial fisheries, most data can be taken from the market in fish and fish produce (African and North American Great Lakes, some large rivers).
- Recreational fisheries are likely to need valuation using data from licenses and expenses, utilising Contingent Valuation and Travel Cost methods (North America, Europe, Australia, some parts of the tropics).

A 2008 report by the WorldFish Centre (Neiland, A., & Béné, C. [Eds.]. *Tropical river fisheries valuation: background papers to a global synthesis*) compiled five regional reviews of valuation studies of tropical river fisheries in Latin America, Africa and Asia.

4.7.8 Some key resources

- Patrick Dugan, at the World Fish Centre, would be in a good position to oversee this project.
- R. Hilborn, at the University of Washington (USA) and E. Jeppesen (Denmark) could be valuable with regards to statistical modelling of fisheries-ecosystem relationships.
- Robin Welcomme, Imperial College, London, and Institute of Fisheries Management, UK.
- Will Darwall, Freshwater Biodiversity Unit, SSC/IUCN, Cambridge, UK: leading the global freshwater assessment.
- Peter McIntyre, School of Natural Resources and Environment, University of Michigan, USA.
- Gerd Marmulla, FAO.
- Robin Abell, Senior Freshwater Conservation Biologist, Conservation Science, WWF-US, Washington DC, USA.
- Carmen Revenga, The Nature Conservancy, Arlington VA, USA.
- Regional expertise:

- *Mekong/Asia:* Eric Baran (WorldFish Centre), Chris Barlow (Mekong River Commission Fisheries Programme); Ashley Halls (Aquae Sulis Ltd, UK).
- *Africa:* Patrick Dugan, Africa Water Resources Database.
- *Latin America/South America:* Miguel Petrere (Universidade Estadual Paulista, Rio Claro, SP, Brazil; for Brazil and much of Latin America); Oriana Almeida (IPAM, Amazonian Institute of Environmental Research, Brazil; for the Amazon); Angelo Antonio Agostinho (Universidade Estadual de Maringá, PR, Brazil).
- *Europe:* Ian Cowx (University of Hull, UK).
- *Australia:* Angela Arthington (Griffith University, Australia).
- *North America:* John Casselman (Queens University, Canada), Stuart Ludsin (Ohio State University, USA).

4.7.9 Participants

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The following experts provided very valuable insights to this section.

Will Darwall (SSC/IUCN, Cambridge, UK)

Patrick Dugan (World Fish Centre)

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4.8 Wild animal products

This section is a fully developed review, including contributions by experts in the field, most of whom subsequently reviewed the text.

4.8.1 Why is this benefit important for human wellbeing?

The harvesting of wild animal products is defined here to include the capture of terrestrial wild animals or their products, including mammals, birds, reptiles, amphibians and invertebrates (for example snails and insects, including honey harvesting). Wild animal harvesting can be broadly classified into three categories: subsistence, commercial, and recreational (Table 2). In subsistence harvesting, the benefits obtained from wildlife (particularly food) are directly consumed or used by, and play a very significant role in the subsistence of, the harvester and its family (e.g. Peres 2000). Commercial harvesting takes place when most of the products are sold for profit (e.g. caiman meat trade; Silveira & Thorbjarnarson 1999; chameleon pet trade; Carpenter et al. 2004). Recreational hunting refers to activities in which the main objective is the personal enjoyment of the hunter, rather than food or profit (e.g. trophy lion hunting in Tanzania; Whitman et al. 2004). There is substantial overlap between these categories. The differentiation between subsistence and commercial harvesting may be subtle (with, for example, subsistence hunters often selling the excess or the most valuable species as a source of income; Wilkie & Carpenter 1999b), and the transition between the two may happen quickly as markets open (e.g. Sierra et al. 1999). Recreational hunting may also have roots in traditional (either subsistence or commercial) hunting activities (McCorquodale 1997).

Table 2. Comparison of main attributes of subsistence, commercial and recreational hunting.

Attribute	Subsistence hunting	Commercial hunting	Recreational hunting
Purpose	Food	Profit	Sport
Users	Rural people, poor	Rural and urban	High income
Volume	High	Very High	Low
Area	Often close to residence, permanent	Distant from residence	Distant from residence, variable
Frequency	Year round	Year round	Weekends, hunting seasons
Principle Prey	Mostly mammals	Mostly mammals	Mostly birds, but medium to large mammals also.
Legality	Variable according to country	Often illegal	Usually legal
Control	Difficult but may be feasible	Often very difficult	Feasible
Social Value	High	Low	Medium

The main benefit obtained from the harvesting of wild animals is arguably food (e.g. Fa et al. 2003), with other benefits including raw materials such as hides (e.g. Iriarte & Jaksic 1986), medicines or substances traditionally considered to have medicinal value (e.g. Alves & Rosa 2005; Ntiamoa-Baidu 1997), pets (e.g. Carpenter et al. 2004), and personal enjoyment (e.g. Wilkie & Carpenter 1999a). Here we focus particularly on subsistence and commercial harvesting for food (referred throughout as 'wild meat') in tropical regions, with some discussion of harvesting for personal enjoyment (recreational hunting).

Wild meat is a valuable source of nutrition across the globe, but its importance as food is particularly high in tropical regions (with some forest-living peoples obtaining up to 90% of animal protein from wild animals; Fa et al. 2003). The literature focuses on large vertebrates (> 1 kg), which seem to contribute the most to wild meat provision (particularly mammals, followed by birds, reptiles and amphibians) and this is the emphasis of this section as well. Invertebrates also have a very significant nutritional role in some areas, but these are frequently ignored in studies of wild food harvesting, and traditional consumption is declining due to a negative bias by Westerners (deFoliart 1999). deFoliart (1999) presents a comprehensive review of the use of insects as food across the world, including for example a study that found that 65 species of insects furnished 10% of the animal protein intake in the Democratic Republic of Congo (compared to 30% from game and 47% from fishing), and the case of Thailand, where more than 80 species of insects (35 families) are used as food, and where locust and grasshopper catching become alternatives to pesticide spraying during plague periods.

Wild meat is important for the poor and landless, especially during times of the year when agricultural production is lower (de Merode et al. 2004), and periods of famine and insecurity or conflict, when normal food supply mechanisms are disrupted and local or displaced populations have limited access to other forms of nutrition (Elliott et al. 2002; Wood et al. 2005; Jambiya et al. 2007). In some regions, though, reliance on wild meat is permanent (see Ntiamoa-Baidu 1997 for a review of wild meat use across Africa). While the dependency on wild meat is greatest in rural communities, townspeople can also be major wildlife consumers (Milner-Gulland et al. 2003). In the transition from rural to urban consumption, wild meat consumption becomes less of a nutritional need (in which case people will readily switch to domestic meat, if it becomes more readily available) and more a matter of preference based on tradition or status (in which case at least some consumption persists even when wild meat becomes substantially more expensive than domestic alternatives) (Milner-Gulland et al. 2003; see Goudarzi [2006] for an extreme example of African wild meat being sold as a luxury product in US and European markets). Urban demand creates the opportunity for trade, and therefore wild meat harvesting is also an important source of income for local people (e.g. Silveira & Thorbjarnarson 1999; Ntiamoa-Baidu 1997).

Estimates of the annual wild meat harvest, mainly mammals, include 23,500 tonnes in Sarawak, 67,000– 164,000 tonnes in the Brazilian Amazon, and 1–3.4 million tonnes in Central Africa (Milner-Gulland et al. 2003 and references therein). Wild meat harvest and trade are often excluded from official statistics (Wood et al. 2005) but the economic value of annual trade has been estimated at, for example, over US$175m for the Amazon Basin and US$200m for Côte d'Ivoire (Rao & McGowan 2002 and references therein).

Recreational hunting is the dominant form of terrestrial wild animal harvesting in the most economically affluent countries. The nutritional value of wild meat is negligible, given wide access to cheaper alternatives. And, quite the opposite of being a source of income, hunting becomes a substantial expense as hunters spend considerable sums for the pleasure and social status of hunting. Regions in Europe and the United States with higher fractions of rural population tend to have a higher frequency of hunters, supporting the hypothesis that the motivation to hunt is based on cultural roots (Heberlein et al. 2002). In the United

States, 5% of the population 16 years old and older went hunting in 2006, spending US$22 billion (USFWS 2007). In the European Union, there are more than seven million hunters (FACE 2007), estimated to spend €10 billion per year (Pinet 1995). Trophy hunting by foreign tourists generates at least US$201 million per year in sub-Saharan Africa, from a minimum of 18,500 clients (Lindsey et al. 2007), while in Eurasia US$33–39 million dollars are generated from 45,000 to 60,000 foreign hunters (Hofer, 2002).

Harvesting of wild animals has costs to human wellbeing as well. It is widely acknowledged that overexploitation is resulting in a worldwide depletion, and sometimes the complete extirpation, of many species. Historically, overharvesting has already contributed to the demise of at least 55 terrestrial animal species (listed by the 2007 IUCN Red List of Threatened Species as Extinct or Extinct in the Wild), and 792 additional species are threatened with extinction from hunting (IUCN 2007). The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), an international agreement between governments that aims to ensure that international trade in specimens of wild animals and plants does not threaten their survival, has listed about 5,000 animals (including terrestrial, marine, and freshwater) for which trade regulations are in place (CITES 2008). Species' extinctions preclude the provision of other present and future benefits. Local depletions and extirpations through overhunting are common (e.g. Brashares et al. 2001) and at a local scale impact human wellbeing as much as global losses. The large-bodied, slow-growing, species that are the most vulnerable to overharvesting (Bodmer et al. 1997) are often charismatic species that can attract tourists and conservation funding, such as tigers, elephants, rhinos, and gorillas (Walpole & Leader-Williams 2002). Hunting also raises ethical and animal welfare issues (e.g. Gilbert 2000), and there is increasing resistance to the idea of killing animals for sport among urban residents in the industrialised world, as highlighted by the recent ban on fox hunting in the UK (Lindsey et al. 2006). Outbreaks in zoonotic diseases (ie shared between humans, domestic animals, and wildlife; e.g. SARS, West Nile virus), causing hundreds of billions of dollars of economic damage as well as mortality in humans, livestock and wildlife, have been attributed to the global wildlife trade (e.g. Bell et al. 2004; Karesh et al. 2005), which has enabled the rapid transmission of zoonoses.

4.8.2 What are the overall trends in the provision of wild animal products?

The Millennium Ecosystem Assessment (MEA 2005) considered that there is a medium to high certainty that wild food provisioning is declining, as natural habitats worldwide are under increasing pressure and wild animal populations are exploited at unsustainable levels (Wood et al. 2005).

The most direct measure of trends in the provision of benefits from wild animal harvesting is derived from the flow of wildlife products, for example through statistics on consumption or trade. However, often these measures cannot be used to extrapolate future (even near-future) trends. Indeed, a steady increase in consumption or in market volumes may persist for some time despite depletion of the source populations. This may occur if there is increased harvesting effort (e.g. traps being replaced by guns), if harvesting extends to species not previously exploited, or if harvesting is expanding to new areas (Crookes et al. 2005). All of these are likely: the higher the demand, the greater the economic incentive to source wildlife products further away, so a thriving market may persist for a substantial period. Ultimately, though, the widespread depletion of animal populations will inevitably result in declines in the supply of wildlife products.

Overexploitation of wild animals is typically linked to human population growth, and consequently it is currently felt most in Asia, followed by Africa and by Latin America (Milner-Gulland et al. 2003). Depletion of animal populations is often associated with habitat loss, but may also occur in forests that otherwise appear to be intact ('empty forest syndrome'; Redford 1992). In Asia, a thriving trade of wild animal products for food, pets and traditional medicines results in widespread population declines and local extirpations (The World Bank 2005; Corlett 2007), with overexploitation being one of the most importants threat to some groups of species, such as amphibians (Stuart et al. 2004) and freshwater turtles and tortoises (van Dijk et al. 2000). In Vietnam, for example, 12 large vertebrates have been extirpated in the past 40 years largely because of hunting (Bennett & Rao 2002). In Africa, the scale of wild meat hunting has accelerated over the last few decades, reaching particularly worrying levels in Central and West Africa (e.g., Brashares et al. 2001; Fa et al. 2006). For example, hunting has been identified as the main cause for the collapse in gorilla and common chimpanzee populations in western equatorial Africa (Walsh et al. 2003). In Latin America, where human population densities and resulting rates of exploitation are still considerably lower than in Africa (Fa et al. 2002), even subsistence hunting can substantially affect population densities and faunal community structure. In the Amazon, the aggregated biomass of the most sensitive species was reduced by half between non-hunted and light-to-moderately hunted sites, and reduced 11-fold between non-hunted and heavily hunted sites (Peres and Palacios 2007). Given that wild meat exploitation in tropical forests is often substantially higher than natural productivity (e.g., Fa et al. 2002) supplies in wild meat protein are predicted to decline (e.g. by 81% in the next 50 years, in the Congo Basin; Fa et al. 2003).

On the other hand, game ranching (the maintenance of wild animals in defined areas delineated by fences) is expanding as a method of wild meat production, particularly in sourthern Africa, although the main aim of these farms is to attract wildlife tourism or trophy hunting (Ntiamoa-Baidu 1997).

As for trends in recreational hunting, participation in the US has declined by about 10% between 1996 and 2001 (USFWS 2007), while numbers of hunters in France are declining by about 3% per year (RSPB 2006). These declines may be explicable by increasing urbanisation (Heberlein et al. 2002), often associated with increasing animal welfare concerns. However, there is evidence that trophy hunting is expanding in Southern Africa and Tanzania, but stable or declining in Central and West Africa (Lindsey et al. 2007).

4.8.3 How is the provisioning of wild animal products affected by changes in wild nature?

The supply of wild meat is largely determined by the <u>ecosystem's productivity of the targeted species</u>. Often there is little consumer preference amongst species and they are mutually interchangeable as protein sources. Hunter effort varies spatially according to drivers such as market access and alternative livelihoods, and combines with overall animal abundance to determine supply. Hunting history affects current supply (Peres 1999).

Large-bodied (average adults ≥ 1 kg) primates, ungulates and rodents account for most of the wildlife biomass hunted by humans for food across the tropics, with ungulates typically making up by far most of the biomass. Robinson & Bennett (2004) reviewed data on the standing biomass of these taxonomic groups across a diversity of relatively undisturbed ecosystems, finding a peak at intermediate levels of rainfall. The increase in biomass with rainfall reflects a positive association between primary productivity and rainfall, while the decline for highest rainfall levels may reflect a reduction in food availability, particularly for ungulates, as much of the plant biomass is in the form of inedible tree trunks and toxic or inaccessible leaves, with a high proportion of the primary productivity high in the canopy (Robinson & Bennett [2004] and references therein). Accordingly, they found that ungulates predominate in open habitats while primates are realtively more common in forested areas. Overall, their results indicate that mammalian standing biomass is low below 100 mm of annual rainfall, that grasslands with more than 500 mm can commonly support mammalian biomasses of between 15,000 and 20,000 kg/km^2, and that the total mammalian biomass in tropical forests rarely exceeds 3,000 kg/km^2 (but see Fa & Peres 2001 for much higher values in Africa). Within a given habitat type, soil fertility can have a major effect on primary productivity and, therefore, on the standing biomass of mammal species; for example, Peres (2000) found that nutrient-rich floodplain forests consistently contained a greater game biomass than nutrient-poor unflooded forests.

The sustainable production of wild meat is affected not only by the standing biomass but also by the species' rate of natural increase. The maximum percentage of standing biomass that can be harvested without resulting in lower absolute harvest (the maximum sustainable offtake rate) is typically lower for long-lived species, estimated by Robinson & Bennett (2004) to exceed 50% for some rodent species, be around 20% for ungulate species, but usually under 5% for primates. Given that, as described above, ungulates and (to a lesser extent) rodents dominate in savanna habitats, while in forest primates play a more dominat role, the maximum sustainable offtake is likely to be higher in the former (maybe as much as 20% of the standing biomass) than in the latter (perhaps 10%; Robinson & Bennett 2004). A higher sustainable offtake rate combined with a larger standing biomass means that savannas are typically more productive than closed forests, and therefore have a higher likelihood of being harvested sustainably (Robinson & Bennett 2004).

While a wide diversity of species is usually harvested, the bulk of the wild meat biomass captured in any one location tends to be dominated by a few species (e.g. Fa et al. 2005, 2006). These, however, change in space and time. At least in South America, large species, such as tapirs, tend to be targeted by hunters first, with efforts turning to smaller species, such as squirrels or rats, as these disappear. This preference for larger-bodied species seems to be explained by a maximisation of return (kg of meat) per unit of hunting effort, rather than intrinsic cultural preferences for particular species (Jerozolimski & Peres 2003). Unfortunately, large species tend to have life-history traits (low reproductive rates, low population densities, long generation time, and long lifespans) that make them particularly vulnerable to overexploitation and less likely to recover once depleted (Bodmer et al. 1997; Jerozolimski & Peres 2003). As the most desirable species disappear, hunters' selectivity declines and they tend to diversify their prey base (Jerozolimski & Peres 2003). Depletion of the large-bodied species may not result in a decline in wild meat supplies; theoretically, it may even result in an increase, as their reduced density allows faster-growing (more productive) species to dominate the ecosystem (density compensation; Peres & Dolman 2000; Jerozolimski & Peres 2003). Sustainable exploitation, in the sense of a future continuous supply of wild meat, may therefore become a more achievable aim in these conditions (Cowlishaw et al. 2005; but see Waite 2007), although this process cannot be considered sustainable in the ecological sense if it requires the extinction or severe depletion of a subset of species. For such a system to be ecologically sustainable, strong regulations would be required to reduce or prevent the exploitation of vulnerable species (Cowlishaw et al. 2005).

Habitat loss inevitably results in the extirpation of habitat-dependent species and the wild food obtained from them. Nonetheless, a temporary increase in the supply of wild meat may be observed during the process of habitat loss, for example as new logging roads are opened, due to increased accessibility to wildlife. In the longer-term, though, supply is expected to decline as populations are depleted by the synergistic effects of hunting and habitat loss (Peres 2001). Habitat fragmentation reduces the habitat available to species averse to the matrix and to edges. Peres (2001) calculated that the minimum fragment area needed to maintain a sustainable harvest of 46 hunted species in the Amazon (given levels of extraction documented to date) was as high as 2000 km^2, with tapirs as the most spatially-demanding species. This is in sharp contrast with the observed size of fragments in the region of the deforestation frontier, with a modal area of less than 1 ha (Peres 2001). However, it is not clear that species diversity per se affects wild meat supply (see below).

Fragmentation also isolates populations, thereby reducing or precluding colonisation of overharvested areas from nearby non-harvested or underharvested regions, disrupting the source-sink dynamics that help maintain the long-term sustainability of game harvesting over large spatial scales (Novaro et al. 2000; 2005). This may explain why surprisingly few local extinctions have been reported in persistently overhunted (but finite) harvested areas within large, non-harvested, forest landscapes (Peres 2001; Ohl-Schacherer et al. 2007). Accordingly, the spatial configuration of harvested and non-harvested areas can affect the resilience of populations to harvest levels (McCullough 1996).

Robinson & Bennett (2004) hypothesised that the conversion of primary forest to secondary habitats (such as forest–farm–fallow mosaics) may increase the supply of wild meat (see also Wilkie 2005). As discussed above, in dense forests much of the plant biomass is in the form of inedible tree trunks and toxic leaves, with a high proportion of the primary productivity high in the canopy. Habitat disturbance opens the canopy, increasing the amount of browse and graze available for herbivorous animals such as ungulates, which can in principle increase the standing biomass of wild animals. However, disturbance is usually associated with increasing human presence (through opening of roads and logging), resulting in a decrease in the biomass of wildlife because of hunting and the introduction of competing livestock. On the other hand, hunting also shifts the composition of the faunal community, so that the biomass of large-bodied, slow reproducing species (including predators) declines, while the biomass of more adaptable, rapidly-reproducing, small-bodied species, might increase. Robinson & Bennett (2004) propose that the consequence of this shift is that even though overall biomass might decline, biomass production might, under certain circumstances, increase with the transition between primary forest and more disturbed habitats. This remains unresolved, with the studies reviewed by Robinson & Bennett (2004) providing some evidence in support of this currently open hypothesis. The effects may differ between continents depending on their respective faunal assemblages. In the Neotropics, most large vertebrates are primarily or partially frugivorous or granivorous (Robinson & Redford 1986b), and so a shift from fruit-based to browse-based plant food resources may not necessarily lead to an overall increase in the standing biomass and/or productivity of wild meat. Irrespective of whether secondary habitats are more or less productive than primary forest, they are certainly not empty of wildlife and they play an important role in wild meat supply in many areas (Daily et al. 2003; Naughton-Treves et al. 2003).

In theory, a wider diversity of targeted species could directly improve the supply of wild meat in two possible ways:

- By increasing the secondary productivity of the ecosystem, if different species exploit ecological resources in a complementary way. This effect has been predicted theoretically and demonstrated in experiments with simplified plant systems (Hooper et al. 2005) but we found no evidence that it may happen for the provision of wild meat.

- Through a portfolio effect, by providing a diversity of options to hunters. Variety does not seem to be a major factor from the consumer's perspective, with the most abundant species often being the most preferred (but see East et al. 2005). A diversity of species could also allow hunters to switch to other species as some become too scarce. In practice, species disappearance tends to follow a nested pattern, with the most sensitive only being present in species-rich systems and the most tolerant species dominating species-poor systems. As discussed below, we found no evidence that the latter are less productive than the former.

Species diversity may nonetheless affect the provision of wild meat indirectly, by influencing ecosystem composition, spatial structure, and ultimately resilience. The species that tend to be the most affected by hunting often play key roles in maintaining tropical forests, performing ecological functions (grazing, browsing, trampling, seed dispersal and excavation) that are disproportionately large relative to their total numbers (Robinson et al. 1999; Wilkie & Lee 2004). In particular, depletion and extinction of frugivorous species seems to have a significant effect on seed dispersal and consequently on the recruitment of particular plant species (Roldan & Simonetti 2001). Large-bodied primates (woolly monkeys and spider monkeys), in the Amazon are key to the recruitment of many large-seeded tree and liana species (Peres & Palacio 2007), while tapirs play a similar role for a large-seeded palm in the Amazon (Fragoso 1997). In the Congo Basin, forest elephants are keystone species, ecosystem engineers, and the bulk of mammalian biomass where they occur. The decline of these species (all of them favoured by hunters) may therefore result in long-term changes in the forest communities, in terms of both floristic composition and spatial structure (Stoner et al 2007).

The abundance and diversity of other (non-targeted) species can theoretically affect wild meat production through either bottom-up or top-down processes (Sinclair & Krebs 2002). Bottom-up effects are more straightforward in that a higher abundance of food species can improve animal productivity. Diversity is also likely to have a positive effect, increasing the stability of food provision across the year. Top-down effects are more contentious. The effect of a higher number of predators is the decline in prey abundance, including many species (such as ungulates, rodents, primates, and squirrels) that are the basis of wild meat supply. And indeed Michalski & Peres (2007) found that a decline in predators leads to an increase in wild meat supply through predator release. However, a diversity of predator species may have a positive effect, if top predators prey on other predator species (mesopredators). Theoretical studies (Palomares et al. 1995) and some empirical results (Crooks & Soulé 1999; Terborgh et al. 2001) provide some evidence for this effect.

Recreational hunting (and trophy hunting in particular) is more selective in terms of the species targeted, and so is likely to be more affected than harvesting of wild meat by the loss of particular species.

Relationship between habitat area and wild meat provision

The diversity of harvested species is predicted to increase with area (MacArthur & Wilson 1967), and indeed Michalski & Peres (2007) found a classic species-area relationship (linear with the logarithm of area) for mammals species in a fragmented landscape in the southern Amazon (Figure 16a). Peres (2001) predicted a non-linear relationship between harvested vertebrates and logarithm of forest fragment area: small fragments (<20 ha) retain practically no species (with or without hunting pressure); larger areas are required to retain the same number of species when hunting pressure is higher (e.g. 90% of the species are retained in fragments of 3000 ha in the absence of hunting, but about 11,000 ha is required under moderate to heavy hunting); areas as large as 200,000 ha may be required to maintain the most spatially-demanding species (tapirs) (Figure 16a). These studies also found that as area declines, a non-random, nested simplification of fauna takes place, with large slow-growing species, or those that live in large wide-ranging groups that occur at low density, disappearing first.

The relationship between area and species richness is not particularly informative, though, as we found no evidence that species richness of harvested species drives wild meat provision. The relevant relationship is one between area and wild meat productivity (biomass that can be harvested, a product of standing biomass and the maximum sustainable offtake rate). We did not find any study which explicitly investigated this relationship. In principle, the relationship between wild meat productivity and area can be:

a) Neutral: productivity *per unit of area* is the same irrespective of whether that unit of area belongs to a small or a large habitat fragment. This is plausible if we assume that animal productivity reflects primary productivity, and that this remains the same irrespective of fragment size. In this case, wild meat production increases linearly with area (Figure 16b, blue line).

b) Positive: larger fragments have larger productivities *per unit area*. This can be the case if the reduction in species richness for declining fragment area translates into a decline in overall wild meat productivity (for example, if it means that some ecological niches become empty, 'wasting' some of the primary productivity). In this case, productivity increases faster than area, at least until the point where area becomes large enough to maintain the complete faunal assemblage (Peres 2001 found this to be about 200,000 ha in Amazonian forests) (Figure 16b, purple line).

c) Negative: smaller fragments have larger productivities *per unit area*. This is plausible if as faunal assemblages become more simplified for smaller areas, they become dominated by species that are on average more productive (e.g., if slow-reproducing species are replaced with fast-growing species). Predator release (Terborgh et al. 2001) may also result in higher productivity in smaller habitat fragments. In this case, productivity increases slower than area, again at least until the point where area becomes large enough to maintain the complete faunal assemblage (Figure 16b, green line).

Edge effects may also affect the shape of the relationship between area and the productivity of wild meat:

d) If the matrix habitat is unfavourable (e.g. a population sink due to increased mortality) it is expected that smaller fragments (with more pronounced edge effects) will have lower productivities per unit of area, and productivity will increase faster than area while the fragment is small enough to have all of its area affected by edge effects (this will vary with the distance at which edge effect are felt; for a 1 km-wide edge effect in a circular fragment, the area would be 314 ha) (this produces the same shape as for Figure 16b, purple line, but the threshold would be a different area).

e) If the matrix habitat is favourable (e.g. by providing additional sources of nutrition, or by being a preferred habitat for a highly productive species) it is expected that smaller fragments (with more pronounced edge effects) will have higher productivities per unit of area, and again productivity will increase slower than area as long as the fragment is small enough to be subject to edge effects (Figure 16b, green line).

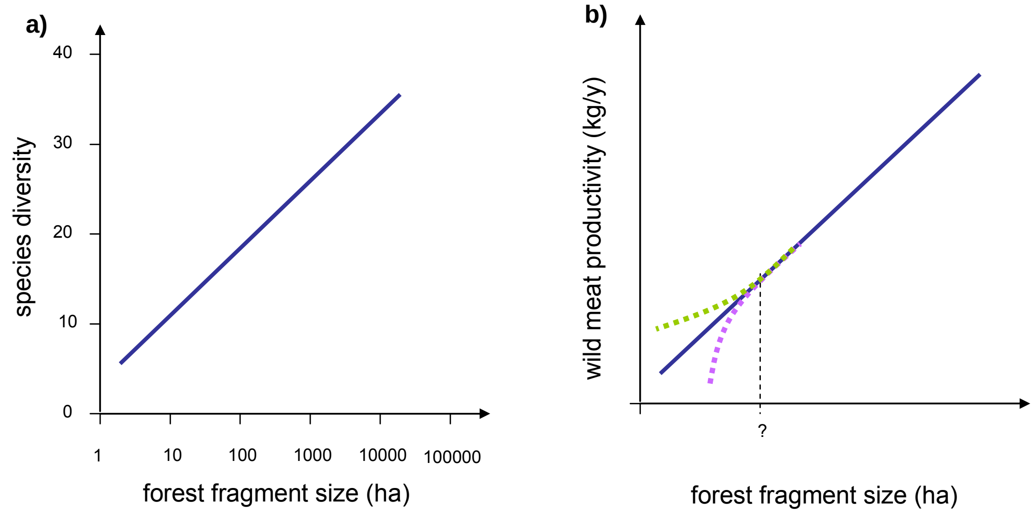

Figure 16. a) Relationship between forest fragment size (note the logarithmic scale) and the diversity of mammal species (from Michalski and Peres 2007). b) Possible shape of the relationship between wild meat productivity (in biomass per year) and forest fragment size (note the linear scale; see text for details).

Michalski & Peres (2007) found a *negative* relationship between forest-patch area and two measures of the aggregate density of mammal species, and no variation for a third measure. However, overall density (number of individuals per unit area) is not an ideal measure of wild meat supply because it does not account for variation in body size across species. Indeed, if larger species are dominant in larger habitat patches, then a negative relationship is expected between area and abundance, even if standing biomass remains constant. Michalski & Peres (2007) found that larger species (e.g., tapir, giant armadillo) only occurred in the largest fragments. On the other hand, larger-bodied species tend to reproduce more slowly, and so for the same standing biomass productivity may be higher for smaller fragments if they are dominated by small, fast-reproducing species (supporting hypothesis c, above).

Michalski & Peres (2007) also found that several species were hyperabundant in small patches. They explained this as possibly resulting from: subsidies by matrix habitats (e.g. presence of exotic fruit trees; supporting hypothesis e); the outcome of density compensation (increase in the abundance of one species as competition with other species declines; supporting hypothesis c); or the outcome of release from top predators (increase in abundance due to predator removal; supporting hypothesis c).

In summary, wild meat production likely increases linearly with area, but a threshold habitat size may exist below which the relationship is non-linear. However, the form of this non-linearity is not readily predictable at present.

4.8.4 What are the main threats to the provisioning of wild animal products?

Overexploitation is a main threat to the provision of benefits from wild animal harvesting, frequently resulting in the exhaustion of the source populations. Hunting pressure in tropical forests has risen dramatically in recent decades, concomitantly with forest loss, increasing human population, greater access for hunters and traders to remaining forests as a result of road building and forest fragmentation, the use of efficient modern hunting technologies (especially firearms and wire snares), loss of traditional hunting controls, and greatly increased commercialization of hunting (Milner-Gulland et al. 2003). Hunter access to inaccessible forests is exacerbated by commercial logging operations: these create an extensive network of roads linked to the national road system, and the trucks that travel them become conduits for a vast commercial trade in wild meat, transporting it from remote areas to towns for sale (Robinson et al 1999).

Overexploitation can lead to the local or global global extinction of particular species, even if these have become so scarce that it would be uneconomic to pursue them individually. This is because hunters pursuing other more common species will still kill scarce species encountered (e.g. Clayton et al. 1997). Even the collection of other non-timber forest products (e.g. Brazil nuts and forest vines) may exacerbate the possibility of this 'piggyback extinction' to take place, by lowering the opportunity costs of hunting (L. Parry, unpublished data).

Increased demand for wild meat may also result from a lack of availability of alternative protein sources (Rowcliffe et al. 2005). Brashares et al. (2004) found that a declines in fish supply in West Africa were associated with increased hunting in nature reserves and ultimately the sharp declines in the biomass of 41 wildlife species.

The introduction of livestock may reduce wild meat supplies in savannah systems, if the domestic species compete with the wild ones for forage (Robinson & Bennett 2004), but on the other hand it may reduce demand for wild meat (and therefore prevent overharvesting) if it provides a more readily accessible source of protein (Bennett 2002). Outbreaks of human commensal species which also prey on harvested species (for example baboons) may cause the decline of harvested species and prevent them from recovering even if hunting activities are subsequently reduced or ceased (J. Brashares, personal observation).

Increased contact between people, livestock and wildlife increases the likelihood of disease outbreaks in any of these groups (Karesh et al. 2005; Rizkalla 2007), therefore posing risks not only to wild meat provision but also to public health and to economic activities.

Social and economic disruption can affect the provision of wild meat by creating the conditions for overharvesting, for example by displacing people and promoting illegal trade. They can also affect recreational hunting, particularly by foreign hunters, by reducing expenditure and willingness to travel.

4.8.5 Are abrupt changes likely in the provision of wild animal products?

At the single-species level, overexploitation can lead to sudden transitions from large harvests to population collapses (Barnes 2002). We predict that benefits derived from a single or just a few species (e.g. a particular type of medicine or fur) will undergo sudden collapses as well. However, benefits from wild meat production, often based on harvesting a broad set of species, boom-and-bust trajectories of individual species are likely to be obscured as hunting effort extends to other species and to other regions. At the regional scale, therefore, changes in overall wild meat supply are likely to be gradual, except when faunal communities become very simplified and hence a sudden collapse may be expected if the last few species all decline abruptly. At the local scale, collapses are likely.

As described above, large-bodied species that are most sensitive to overharvesting often play key roles in maintaining their habitats (e.g. through seed dispersal), and so the disappearance of these species will result in a permanent change in ecosystem structure and composition (Fragoso 1997; Robinson 1999; Fragoso et al. 2003; Peres & Palacio 2007). This in turn may influence habitat quality for other species, thereby affecting the provision of wild meat in the long-term. It may nonetheless take decades or even centuries for the effects of this process on wild meat production to become obvious.

Overall, the key factors that are more likely to either result in or prevent a very quick depletion of wild meat supply are socioeconomic in nature, rather than biological. These include rights and access to the resource, management of the resource, availability of alternative protein sources, markets, economics of harvest, and incentives to harvest species sustainably.

Recreational hunting, and in particular international trophy hunting, is also particularly sensitive to socioeconomic changes, and is likely to collapse as a result of political instability in a region (e.g. war, civil disruption). While its reliance on a smaller set of species could in principle make this activity more prone to disintegrate, in practice it is unlikely that this will happen often in cases where the exploited species and populations are actively managed.

Overall, we predict that there is a very high probability that individual populations of harvested species (particularly those harvested for subsistence or commercially) will collapse in the near future. Local collapses in meat provision are also expected. However, there is a low probability that this will translate into a sudden collapse in the provision of overall wild meat at a regional level.

Socioeconomic changes are the most likely factors that could result in a sudden change in the supply of wild meat or in recreational hunting.

4.8.6 Can we quantify and map the global provision of this benefit/process and how it might change?

We aim to obtain a method or model for generating, from a given map of land cover, a surface of the estimated maximum sustainable production of wild meat. Given two maps (two 'states of the world') we would then be able to estimate the difference in the provision of wild meat between one state and the other. The economic value of this difference would then be calculated using information on demand and access, having into account that value (prices) are a function of both supply and demand/accessibility (wild meat produced in areas where there is no demand or which are innaccessible has an economic value of zero).

A first cut model would calculate local production values (kg/km^2/y), derived from empirical data for particular sites, and then generalised from local production estimates to larger-scale patterns, in order to create a map of potential wild meat production for the regions analysed, under current circumstances. This map would be heavily biased by the fact that production is dynamic, with different population sizes producing different levels of offtake at different hunting pressures; in order to generalise robustly about maximum sustainable yield values, a time series of population size and offtake estimates is needed from which to calculate sustainable offtake rates - of which there are very few in tropical forest areas.

We suggest following Robinson & Bennett (2004) in focusing on the extraction of large (> 1kg) mammals, as these dominate supply of wild meat as well as available data on species densities. We also suggest restricting the analysis to Central and South America, sub-Saharan Africa, and tropical Asia and Australasia, as these are the main regions where wild food is particularly important for human wellbeing.

A first approach for calculating local production rates is the widely used Robinson and Redford (1991) model (e.g. Peres 2001; Fa et al. 2002; Robinson & Bennett 2004). This predicts the maximum sustainable wild meat offtake for each species in a given habitat, that is, the percentage of standing biomass that can be harvested without resulting in population decline. Calculating this offtake requires information on:

- The carrying capacity of the habitat for each species, typically measured as the density (number of adult individuals per unit area) of the species in unexploited areas. This can also be estimated from empirical relationships between density, diet and body size.

- The maximum intrinsic rate of natural increase (r_{max}) for each species, the maximum rate of increase that a population can achieve under natural conditions without significant intraspecific competition (Robinson and Redford 1986). This is very difficult to measure directly (Milner-Gulland & Akçakaya 2001). Fast-reproducing species such as ungulates and rodents have higher intrinsic rates of natural increase than primates.

- A correction factor to account for natural rates of mortality (generally varying between 0.2 and 0.6), that is lower for longer-lived species.

Both r_{max} and carrying capacity can be roughly estimated from their allometric relationships with body size, corrected by taxonomic group and geographic location (Rowcliffe et al. 2003).

Using Robinson and Redford's (1991) model, the maximum sustainable offtake rates are typically lower for long-lived species: annual offtake rates for some rodent species exceed 50% of the standing biomass; are generally lower, but frequently above exceed 20%, for ungulates; and are very low, usually under 5% of standing biomass, for primates (Robinson & Bennett 2004).

The overall production of a given area (wild meat biomass that can be extracted per year) therefore depends on the particular set of species in the community, as their life history traits, body size and relative abundance affect the final result. For example, using this method Fa et al. (2002) calculated a production of 1,111 kg/km^2/year for the Congo Basin and of 488 kg/km^2/year for the Amazon.

There are, however, important limitations with this approach (Milner-Gulland & Akçakaya 2001), and we strongly recommend that a more robust model (or ideally, a battery of different models) is used to calculate sustainable production rates.

The model by Robinson & Redford (2001) assumes that the aim is to maintain populations at their original size, yet harvest may bring populations to any lower population size and still be sustainable provided that the population remains stable. These lower population sizes may be more desirable to maintain pest populations at low densities. An *a priori* decision is therefore needed about the ideal size of each managed population. One option is to assume that all target populations should remain above some percentage of their carrying capacity. The tradeoff is then between maximising production (at a population size of 50% of carrying capacity, under the simplest assumptions) and reducing the risk of population extirpation in the face of environmental stochasticity (at a population size greater than 50% of carrying capacity). A number of authors have suggested 75% of K as a suitable precautionary reference point (Roughgarden & Fraser 1996).

Having decided the target population size, one needs to estimate how much wild meat could be harvested sustainably. This requires understanding the rate at which populations can grow. Robinson & Redford (2001) use the maximum intrinsic rate of natural increase, but this overestimates the rate at which populations can grow unless they are at a very low levels. For higher population sizes, the growth rate tends to zero as carrying capacity is approached, due to intra-specific competition (density dependence). Population growth rates should therefore be scaled according to population size (i.e. how close a population is from carrying capacity). A first approach for a growth model incorporating density-dependence is the logistic function (Milner-Gulland & Akçakaya 2001). Assuming logistic growth, the maximum sustainable production at 50% of carrying capacity is ¼ of the intrinsic rate of increase multiplied by the carrying capacity. However this is a single-species model which would be unsuitable for complex multi-species systems such as bushmeat harvesting. There has not yet been any work done to calculate a maximum sustainable production rate for such systems.

A substantial difficulty with the application of any of these models arises in areas that are already being harvested, which is the case for the vast majority of tropical forest areas. Here, the current population densities cannot be considered to be the population size at carrying capacity. Measures of offtake need to include not only reported captures but also collateral mortality – instances in which animals are fatally wounded by hunters yet not recovered. In one game harvest study in the Amazon, modern colonists hunting with shotguns caused an estimated 9% collateral mortality (L. Parry, unpublished data), while wastage rates in snares in Africa have been estimated at up to 28% (Muchaal & Ngandjui 1999) and gun hunters in central Africa may fail to retrieve as much as 21% of the animals they kill (Rist 2007).

Finally, estimates of productivity should incorporate uncertainty in parameter estimates, and appropriate methods are becoming available for doing so (Milner-Gulland & Akçakaya 2001).

Moving from local estimates of population productivity for individual species to large-scale surfaces of overall wild meat productivity could take two possible routes. One could generalise first on a species-by-species basis, using information on habitat suitability to infer changes in species density, to produce a map of potential productivity for each species. These could then be added to produce overall productivity maps. A second route is to aggregate the productivity of all species at a given site to obtain an overall estimate of wild meat productivity at current population sizes and harvest rates, and then generalise such values across space based on bioclimatic and land cover data. We recommend the second route, as the former would require much finer, species-by-species, data that are unlikely to be available, as well as making untested assumptions about the interspecific interactions between species.

High quality empirical data would inform better models, as more explanatory variables can be integrated to explain local production. For example, the incorporation of adequate soil fertility data will improve production models since nutrient-rich environments have a greater game biomass than nutrient-poor ones; as shown by Peres (2000) for floodplain and unflooded forests in the Amazon. Additionally, livestock densities will affect the density of wildlife through competition, thus livestock distribution maps can be incorporated, as well as information derived from the Human Footprint Map on the degree of human disturbance, fragmentation and alteration (Sanderson et al. 2002).

Data limitations typically mean that only a very crude model can be generated. Existing data on local species composition and densities of individual species (e.g. Fa & Purvis 1997; Peres 2000; Robinson & Bennett 2004) are still very patchy, although improving. Studies also integrating accurate measures of offtake are even scarcer, as are studies for secondary habitats (including secondary forest and agricultural land).

Moving on from subsistence and commercial harvesting to recreational hunting, we suspect there is a wealth of data on the productivity of different habitats (e.g., forests, wetlands, mountain areas) for target species of birds and mammals hunted domestically in Europe and North America. This is expected given the degree to which hunting is generally managed in these regions. However, we do not know if such data are concentrated or dispersed. If the former, models of productivity for recreational hunting would not only be possible but also substantially better than the models for wild meat harvesting in tropical regions, as they would rely on better data and apply to much less complex communities (with fewer species). Modelling of recreational hunting of migratory species (such as ducks) would however pose specific challenges.

As for international trophy hunting, we are under the impression it mostly takes place in private land or concessions that are actively managed for the target species. Sustainable harvests are likely to be ensured by managers, who not only control harvest directly but also enhance the habitat to increase species' productivities. The economic value that can be gained from this activity is therefore determined much more by the capacity of operators to attract tourists than by biological constraints. In this sense it is likely to behave much more like international tourism (Section 4.12), and should be modelled and valued accordingly.

Adequacy of scenarios

The key information that the scenarios need to produce is maps of landcover. Other layers likely to be important include: human density (determining demand), transport routes (determining access), livestock density, and the bioclimatic variables used to model variation in productivity across space.

What can be done in Phase 2? At what cost? By whom[4]?

A first-cut, crude, pan-tropical, model is possible given the available analytical tools, and could probably be developed within one year given 24 researcher-months. Several research groups would be well positioned to do this work (see key experts below).

The quality of the model would rely crucially on the underlying data. Maximising the quality and quantity of the data that could be made available would be best done by establishing collaborations across institutions working in different areas. Our recommendation is therefore that this work builds from a consortium, brought together through two NCEAS-style workshops.

We suspect that a model of recreational (domestic) hunting is possible for Europe and North America, if data on game productivity are concentrated in particular institutions (perhaps CIC, see key resources). We recommend investing some time in pursuing this possibility as if such data repositories exist, creating a model would be both feasible and economically relevant.

4.8.7 Insights for economic valuation

Published economic valuations of wild meat harvesting are typically based on the current rates of extraction (e.g., Gram 2001; de Merode et al. 2004), yet these are known to be frequently unsustainable (e.g., Fa et al. 2002). Spatially-explicit models of the potential value of habitats for bushmeat production cannot therefore be built from studies on current values of wild meat extraction. Instead, biophysical models of sustainable production could be generated first, based on very crude models built using allometry to estimate r_{max} and carrying capacity, and the few studies that exist on population sizes in unhunted and lightly hunted areas. An estimate of the associated uncertainty could also be included.

These predictions could be compared with spatially explicit maps of current bushmeat production, drawn from broad-scale surveys (such as are being developed by the Bushmeat Crisis Task Force, www.bushmeat.org). Crude extrapolation from existing data to unsurveyed areas on the basis of human population density, vegetation type and transport routes would be possible, and these could highlight areas in which hunting is most likely to be highly unsustainable. Such extrapolations need to consider that rates of sustainable

[4] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

offtake will also be affected by access, which will increase as area falls and edge:area ration rises.

Current economic values could be obtained by overlaying data on current prices and consumption. However because supply and demand are interlinked, and dependent on availability of both bushmeat and alternatives (both for consumers and producers), as well as income and tastes, use of these same data to estimate the potential economic value of sustainable bushmeat harvest would involve heroic assumptions.

Further extrapolation to alternative scenarios might be possible, but only if an understanding of the underlying drivers of changes in bushmeat hunting pressure and value were obtained. This has not yet been done even at the regional or local level, although there are datasets available for which this could be done.

4.8.8 Some key resources

Wild meat:

- WCS Hunting and Wildlife Trade Program, led by Liz Bennett. Also at WCS are John Robinson and Kent Redford.
- Imperial College Conservation Science group, led by E.J. Milner-Gulland.
- Durrell Wildlife Conservation Trust, with John Fa as director of Conservation.
- Bushmeat Research Programme at the Zoological Society of London, led by Marcus Rowcliffe and Guy Cowlishaw.
- Justin Brashares' group at University of California Berkeley, including Karen Weinbaum.
- Carlos Peres' group at University of East Anglia, including Luke Parry.
- Washington-based Bushmeat Crisis Task Force, led by Heather Eaves.
- Robin Naidoo, Taylor Ricketts and Neil Burgess at WWF-US.

Recreational hunting:

- FACE-Europe, a European federation of national hunters' associations of the Member States of the European Union and other Council of Europe countries (representing the interests of some 7 million European hunters).
- Ducks Unlimited, a North-American NGO focused on the conservation of waterfowl and wetlands for hunting purposes.
- CIC – International Coucil for Game and Wildlife Conservation a Hungary-based non-governemental organisation focused on the management of wild-living resources.

4.8.9 *Participants*

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4.9 Fresh water provision, regulation, and purification

This section is a fully developed review, including contributions by experts in the field, some of whom subsequently reviewed the text.

4.9.1 Why is this process important for human wellbeing?

Fresh water is fundamental to the survival of every single human, for direct consumption and to maintain an adequate food supply (Pimentel et al. 2004). Although water covers 71% of the planet surface, globally freshwater is a scarce resource: only 2.5 % of water is fresh, and nearly three quarters of that is frozen, and most of the remainder is present as soil moisture or lies deep in the ground. The principal sources of fresh water that are available to society reside in lakes, rivers, wetlands, and shallow groundwater aquifers—all of which make up but a tiny fraction (tenths of 1%) of all water on Earth. This amount is regularly renewed by rainfall and snowfall and is therefore available on a sustainable basis (Vörösmarty et al. 2005; Arthurton et al. 2007). Furthermore, water is unevenly distributed across the planet's surface, with abundant supplies in regions such as the wet tropics and absolute water scarcity across the desert belts and in the rain shadow of mountains (Vörösmarty et al. 2005).

Fresh water is essential first and foremost for drinking. Each person needs only 20 to 50 liters of water free of harmful contaminants each day for drinking and personal hygiene to survive, yet there remain substantial challenges to providing this basic service to large segments of the human population. Half of the urban population in Africa, Asia, and Latin America and the Caribbean suffers from one or more diseases associated with inadequate water and sanitation (Vörösmarty et al. 2005). Globally, more than 1.1 billion people lack adequate access to clean water and 2.6 billion people – half of the developing world's population – lack access to adequate sanitation. Every year some 1.8 million children die as a result of diarrhoea and other diseases caused by unclean water and poor sanitation. The world's governments set as a global target under the Millennium Development Goals to halve by 2015 the proportion of world population without sustainable access to safe drinking water and basic sanitation (Goal 7, target 10) (UNDP 2006).

Water withdrawals (extraction from a watershed) use may be consumptive (not directly returned to the watershed; e.g. water used in agriculture which is lost through evapotranspiration) or non-comsumptive (e.g. water used in industrial cooling stations).

Household water requirements represent a very small fraction of water use. Water use is dominated by agricultural withdrawals (70% of all use and 85% of consumptive use), including livestock production, followed by industrial and only then domestic applications. For example, approximately 1000 litres of water are required to produce 1 kilogram of cereal grain, and 43,000 litres to produce 1 kilogram of beef (Pimentel et al. 2004). Fresh water is also fundamental for inland fisheries, including aquaculture (Section 4.7), for recreation (Section 4.12), for the production of hydroelectric energy and for transport (waterways). Indirectly, freshwater water flows also contribute to marine fisheries production (Section 4.6) (Table 3).

Over the last few centuries, global water use increased roughly exponentially, and substantially quicker than population growth, as a result of both population growth and economic development, and in particular the expansion of irrigated agriculture (Vörösmarty et al. 2005). Global water withdrawals now total ~3900 km^3 yr^{-1}, or ~10% of the total global renewable resource, and the consumptive use of water is estimated to be ~1800 to 2300 km^3 yr^{-1} (Foley et al. 2005). Water use efficiency is now improving, with per capita use rates dropping as of around 1980 from about 700 to 600 cubic meters per year (Vörösmarty et al. 2005). The aggregate global withdrawal continues to increase, though, with water withdrawals predicted to increase, by 2050, by 50% in developing countries, and 18% in developed countries (UNESCO-WWAP 2006).

While demand is increasing, freshwater resources are under threat by unsustainable water extraction and pollution. Particularly serious is the uncontrolled rate of water withdrawal from aquifers, which in many regions (especially arid ones) is significantly faster than the natural rate of recharge. This mining of groundwater reserves poses a serious threat to water supplies in world agricultural regions, especially for irrigation (Postel 1999). Increases in pollution of surface and groundwater resources not only pose a threat to public and environmental health but also contribute to the high costs of water treatment, thus further limiting the availability of water for use (Postel 1999).

Ecosystems play important roles in the hydrological cycle, contributing to water provision (quantity, defined as total water yield), regulation (timing, the seasonal distribution of flows) and purification (quality, including biological purity as well as sediment load) (Dudley & Stolton 2003; Bruijnzeel 2004; Brauman et al. 2007). Different benefits are affected differently by each of these aspects of water regulation (Table 3). The production of these benefits may be synergistic or competitive, with water management for one benefit possibly affecting others (e.g., improvement of water quality for drinking purposes through reforestation resulting in a decline in water quantity for hydroelectric production). And use may be extractive (water is removed from the system), in-situ (when the benefits are enjoyed without requiring water to be extracted), and indirect (when water contributes to the benefit provision only indirectly).

Table 3. Fresh water provision and regulation contributes to the production of a diversity of benefits. Water provision (quality), regulation (timing) and purification (quality) are not equally important for each of these benefits (estimates of relative importance: + somewhat important, ++ important, +++ very important).

Type of use	Benefits	Water quantity	Water timing	Water quality
Extractive use	Domestic water consumption	+	++	+++
	Crop irrigation	+++	+++	+++
	Livestock production	++	++	+++
	Industrial use	++	++	+
In-situ use	Inland fisheries (including aquaculture)	++	+++	+++
	Hydroelectric energy production	+++	+++	+
	Transport: waterways	+++	+++	+
	Recreation (sport, tourism)	++	++	+++
Indirect use	Marine fisheries	++	++	+++

4.9.2 What are the overall trends in fresh water provision, regulation, and purification?

The Millennium Ecosystem Assessment (MEA 2005) considered that there is a medium to high certainty that freshwater provisioning, regulation and purification has been degraded in the recent past, caused by unsustainable use for drinking, industry and irrigation, and that there has been a decline in water purification services. Dams are increasing our capacity to use hydroelectric energy and water regulation services have changed in a variable way, depending on ecosystem change and location.

4.9.3 How is the provision, regulation, and purification of fresh water affected by changes in wild nature?

Water provision (quantity)

In reviewing what is known about the impact of biodiversity on water quantity, we rely extensively on the exhaustive review Bruijnzeel (2004). Here we summarise key points about the influence of forest cover on regional climate (rainfall) and total water yield; in later sections we précis what is known about its effects on the timing of water yields, and on erosion and sedimentation.

The quantity of water delivered from a watershed is conventionally measured only as surface water output and reported as mean annual watershed yield. Ecosystems, however, affect the available quantity of both surface and groundwater (Brauman et al. 2007).

Vegetation, and forests in particular, significantly affect the quantity of water circulating in a watershed, although not necessarily in the way that is popularly perceived. Dealing first with precipitation, while it is commonly assumed that forests generate rainfall, this is not straightforward to test, particularly in tropical regions, because rainfall can be highly variable in space and time (Bruijnzeel 2004). The rationale for this effect is that the higher evapotranspiration and greater aerodynamic roughness of forests (compared to pasture and agricultural crops) leads to increased atmospheric humidity and moisture convergence, and thus to higher probabilities of cloud formation and rainfall generation. Early demonstrations of this effect have however been questioned, with the suggestion that observed higher rainfall in forested areas was an artifact of orographic effects (remaining forests tend to be in uplands where clouds are forming because of atmospheric cooling of rising air) or of differences in exposure of measurement instruments to wind and rain. Bruijnzeel's (2004) review concluded that there is increasing evidence that large-scale land use conversion affect cloud formation and rainfall patterns, although the effects may be small and they depend on the particular land use conversion. For example, simulation predicts large-scale Amazonian forest conversion to pastureland would result in a mere 7% reduction in annual rainfall; conversion between primary forest and secondary forest or even some crops (e.g. tea plantations) is expected to have little effect on levels of evapotranspiration; and forest conversion to rice paddies is expected to increase cloud formation and rainfall (Bruijnzeel 2004 and references therein).

It is well-established that, in coastal or montane regions subject to wind-driven fog or clouds (cloud forests), the presence of tall vegetation can significantly increase the amount of water reaching the forest floor as canopy drip (Bruijnzeel 2004), although the degree of fog interception varies (e..g, with altitude, wind exposure and leaf morphology; Cavelier & Goldstein 1989). It has been also been suggested that deforestation of upwind lowland forests can reduce cloud cover on adjacent upland cloud forests, affecting moisture interception by the canopy (Lawton et al. 2001).

In contrast to popular belief, it is well-established that the dominant impact of vegetation, and of forests in particular, is one of *reducing* the surface water output (which ultimately forms rivers), thereby reducing watersheds yield (Dudley & Stolton 2003; Bruijnzeel 2004; Brauman et al. 2007). This effect is due to evapotranspiration, whereby plants draw water from the soil into the atmosphere as part of photosynthesis. Trees generally use more water than other plants, as their roots allow them to absorb water from deeper in the soil. Also, being aerodynamically rougher than short, smooth vegetation, they increase the efficiency of gas exchanges with the atmosphere (Brauman et al. 2007). Under mature tropical rain forest, typically 80–95% of incident rainfall infiltrates into the soil, of which ca. 1000 mm per year is transpired again by the trees when soil moisture is not limiting, whereas the remainder sustains streamflow (Bruijnzeel 2004). Vegetation also intercepts rainfall, and evaporation from wet leaves may correspond to 10% of actual evapotranspiration.

Bruijnzeel (2004) reviewed the results of controlled experiments comparing water yield before and after deforestation, finding that in all cases the removal of more than 33% of forest cover resulted in significant increases in overall annual streamflow during the first 3 years, with changes in water yield roughly proportional to the fraction of biomass removed. As forest regenerates, or is replaced by forest plantations, water yields tend to decline again to pre-clearing levels, a process which happens faster in tropical regions (where vegetation grows more quickly) than in temperate areas. In contrast, conversion to cropland or pastureland may result in permanent increases in annual water yields. This reflects not only the diminished capacity of short vegetation to intercept and evaporate rainfall (given its lower aerodynamic roughness and generally smaller leaf area) but also its reduced capacity to extract water from deeper soil layers during periods of drought (because of more limited rooting depth) (Bruijnzeel 2004 and references therein). While there are no reported declines in annual streamflow totals following lowland tropical forest removal, from the opposite appears to hold for the conversion of montane cloud forests to cropland or pastureland. This is because evapotranspiration in generally low in cloud forests, while cloud forest-clearing typically results a reduction in flows generated by cloud and fog interception (Bruijnzeel 2004; Postel & Thompson 2005).

Water requirements are different for different plant species, so species composition of vegetation may affect water yields. Young and invasive plants generally have disproportionately large impacts on water quantity because vigorously growing vegetation tends to use more water than mature vegetation. In arid areas, adaptations such as dry-season senescence of native vegetation may limit its water use, whereas an introduced species that lacked these traits would consume water over longer periods during the year (Brauman et al. 2007 and references therein). In South Africa, for example, the spread of non-native eucalyptus, pine, black wattle, and other thirsty trees into the native fynbos (shrubland) watersheds of the Western Cape is having negative consequences for water supplies, as their evapotranspiration requirements greatly exceeds that of the low-stature and drought-tolerant fynbos vegetation (van Wilgen et al. 1996).

Lakes and freshwater wetlands store about 50 times as much water as rivers (Vörösmarty et al. 2005) and, if well-managed, provide a stable water supply for domestic, agricultural, and industrial use (Finlayson et al. 2005). Some wetlands provide an important role in aquifer recharge, therefore contributing to groundwater supply. However, some wetlands exist because they overlie impermeable soils or rocks and there is, therefore, little or no interaction with groundwater (Fenlayson et al. 2005). Wetlands evaporate more water than other land types (including grassland, forests, and arable land) and therefore reduce annual water yields (Bullock & Acreman 2003).

A less obvious role of biodiversity on water provisioning is through the provision of ice nucleators (atmospheric particles that serve as condensation and ice nuclei in clouds) that lead to precipitation, a substantial fraction of it are bacteria (Christner et al. 2008). Wild nature may also contribute to water quality through the provision of ideas or species (e.g. bacteria, reeds) that are used in water treatment plants (covered in Section 4.14).

Water regulation (timing)

In areas with seasonal rainfall, the distribution of streamflow throughout the year is often of greater importance than total annual water yield. This is particularly important to agricultural production (as irrigation is most important during the dry season), to hydroelectric production (as energy supply is not be possible if streamflow is insufficient), and to transportation (as waterways cannot be used if water level is too low) (Table 3). Bruijnzeel (2004) reviewed the available evidence on the effects of vegetation on the seasonal distribution of water flows, and this section draws heavily on this reference.

Forest clearance often has dramatic consequences to the soil's characteristics. Bare soil becomes exposed to intense rainfall, topsoil is often compacted by machinery or overgrazing, soil faunal activity decreases dramatically, and often part of the area is occupied by impervious surfaces such as roads and settlements (Bruijnzeel 2004, and references therein). All of these contribute to gradually reducing rainfall infiltration in cleared areas. As a result, catchment response to rainfall becomes more pronounced, with large storm runoff during the rainy season translating into lower recharging of the soil and the groundwater reserves that feed springs and maintain baseflow (deep groundwater outflow). This reduced recharge associated with deforestation typically exceeds reductions in evapotranspiration, so that overall, forest clearance leads to diminished dry season (or 'minimum') flows (Bruijnzeel 2004). Note that this effect is attributed to changes in the soil, and not directly to the forest loss per se. Indeed, if soil surface characteristics after clearing are maintained sufficiently to allow the continued infiltration of (most of) the rainfall, then the reduction in evapotranspiration associated with forest removal will result in an increase in dry season flow. Infiltration properties may be conserved through the establishment of a well-planned and maintained road system plus the careful extraction of timber in the case of logging operations, or by applying appropriate soil conservation measures after clearing for agricultural purposes (Bruijnzeel 2004 and references therein). Effects of deforestation on baseflows are expected to be more pronounced following severe surface disturbance in the case of deep soils with large storage capacity than in the case of more shallow soils which have little capacity to store water anyway (Bruijnzeel 2004).

The same conditions that reduce water infiltration (resulting in lower baseflow) result in a higher surface runoff (saturation overland flow), resulting in higher peakflows during the wet season. However, even with minimum soil disturbance, there will still be increases in peakflows after forest removal, because the associated reduction in evapotranspiration will cause the soil to be wetter and therefore more responsive to rainfall (Bruijnzeel 2004). The fraction of water that flows as surface runoff also depends on the antecedent soil moisture status, and storm characteristics: the runoff response is higher if the soils are already very wet from previous rainfall; and once the soil becomes saturated any further rainfall will increase the saturation overland flow (see section 4.13 for a discussion of the effects of forests on flooding).

Reforestation may not immediately restore the modulating effects of the original forest in enhancing low flows in the dry season and reducing peak flows in the wet season, as the soil storage and infiltration capacities lost with deforestation may take years to recover. In fact, the most immediate effect of reforestation may be a decline in water yields that is particularly felt in the dry season, as a result of a rapid increase in evapotranspiration. This is particularly likely to occur for fast-growing (frequently exotic) tree plantations. Cloud forest reforestation, on the other hand, is more likely to quickly result in enhanced low flows as a result of moisture interception by the vegetation (Bruijnzeel 2004).

Inland waters, such as lakes and wetlands, are traditionally considered to be very important for the temporal regulation of water flow, both by accumulating water during wet periods (reducing peak flow) and providing a reserve of water during dry periods that maintains base flow in adjacent rivers (Finlayson et al. 2005; Vörösmarty et al. 2005). A comprehensive review by Bullock & Acreman (2003) found support for the former effect, but evidence against the latter. Evidence that wetlands have an effect in reducing or delaying floods is particularly convincing for floodplain wetlands, but less so for headwater wetlands (e.g., bogs and river margins), for which a substantial minority of studies report wetlands are associated with increased flood peaks. In contrast, most (two-thirds) of the studies concluded that wetlands reduce flow in the dry season, backed by overwhelming evidence that evaporation from wetlands is higher than from non-wetland portions of the catchment during dry periods (Bullock & Acreman 2003).

Water purification (quality)

Water quality is a measure of the chemicals, pathogens, nutrients, salts, and sediments in surface and groundwater. The importance of water quality to domestic use, particularly to drinking supply, is obvious (Dudley & Stolton 2003). But water quality is also very important for food production (including crops, livestock and inland and marine fisheries) and for recreational use (Table 3) (Vörösmarty et al. 2005). Sediments reduce the storage capacity of reservoirs, thereby affecting water supply and hydroelectric production (Postel & Thompson 2005; Arthurton et al. 2007).

Ecosystems with intact groundcover and root systems are considered very effective at improving water quality. Vegetation, microbes, and soils remove pollutants from overland flow and from groundwater by: physically trapping water and sediments; adhering to contaminants; reducing water speed to enhance infiltration; biochemical transformation of nutrients and contaminants; absorbing water and nutrients from the root zone; stabilising eroding banks; and by diluting contaminated water (Brauman et al. 2007).

Streamside ecosystems such as riparian forests reduce nutrient movement to streams, therefore playing a key role in controlling nonpoint sources of pollution by sediments and nutrients in agricultural watersheds (Naiman & Decamps 1997). These ecosystems can trap sediments and sediment-bound pollutants in surface runoff (e.g. removing 80-90% of sediments leaving agricultural fields; Naiman & Decamps 1997) as well as absorb nutrients dissolved in the water, such as nitrogen and phosphorus (e.g. reducing local nitrate concentrations from cropland runoff by 5% to 30% per meter width; Brauman et al. 2007). These buffer ecosystems can thus potentially reduce water treatment costs for downstream users.

The macrophytes and microbes that promote denitrification and other biochemical processes that improve water quality are particularly abundant in wetlands, which are so reliable at removing suspended solids, phosphorus, and nitrogen from wastewater that they are regularly integrated into treatment plants (e.g. Sundaravadivel & Vigneswaran 2001). Wetland biota can also remove waterborne toxins and heavy metals from the water (e.g. Simpson et al. 1983). The effects of wetlands in environmental filtration are potentially very large. It has been estimated that converting less than 10% of the Mississippi Basin to wetlands and riparian forest would reduce 10% to 40% of the nitrogen currently creating the hypoxic zone in the Gulf of Mexico (Mitsch et al. 2001).

Forests and other mature ecosystems generally improve water quality in a catchment, by reducing surface erosion and increasing water infiltration and therefore soil filtration of pollutants. Surface erosion is rarely significant in areas where the soil surface is protected against the direct impact of the rain through a litter layer maintained by some sort of vegetation (Bruijnzeel 2004). Erosion rates can increase very significantly with deforestation, as soils tend to lose organic matter and become compacted and crusty, with impaired infiltration capacity (see above). Increased surface runoff increases erosion and watershed sediment yields. Fast water flow as surface runoff also means that pollutants built up in ecosystems (e.g. through decomposition, fertilizer application) are quickly transported to rivers, rather than being filtered through the soil (Brauman et al. 2007).

It seems to be the case that there is high functional redundancy in the effects of species on the provision and regulation of freshwater water, for example in the effect of plants in protecting against soil erosion. Some species or groups of species, however, may have particularly important roles. Freshwater mussels are particularly effective biofilters, filtering suspended particles (such as clay, silt, bacteria and phytoplankton and small zooplankton) and bio-deposit the particulate matter to the sediment floor, increasing water quality and clarity (McIvor 2004). Beavers are bio-engineers, creating dams that affect hydrological flow and nutrient cycling, improving downstream water quality (Naiman et al. 1986).

What is the relationship between habitat area and the impact of ecosystems on the provision, regulation and purification of fresh water?

It is likely that a minimum area is needed for ecosystems to have a significant effect on water provision and regulation. Data from the US suggest a minimum 25 m riparian buffer width to provide nutrient and pollutant removal, and a minimum of 50 m to provide detritus removal and bank stabilisation (Scherr & McNeely 2008).

In an analysis of 27 US water suppliers, treatment costs for drinking water deriving from watersheds covered at least 60% by forest were half of the cost of treating water from watersheds with 30% forest cover, and one-third of the cost of treating water from watersheds with 10% forest cover (Ernst 2004; Postel & Thompson 2005). Re-plotted as a function of water treatment costs avoided (Figure 17a) this indicates somewhat diminished returns for increasing fraction of watershed covered with forest.

Combining these observations with those minimum area effects, we speculate that the overall relationship between percentage of watershed covered by a particular beneficial ecosystem (e.g. forests, wetlands) and the regulation of water quality is such that no benefit is noticeable for small areas, followed by a quick increase that tends to flatten out for larger ecosystem coverage (Figure 17b).

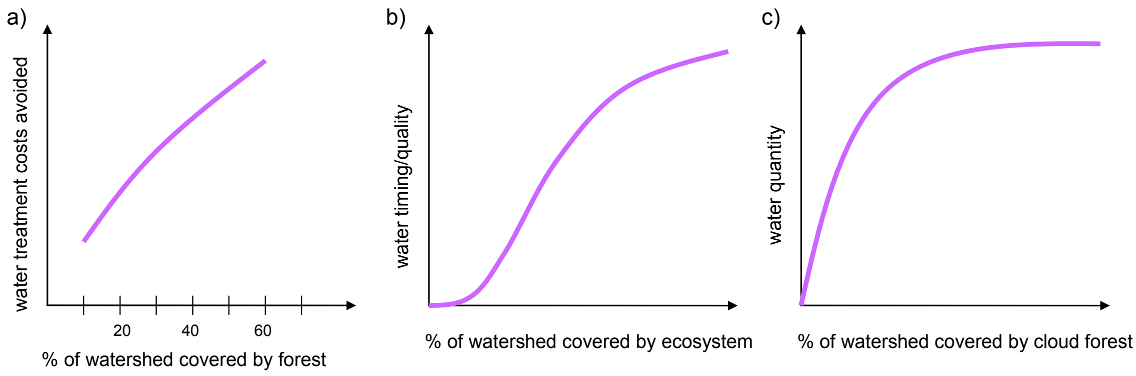

Figure 17. Relationship between area covered by an ecosystem and its impact on the provision and regulation of freshwater. a) Relationship between percentage of watershed covered by forest and avoided water treatment costs (from Postel & Thompson 2005). b) Speculated relationship between ecosystem cover in a watershed and regulation of water timing and quality. c) Speculated relationship between cloud forest cover and water quantity.

We did not find references with the shape of the relationship between regulation of water timing and ecosystem coverage, but suggest it will be similar to that for water quality, with very small or negligible effects for very low coverage and diminishing effects for high coverage (Figure 17b).

As for water provision, as mentioned above the effect for both forests and wetlands seems to be one of declining total water yields for increasing ecosystem coverage. If evapotranspiration increases linearly with area, this effect might perhaps be linear. As for cloud forests, assuming that most water interception takes place in the most exposed areas of the forest (with core areas of the forest sheltered from cloud/fog contact by surrounding forest), we again speculate that the quantity of water captured is likely to increase rapidly for a small amount of forest, and then increase with diminishing returns for larger forest coverage (Figure 17c; although note that we know of no studies that have looked at this).

4.9.4 What are the main threats to the provision, regulation and purification of fresh water?

Water provision (quantity)

Global deforestation is estimated to have reduced global vapour flows from land by 4% (3000 km^3/yr), increasing water yields, although these gain s have been mainly offset by increased vapour flow caused by irrigation (2,600km^3/yr) (Gordon et al. 2005).

No global data seem to exist on trends in cloud forest cover in particular, but these, and their water provisioning services, are threatened by deforestation (e.g. for coffee plantations and timber extraction) and by climate change (which is raising the altitude of the cloud cap) (Bubb et al. 2004).

Invasive species are in some areas compromising local water yields (e.g., Enright 2000; Zavaletta 2000).

Unsustainable exploitation (mining) and pollution of water deposits, particularly of aquifers, is reducing the capacity of ecosystems to provide what should be a renewable resource (Pimentel et al. 2004).

Water regulation (timing)

Land conversion, including widespread loss of forests (Mayaux et al. 2005) and wetlands (Zedler & Kercher 2005) and changes in soil properties due to inappropriate soil management (Bruijnzeel 2004), are affecting the capacity of ecosystems to regulate seasonal water flow, resulting in increases in peak flows and declines in dry flows in many regions (Postel & Thompson 2005).

Water purification (quality)

Again, land conversion, including loss of forests and wetlands and soil degradation, is affecting the capacity of ecosystems to filtrate and purify water. At the same time, agricultural, urban and industrial effluents are adding large quantities of pollutants aquatic systems in particular, including groundwater (Postel & Thompson 2005).

Mussels are declining globally, affecting their role as biofilters, as result of widespread changes such as damming of rivers, land use change, and pollution and invasive species (Aylward et al. 2004).

4.9.5 Are abrupt changes likely in the provision and regulation of fresh water?

Most changes to the capacity of ecosystems to regulate and provide freshwater seem to derive from, and be generally proportional to, land use change. There are however situations in which a relatively small additional change may trigger a disproportionate – and sometimes difficult to reverse – response from ecosystems' hydrological function (Gordon et al. 2008). A few examples are presented below.

Human-induced eutrophication can trigger sudden shifts in lakes and reservoirs from clear to turbid due to algal blooms (Scheffer et al. 2001). These blooms may include toxic cyanobacteria and causes major problems in water treatment works, particularly those where treatment is by direct filtration (Hitzfeld et al. 2000). They also affect freshwater fisheries and recreational use of water bodies. Reduction of nutrient concentrations is often insufficient to restore the original state, with restoration requiring substantially lower nutrient levels than those at which the regime shift occurred (Scheffer et al. 2001). Eutrophication may also trigger sudden shifts in estuarine or coastal ecosystems with the creation of dead (hypoxic) zones, affecting fisheries (Rabalais et al. 2002).

Deforestation on steep slopes may result in changes in soil properties that are not easily reversed as topsoil is removed by erosion, thereby making it difficult for new vegetation to get established. In particular, if gullies are not treated at an early stage, they may reach a point where restoration becomes difficult and expensive, as the moderating effect of vegetation on actively eroding gullies is limited and additional mechanical measures such as check dams, retaining walls and diversion ditches become necessary (Bruijnzeel 2004). Active gully erosion substantially increases catchment sediment yields, affecting water quality as well as the storage capacity of reservoirs.

Cloud forest loss may also result in a regime shift that may be largely irreversible. In some areas, such forests were established under a wetter rainfall regime, thousands of years previously. Necessary moisture is supplied through condensation of water from clouds intercepted by the canopy. If the trees are cut, this water input stops and the resulting conditions can be too dry for recovery of the forest (Folke et al. 2004).

Climate change can bring potentially trigger sudden changes, particularly in regions where ecosystems are already highly water-stressed.

Overall, we predict that there is a medium to high probability of localised regime shifts in the capacity of ecosystems to regulate and provide freshwater and the benefits we derive from it. Regional-scale shifts are more likely to be associated with water eutrophication. Global-scale shifts associated with climate change are perhaps possible.

4.9.6 Can we quantify and map the global provision, regulation, and purification of freshwater, and how it might change?

There does not seem to be a large-scale hydrological model that simultaneously integrates changes in water quantity, quality and timing. The Natural Capital Project (Nat Cap) is developing models that account for all three aspects of water provisioning and regulation. Nat Cap is using a tiered approach where the first models (currently under development) will use commonly available data and simple models to estimate the relative provisioning of water yield, baseflow (dry season flow) support, flood regulation, water quality and sediment yield. Higher tier (more advanced) models are being developed which will be quantitative and allow valuation.

Most previous modelling efforts have focused on mapping water yield, often in order to compare with water demands for evaluating shortfalls, or the impacts of climate change on water availability.

Water yield maps are usually generated from a water balance map for individual grids across the Earth's surface, taking into account water input from precipitation, output from evapotranspiration, and water flow across cells.

A number of global hydrological models exist (see Döll et al. 2003 for a review), including: WaterGap2 developed at the Universities of Kassel and Frankfurt, Germany (Alcamo et al. 2003a,b, 2005, 2007; Döll et al. 2003); Macro-PDM, developed by Nigel Arnell (Arnell 1999, 2004); the VIC model developed at the University of Washington (Liang et al. 1994; Nijssen et al. 2001); and the WBM model developed by the Water Systems Analysis Group at the University of New Hampshire (Vörösmarty et al. 1998). All of these models are driven by monthly climatic variables and they all use a spatial resolution of 0.5° (the highest resolution of available climatic input data). They vary in the way they treat variables such as soil water storage, surface runoff and groundwater recharge. They all require calibration/correction against observed water flows in order to produce a good agreement of model results, which suggests shortcomings in the models or the way they are applied. Whilst the basic hydrological concepts used are sound and have been seen to work at hillslope to river basin scale, the application at large-scales relies on gridded data that does not match well the scale of hydrological processes.

While the focus of most research has been on climate change, some of these models do incorporate land use, albeit in a very crude way (e.g. categories such as agriculture, forest, scrub, wetlands). None of them seems to have been successfully validated for its capacity to simulate the effect of landuse changes on water yields. So while theoretically one can change the landuse parameters in a model and it will respond with changes in estimated river flows (e.g. Douglas et al. 2007), it is not clear how meaningful such results would be. Adequately modelling the impacts of specific changes in land use on water provision therefore requires further development.

With these caveats in consideration, if it is considered acceptable to use such calibrated models with their remaining high error bands (especially in snow affected regions) then water yield is the aspects of water provision and regulation that can best be modelled globally.

Global mapping of water regulation (timing) does not seem to have been attempted yet, although those models which distinguish between surface and groundwater resources may be able to be adapted to look at flood mitigation/dry season flow sustainability. The standard temporal resolution of the models is monthly, but they are typically only validated for long-term annual averages, so estimated changes in monthly monthly values (e.g. Douglas et al. 2007) are likely to be associated with high error margins.

A different type of model investigates biosphere-hydrosphere interactions. These include the LPJ Dynamic Global Vegetation model, currently being developed at the Potsdam Institute for Climate Impact Research (PIK) in Germany (Gerten et al. 2004). This model estimates impacts of changes in vegetation types on the terrestrial water balance, accounting for example for the effects of changes in CO_2 on vegetation evapotranspiration and therefore water yields. It is therefore more responsive to changes in types of vegetation cover than stand-alone hydrological models. These models have focussed on natural vegetation (Gerten et al. 2004), although an agriculture mode is now under development. Global vegetation models are less effective at modelling hydrological processes, not accounting, for example, for groundwater flows (which are key to modelling the timing of water provision).

Regional models of water quality include the SWAT (Soil and Water Assessment Tool) model, developed to assist water resource managers in assessing the impact of management and climate on water supplies and non-point source pollution in watersheds and large river basins (Arnold & Fohrer 2005). The SWAT2000 includes as model components weather, hydrology, erosion/sedimentation, plant growth, nutrients, pesticides, agricultural management, stream routing, pond/reservoir routing, bacteria transport routines and urban routines. However, SWAT this relies on detailed data not available at the global scale. No global mapping/modelling of water quality seems to have been developed yet. Such models are complex in that they depend not only on water yield models but also on correctly defined hydrological pathways (e.g. rapid surface and subsurface flow, versus long term groundwater). Impacts on water quality as often very localised in time and/or space, and so the models need to be able to respond to different types of pollution sources, including point (e.g. industrial) and diffuse (e.g. agricultural) sources as well as event-based (e.g. oil spill) and long-term chronic impacts (e.g., leakage from mining fields).

In summary, global hydrological models are less developed for water quality and timing than for quantity, which is unfortunate as the former are precisely those aspects of water regulation that seem to be most closely linked to ecosystem extent and condition. A global valuation of the impacts of biodiversity and ecosystem change on water resources therefore requires additional modelling efforts.

Ideally, the different aspects of water provision and regulation should be addressed in an integrated way (rather than by having independent models for quantity, for timing and for quality) as there are likely to be synergies and tradeoffs between these components. We therefore recommend that Phase II builds from the efforts currently being developed by the Natural Capital Project to model the effects of landuse change on water services. The hydrology module of the Natural Capital Project is based at Cranfield University, where previous work also included the development of methodologies for large scale risk assessment of diffuse source contaminants (e.g. Kannan et al. 2007a, b).

What can be done in Phase II? At what cost? By whom[5]?

We recommend that investment is made in developing large-scale hydrological models that inform how different scenarios of ecosystem condition and extent (i.e. different land uses) vary in each of the aspects of water regulation and provision: quantity, time and quality.

One group that seems well positioned to address an integrated approach works at Cranfield University, in collaboration with the Natural Capital Project. It is predicted that with an investment of 24 researcher-months they would be able to produce basic global models for all three water ecosystem services within one year.

Adequacy of scenarios

The scenarios being assessed in Phase 2 need to produce:

- global maps of forest cover, distinguishing cloud from other forest types
- global maps of wetland distribution
- global maps of agricultural production (inc irrigated)
- global maps of sources of pollution, including different types of pollution sources, including point (e.g. industrial) and diffuse (e.g. agricultural) sources

[5] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

4.9.7 *Insights for economic valuation*

Assuming it is possible to predict the direction and magnitude of the impacts of land-use changes on a watershed's hydrological services, valuation of these services needs to consider that water flow is directional, and so users are typically downstream. The specific attribute of water provision (quantity, timing or quality) and its value will depend critically on the type of use (e.g. urban water supplies, irrigated agricultural production, , etc. – Table 1). Importantly, the value of a change in total water yield, in its timing, or in quality, will depend on how far these constrain benefits – for example, the value of an extra litre of water for crop irrigation will vary depending on whether irrigation is currently practiced, and if so, whether water availability (rather than, say, soil fertility) limits crop yields.

There are likely to be trade-offs within catchments, with the same change in land use (e.g. declining forest cover) increasing the provision of one hydrological service (e.g. total water yield) but reducing the provision of another (e.g. water purification). Hence, the determination of the net effect of a land-use change on the overall value of hydrological services would need to consider impacts on water quality, water timing, and water quality, and how these impacts in turn affect both downstream users and activities. To date, such comprehensive analyses have rarely been done (see Aylward 2005 for a review).

4.9.8 *Some key resources*

- The hydrology module of the Natural Capital Project is being developed by Sue White at Cranfield University and Guillermo Mendoza at Stanford University.
- The WaterGap2 (Water – Global Assessment and Prognosis) model is being developed by Joseph Alcamo at the Center for Environmental Systems Research at the University of Kassel, and Petra Döll at the University of Frankfurt, Germany.
- The WBM model is being developed by Charles Vörösmarty by the Water Systems Analysis Group at the University of New Hampshire, USA.
- The Macro-PDM model developed by Nigel Arnell, currently at the Walker Institute for Climate System Research, University of Reading, UK.
- The VIC model is being developed by Dennis Lettenmaier at the University of Washington (Liang et al. 1994; Nijssen et al. 2001).
- The Global Water System Project (GWSP - http://www.gwsp.org/) is currently running a project to compare various global water models, including those mentioned above.
- LPJ Dynamic Global Vegetation model, currently being developed at the Potsdam Institute for Climate Impact Research (PIK) in Germany.
- SWAT (Soil and Water Assessment Tool), a public domain model actively supported by the USDA Agricultural Research Service at the Grassland, Soil and Water Research Laboratory in Temple, Texas, USA.

4.9.9 *Participants*

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4.10 Wild timber, plant fibres and fuel wood

This section is a fully developed review. It greatly benefited from expert contributions (see below) but it has not been reviewed by experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.10.1 Why is the production of wild timber, plant fibres and fuel wood important for human wellbeing?

We define wild plant timber, fibres and fuel wood are those obtained from natural forests, including primary forests (with no, or no visible, indications of past or present human activity), modified natural forests (including secondary forest and selectively logged primary forest; established through natural regeneration), and semi-natural forests (established through assisted natural regeneration, planting or seeding) (FAO 2006a). This therefore excludes products obtained from forest plantations (intensively managed, regularly-spaced and/or even-aged stands, often monocultures, often of exotic species; FAO 2006a), which for the purposes of this report, are treated as crops. Timber products are generally referred to as roundwood (logs), and these can be used to produce industrial roundwood or woodfuel. Industrial roundwood can converted into woodpulp (used to produce paper and paperboard) or transformed into a diversity of wood products, including sawnwood, veneer, plywood, and reconstituted panels. Woodfuel includes both fuelwood used directly and that used to produce charcoal. Fibres include non-timber forest products such as rattan and bamboo. These forest products provide raw materials and energy, their production, processing and trade underpinning the livelihoods of millions. In this review we focus primarily on timber production as data are more reliable than for the production of fibres and fuel.

The FAO 2005 Forest Resources Assessment (FRA; FAO 2006a) summarises global statistics for the production of timber, and fuelwood. In 2005, an estimated 1,623 million m^3 of industrial roundwood were produced globally (mainly in North and Central America [38%], in Europe [31%], and in Asia [15%]), with a value of US$ 56,750 million. The FRA also reported 1,777 million m^3 of fuelwood produced (mainly in Asia [44%], Africa [31%], and South America [11%]) with a value of US$ 7,050 million. These values are likely to be underestimates, as countries usually do not report illegal removals and informal fuelwood gathering (FAO 2006a). The reported figures on fuelwood removals are particularly weak, as a large part of fuelwood-gathering is informal and not all of it comes from forests. Worldwide, estimates suggest that illegal activities may account for over a tenth of the total global timber trade, representing products worth at least $15bn a year (Brack 2007), corresponding to about 25% of the legal total in 2005 (FAO 2006a).

The 2005 FRA does not separate between production in natural forests and in plantations, but according to the 2000 FRA (FAO 2001) forest plantations were estimated to supply about 35% of global roundwood in 2000, anticipated to increase to 44% by 2020. This indicates that more than half of the total global roundwood production is still obtained from natural forests (including primary and semi-natural forests).

Tropical countries (producer countries member of the International Tropical Timber Organization, ITTO) produce about 10% of global industrial roundwood, with Brazil, Malaysia, India, Indonesia and Nigeria as the main producers and Brazil, India, Indonesia, Malaysia and China as the main consumers (ITTO 2007).

Pulpwood is one component of industrial roundwood production, and the production of pulpwood from forest ecosystems is often tightly integrated with the production of other solid wood products. Pulpwood is derived from a variety of wood sources, ranging from the harvest of fast-growing young trees in plantations managed specifically for pulp production, to the small or lower-quality stems removed from managed forests to improve forest quality or health, to the shavings, trimmings, and other wood produced in the manufacture of sawn wood products (Sampson et al. 2005). Global woodpulp production totalled 168 million tonnes in 2003, mostly in North America (79%) and Europe (48%) (FAO 2005). Pulpwood accounts for about a third of the roundwood harvested (including fuelwood). In 1995, about 17% of the wood for paper came from primary forests (mostly boreal), 54% from regenerated forest, and 29% from plantations (Sampson et al. 2005).

Bamboo is naturally distributed in the tropical and subtropical belt (between approximately 46° north and 47° south latitude; Lobovikov et al. 2007). It is now moving out of the craft-industry phase and now provides raw material for preindustrial processing and for industry products (bamboo shoots, construction poles, panelling and flooring products, pulp, charcoal, etc.), thus gaining significance as both an internationally traded commodity and a tool for livelihood and industrial development. Bamboo occupies more than 36 million ha, i.e. about 3.2% of total forest area (although not all bamboo is grown in forests), particularly in India, China and Indonesia. A substantial part of this area (in Asia, 30%) corresponds to bamboo plantations (Lobovikov et al. 2007). International trade in bamboo amounts to about US$2.5 million, with national and local trade likely to be a few times higher. Reliable statistics are still lacking, as most of the economic activities related to bamboo are not recorded officially (Lobovikov et al. 2007).

Rattan originates in the Old World, with distribution limited to tropical and subtropical Asia. It is collected almost exclusively from natural forests, with Indonesia supplying over 90% of the world's commercial rattan cane. Worldwide, over 700 million people trade in or use rattan for a variety of purposes, and the global trade (domestic and export) and subsistence value of rattan and its products is estimated at over US$7,000 million per annum. Rattan furniture manufacturing is frequently highly labour-intensive, employing well over one million people in Asia (Dransfield et al. 2002).

Production of wood and non-wood forest products is the primary commercial function of 34% of the world’s forests, while more than half of all forests are used for such production in combination with other functions, such as soil and water protection, biodiversity conservation and recreation. Yet only 3.8% of global forest cover corresponds to forest plantations (although this is increasing), indicating that a substantial fraction of natural forests are used for productive uses (FAO 2006a).

Timber, plant fibres and fuelwood are renewable products that can be harvested indefinitely if extracted in a sustainable way. In practice, harvesting rates are often unsustainable, leading to the degradation of the resource base and of the benefits derived from it. Sustainable forest management (SFM) has been proposed as a way to ensure long-term production of timber, as well as the rest of goods and services that natural forests provide, maintaining the economical, social and environmental benefits that derive from them. SFM has been proposed as a more economically advantageous land use option than logging in the humid Brazilian Amazon (Schneider et al. 2002), where it would provide a steady long term supply of timber, jobs and a constant flow of income. In Papua New Guinea, it is estimated that unsustainable management reduces the value of the forest by about US$2,300 (of Net Present Value) for every hectare logged (80% corresponding to carbon, the remaining to other environmental services; Hunt 2006). Sustainable management is particularly important in tropical forests, which harbour most of the world's terrestrial biodiversity.

Despite the economic value of forests, the ITTO estimated that (as of 2005) less than 4.5% of the permanent natural forest estate in ITTO member countries (defined as "land, whether public or private, that is to be kept under permanent forest cover to secure their optimal contribution to national development"), was sustainably managed. This area, which corresponds to only about 2% of total forested land, includes 7.1% of the area occupied by natural production forests, 2.4% of the protection forests, and 4% of plantations in ITTO countries (ITTO 2006b).

The implementation of sustainable forest management is embedded in numerous international agreements and initiatives, including: the Convention on Biological Diversity's 2010 Biodiversity Target, with forest area under sustainable management as an indicator of progress towards the Target (CBD 2006); the ITTO's Objective 2000 of achieving exports of tropical timber and timber products from sustainably managed sources (ITTO 2006a); and the United Nations Forum on Forests' Non Legally Binding Instrument on All Types of Forests, an agreement by the UN members on an international instrument for sustainable forest management (UNFF 2007).

Sustainable forest management may be interpreted in two ways: in a strict forestry sense, allowing for a continuous flow of timber and non-timber products; and in a biodiversity conservation sense, which also allows for the persistence of other forest species. Here we look at both forms of sustainability.

4.10.2 What are the overall trends in the production of wild timber, plant fibres and fuelwood?

The overall forest area has declined slightly (0.2% per year) between 1990 and 2005, including a more pronounced decrease in the area of primary forest (0.5% per year) and an increase in forest plantation area (2.38% per year) (FAO 2006a). In absolute terms, global deforestation is estimated to be 13 million ha/yr, a figure that, according to the FAO records, has remained relatively constant for the last fifteen years (an average of 13.1 from 1990-2000 and an average of 12.9 from 2000-2005) (FAO 2006a). However, other, generally higher estimates have been made, and FAO's figures (which are based on self-reporting by countries) have been criticized (e.g., Achard et al. 2002; Grainger 2008).

A 2005 FAO report reviewed data on the trends of wood and non-wood forest products between 1961 and 2003 (FAO 2005), but values are presented aggregated for all types of forest, and so include both natural forests and plantations. The overall production of roundwood (including industrial roundwood and fuelwood) has generally increased since the 1960s, but it declined in the early 1990s with the demise of the Soviet Union (Sampson et al. 2005) and has since been increasing at a slow rate (FAO 2005). A similar pattern is found for the production of industrial roundwood (FAO 2005). The production of fuelwood has been increasing at a relatively slow rate, with increases in Africa and South America, but declines in North and Central America and Europe, and a generally stable production in Asia (the largest producer) (FAO 2005).

Paper pulp production has generally been increasing since the 1960's, but at a slower rate since the early 1990's, reflecting particularly a decline in production in North and Central America (the largest producer) (FAO 2005). The share of wood production in plantations has been increasing, with the fraction of global roundwood produced there projected to increase from an estimated 35% in 2000 to 44% by 2020 (FAO 2001).

Bamboo production has boomed in response to a shift from low-end craft materials and utensils to high-tech, value-added commodities such as such as laminated panels, boards, pulp, paper, mats, prefabricated houses, cloth and bamboo shoots. China, the biggest producer, increased production nearly five-fold between 1990 and 2005 (Lobovikov et al. 2007). The rapid growth in bamboo use is bringing concern about the sustainability of global bamboo resources, but data on the actual status and dynamics of the bamboo resource base are still very patchy. Overexploitation has in some regions affected bamboo availability, stimulating the development of plantations (particularly in Asia, where 30% of bamboo is planted).

The rattan industry also expanded rapidly, particularly from the 1970s until the early 1990s, but overexploitation and wasteful resource utilization led to the depletion of the stock in some regions, particularly for the most desired species. Since the mid-1990s, the reduction in supplies of rattan caused by overexploitation and steady loss of forest habitat has posed a serious threat to the rattan industry. As with bamboo, plantations are becoming increasingly important, either in logged-over forest areas or as an agroforestry crop in rubber or other tree plantations (Dransfield et al. 2002).

4.10.3 How is the provision of wild timber, plant fibres and fuelwood affected by changes in wild nature?

The production of timber, plant fibres and fuelwood relies on the forest growing stock (volume of living trees above a given size), which in turn depends on forest area, forest condition, and forest composition. Forest area decreases with deforestation and increases with afforestation. Forest condition is reduced by degradation, for example through logging, fire, or windfelling. Highly degraded forests have lower timber yields not only because of a reduction in the growing stock but also because they are more susceptible to being infested by vines (Laurance et al. 2001) and attacked by pests (Foahom 2002). The forest composition, namely the presence and biomass of target species, is crucial because not all species have commercial value.

The question of whether forests are more productive if they have higher tree species diversity (not restricted to target species) has been addressed by a few studies, with mixed results. For example, tree species diversity was found to have a negative relationship with above-ground biomass in natural forests of Central Europe (Szwagrzyk & Gazda 2007), no relationship with productivity in Aleppo pine and Pyrenean Scots forests of Spain (Vilà et al. 2003), and a positive effect on wood production in early successional Mediterranean type forests (Vilà et al. 2007). Although species diversity might lead to higher productivity in the forest, the proportion of commercial species in more diverse sites is typically lower (FAO 2006a). On the other hand, species richness has been found to increase yield in tropical tree plantations, due to increased growth of individual trees (Potvin and Gotelli 2008), and it may reduce the impact of pests on timber species.

The genetic diversity of the target species is important for timber yields, at least in the long term, as it affects the resilience to environmental or biotic change (Buchert et al. 1997;

Jennings 2001). Furthermore, harvesting for the largest trees with the best form may result in a selection process that favours poor quality trees (e.g., with multiple stems rather than a single straight stem; Jennings et al. 2001).

Timber production can also be affected by changes in the diversity and abundance of other (non-harvested) species, particularly those who play a role as pollinators and seed dispersers (Jansen & Zuidema 2001). For example, a study of the Guianas (Guyana, Suriname and French Guiana) found that most timber species were dispersed by mammals (51%) or birds (21%), with only 20% dispersed by the wind (Hammond 1996). Indeed, obligate-outbreeding animal-dispersed genera seem to be particularly susceptible to forest fragmentation (Laurance et al. 2006). The effects of the loss of pollinators or seed dispersers may take decades or centuries to become evident, though. The loss of keystone species may potentially affect timber production: a fragmentation experiment in Venezuela found that the loss of top predators led to cascading effects as increasing populations of herbivores prompted a decline in seedlings and saplings of canopy trees (Terborgh 2001); how much these effects could impact, or are already impacting, timber species is unknown.

Soil microbes have key roles in terrestrial plant communities, affecting productivity (particularly by influencing nutrient uptake) as well as plant diversity and community composition (van der Heijden et al. 2008). About 80% of all terrestrial plant species have symbiotic associations with mycorrhizal fungi, including species that are completely dependent on the latter for growth or survival; for example, seedlings of the Amazonian timber species *Dicorynia guianensis* are unable to absorb phosphorus in the absence of mycorrhizal associations (van der Heijden et al. 2008). Land use changes (e.g. intensification, chemical contamination, logging) are known to affect the soil biota, and therefore are likely to affect timber production (van der Heijden et al. 2008).

Relationship between habitat area and the production of wild timber, plant fibres and fuelwood

Everything else being equal (e.g., rainfall, soil fertility) timber production in natural forests is expected to have a generally linear relationship with area (Figure 18). Indeed, forest area is a commonly used index in evaluating or monitoring the productive capacity of forests (e.g., Haynes 2003). It is likely that this relationship breaks down for very small areas though, with forest fragmentation affecting the continuous supply of timber. Edge effects caused by environmental (e.g. light, humidity) and biotic (e.g., species composition) contrasts between the forest patch and the matrix may result in decreased timber production because of higher mortality rates of large, high density, slow growth trees, rapid species turnover, reduction in biomass, and increase in vine density (Laurance et al. 2006). Increased tree mortality at the edges may result in forest patches that effectively shrink in size overtime (“receding edges”), particularly for smaller (<5000 ha) fragments (Gascon et al. 2000). Fragmentation also increases propensity for forest fires, which can operate as a large-scale edge effect affecting regions up to 2.4 km from the forest edges (Cochrane and Laurance 2002).

Timber supply (and value) interacts with edge and area for economic reasons related to accessibility, and these would need to be properly accounted for in the economic valuation phase (see below).

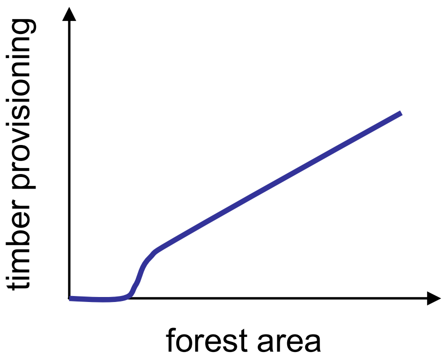

Figure 18. Predicted relationship between forest area and timber provisioning.

4.10.4 What are the main threats to the provision of wild timber, plant fibres and fuelwood?

Overexploitation of forest resources is the largest threat to the long-term provision of wild timber, plant fibres and fuelwood. Overexploitation of the targeted species can lead to long-term population depletions. For example, unsustainable logging of high-value mahogany has hampered their populations to the extent that all three species (*Swietenia* sp.) are now listed by the Convention on International Trade of Endangered Species (CITES 2008), and are listed as globally threatened in the IUCN Red List of Threatened Species (IUCN 2007). Of ~200 species in Genus *Shorea* (Dipterocarpaceae), a group of hardwood rainforest species from Southeast Asia, 139 species are listed in the IUCN Red List of Threatened Species as threatened with extinction, and one is already extinct (IUCN 2007). Both rattan and bamboo have been overexploited in some regions, resulting in population depletions and a limitation in supplies to the industries using these as raw materials (Sampson et al. 2005). Globally, there has been a decrease in the stocks of commercial species (FAO 2006a).

Harvesting may also cause a decline in the genetic diversity of commercial species, both because it reduces population size and because it potentially introduces a selective process that eliminates particularly desirable genes (Buchert et al. 1997; Jennings 2001). For example, the selective logging of the best individuals (high grading), can result in a selection against trees with straight trunks, hampering future-term productivity (Fredericksen et al. 2000).

Forest loss and degradation, including through fragmentation, are resulting in the decline of the standing stock, and therefore in the product base, for forest timber and non-timber forest products. Since 1990, approximately 6 million hectares of primary forest have been lost or modified each year (FAO 2006a) and degraded forests are becoming the predominant type of forest in many tropical timber-producing countries (ITTO 2002). Highly degraded forests have lower timber yields and are more susceptible to becoming infested by vines (Laurance et al. 2001), attacked by pests (Foahom 2002), or affected by fire (Sist et al. 2003).

4.10.5 Are abrupt changes likely in the provision of wild timber, plant fibres and fuelwood?

Given the predicted linear relationship between forest production and area (Figure 18) it is likely that a decline in the provision of wild timber, plant fibres and fuelwood takes place gradually, proportionally to the decline in the forested area. Fragmentation, however, may result in a much quicker decline in forest productivity than what would be expected given the area of remaining forest, particularly if the matrix surrounding forest patches is 'harsh' (Laurance et al. 2001).

Climate change has also been implicated in increasing forest fire risk (e.g. Westerling et al. 2006).

The combined effects of fragmentation and climate change may conspire to prompt an abrupt increase in fire risk, which may be particularly devastating (and less likely to be reversible) in tropical rain forests, as species are not ecologically adapted to fire and each fire event tends to increase the likelihood of that future ones will take place (e.g., Laurance 1998).

> Overall, we predict that the provision of wild timber, plant fibres and fuelwood will tend to decline proportionally to the decline in forest loss and degradation, which on a global scale is unlikely to be abrupt.
>
> Regionally, the effects of forest fragmentation may prompt a tipping point in which timber production becomes non-viable economically and ecologically.
>
> At larger scales, whether there is a possibility of an abrupt change in forest cover – and consequently in the production of timber and non-timber forest products – seems to depend on the combined effects of climate change and fragmentation, and how they affect the frequency and impacts of forest fires.

4.10.6 Can we quantify and map the production of wild timber, plant fibres and fuelwood, and how it might change?

State of knowledge and data availability

Here we focus on timber production. Ideally we would like to obtain a global model of potential sustainable timber production from natural forests, which could be used to quantify and map changes in timber production under different scenarios, given variation in landuse and in climate.

The ITTO (2006b) defines sustainable forest management (SFM) as "the process of managing permanent forest land to achieve one or more clearly specified objectives of management with regard to the production of a continuous flow of desired forest products and services without undue reduction in its inherent values and future productivity and without undue undesirable effects on the physical and social environment". The concept of SFM has evolved from focusing only on sustainable timber yields to including also concerns for the long-term maintenance of forest biodiversity. Criteria and indicators for sustainable forest management have been developed as part of several international processes (e.g. Montréal Process, on criteria and indicators for the conservation and sustainable management of temperate and boreal forests, [Montréal Process 2003]; Tarapoto Process on the sustainable management of Amazonian forests [Elías 2004]). Sustainable timber yields (STY) are key to SFM, but they are complex to calculate and therefore limited data are available on what the real yields are in different parts of the world. Furthermore, current STY estimates seem to be exaggerated, as timber yields are generally declining even in reportedly well-managed forests (Putz et al. *in prep*), raising the question of whether there are any real examples of sustainable forest management in tropical forests (Putz & Zuidema *pers comm*). Even if timber yields are maintained, the economic value of timber harvests may decline as the harvest diameter and/or log quality decrease, or as harvest expands into other (less valuable) species (Putz et al. *in prep*). The difficulty in finding examples of truly sustainable forest management in tropical areas is a reflection of the fact that this is seldom the most lucrative land use to private landowners, and more profit can typically be obtained by extracting all the timber that can be profitably harvested and either abandoning the area or converting it agriculture, pasture land, or to forest plantations (Putz et al. *in prep*).

A model focused on sustainable production from natural forests does not exist, but could possibly be developed from existing models, such as the Global Fibre Supply Model (Bull et al. 1998) and the European Forest Information Scenario Model (EFISCEN; Nabuurs et al. 2001). These are based on national level forest inventories, including data on forest type, area, growing stock, and net annual increment per age class. These models are capable of producing predictions under different scenarios; for example, Nabuurs et al. (2001) compared wood production under a business-as-usual scenario with one where maximum sustainable production is not exceeded. These models produce results at a coarse (country-level) scale, and they depend on the availability of good national inventories, which are not available for many countries (particularly developing countries).

Another possible approach for modelling variation in sustainable timber yields across space, and at a finer scale, would be based on generalising from known STYs across the world using data on climate, geography, geology, topology and forest cover obtained from satellite imagery (e.g. Hijmans et al. [2005] for climate data; UNEP-WCMC et al. for a map of forest cover classified according to forest types). The key limitation to this approach is the above-mentioned difficulty in obtaining accurate and reliable estimates of sustainable timber yields (STYs), particularly in tropical forests. Types of studies with relevant information for the collection of STY data include: valuation studies estimating values of sites with SFM/reduced impact logging (RIL) with reported yields (e.g., Secretariat of the Convention on Biological Diversity 2001; Merry et al. 2002), case studies (e.g., Williams et al. 1997; Iwokrama 2007; Tropenbos International 2008), studies estimating or reporting on yields of sustainable practices (Eba'a Atyi 2000; McLeish & Susanty 2000; Kammesheidt et al. 2001; Schwab et al. 2001; Glauner et al. 2003; Sist et al. 2003; van Gardingen et al. 2003, van Gardingen et al. 2006; Huth & Tietjen 2007; Ruger et al. 2007), national analyses reporting on estimated yields of sustainably managed sites (e.g., do Prado 2005), and reports from regional partnerships (e.g., Congo Basin Forest Partnership 2006).

A very crude approach to obtaining a global map of sustainable timber yields would be to collect information from studies on SFM across the world and assume average STYs for particular forest types are representative. For example, the ITTO considers 1 m^3 per hectare per year a "widely accepted estimate of tropical forest productivity" (ITTO 2006b), while Pulkki (1997) estimated that the average possible yield of tropical forests of Africa, Latin America and the Caribbean under RIL is 20 m^3/ha, and in Asia and Oceania is 40 m^3/ha, each over 40 year cycles.

What can be done in Phase II? At what cost? By whom?

We predict that it will be possible to produce within one year a first-cut global model to evaluate how sustainable timber production in natural forests is likely to be affected by changes in land cover. We recommend either building from existing forest production models based on national inventories or attempting to model variation in sustainable yield across space, based on environmental variables.

We suggest that the development of this model would require involving key institutions such as FAO, ITTO, and the European Forest Institute, and we recommend consulting with Francis Putz, Peter Zuidema and Gert-Jan Nabuurs for identifying the appropriate research group[6]. We predict that the coarser version of the model would be possible with an effort of 12 researcher-months, while the more elaborate model would require 30-40 researchers-month.

4.10.7 Insights for economic valuation

The models described above would produce information on potential sustainable timber yields, but actual timber harvesting depends on accessibility. This interacts with the area and configuration of the forest patches: on the one hand, very large patches have reduced accessibility; on the other hand, timber harvesting may not be economically viable in small fragments, particularly if different patches have different owners who may or not decide to harvest (Haynes 2003). The economic evaluation of timber production under different scenarios would need to be take accessibility into account, for example proximity to roads or rivers, and the slope of the forest area.

4.10.8 Some key resources

- Global Forest Resources Assessment FAO (2005): The most up to date global source of information on forests status and trends.

- Status of Tropical Forest Management 2005 ITTO (2006b): The latest evaluation on the state of forests of the ITTO producer countries as well as the degree to which tropical forests are being managed sustainably.

- Nabuurs et al. (2007): Use of EFISCEN to project four scenarios of wood supply for Europe based on different management regimes, including sustainable management.

[6] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

- Bull et al. (1998): Report on the general objectives, methods and results of the Global Fibre Supply Model. Also of relevance are the Global Fibre Supply Study Working Paper Series which complement the report.

- Special issue of the Canadian Journal of Forest Research (**3**:2003) on papers presented at the event "*Forest Modelling for Ecosystem Management, Forest Certification, and Sustainable Management Conference*" (Vancouver, Canada). Useful papers covering a range of issued related to forest modelling, especially those by Landsberg, Glauner et al. and Vanclay.

- Schwab et al. (2001): Comprehensive review of 266 articles on Reduced Impact Logging.

4.10.9 Participants

Authors

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Contributors

The following experts provided very valuable insights on which this section was based. We apologise to them that time constraints prevented us from circulating the text for revision.

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4.11 Wild medicinal plants

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.11.1 Why are wild medicinal plants important for human wellbeing?

Here we discuss the benefits derived from plants that are directly harvested from the wild to be used (whole, in parts, or its extracted compounds) for their real or presumed medicinal properties. Bioprospecting for pharmaceutical compounds is discussed in Section 4.14. The difference between these two sections is that wild medicinal plants are continuously harvested from natural habitats, while bioprospecting only requires a one-time use of biodiversity, as the compounds isolated can then be replicated in the lab. Animals can also be harvested for medicinal products, but we do not cover those here.

It is estimated that more than 70,000 medicinal plant species are currently used worldwide (Schippmann et al. 2006). About 3,000 medicinal and aromatic plant species are reported to be traded internationally, while a higher number are thought to be traded locally, nationally and regionally (Medicinal Plant Specialist Group, 2007). Medicinal plants can be cultivated as crops, but the great majority (70-80%) are collected from the wild (Hawkins 2008), a pattern that is expected to continue (Medicinal Plant Specialist Group, 2007). In terms of biomass, it is estimated that over 50% of medicinal plant material is sourced from cultivation (Kathe 2006) yet less than 1% of all species used as medicinal plants are used in formal cultivation for commercial production (Schippmann et al. 2006). Given their importance to human healthcare, economic subsistence, culture and livelihoods, medicinal plant species have been considered “one of the most significant ways in which humans directly reap the benefits provided by biodiversity” (Aguilar-Støen and Moe, 2007).

An estimated 60-80% of the world’s population depends on traditional medicine, including medicinal plants, to meet their primary healthcare needs (Hamilton 2004; Aguilar-Støen & Moe 2007). Those relying mostly on traditional medicine are mainly from developing countries (Aguilar-Støen & Moe 2007), particularly in Africa (where traditional medicine is used by up to 80% of the population) followed by Asia and Latin America (WHO 2002). Those with the lowest incomes benefit the most from the service provided by medicinal plants, as only 15% of pharmaceutical drugs are consumed in developing countries and the financially poor have little to virtually no access to them, especially in rural areas (Hamilton 2004). In addition, medicinal plants may be an important factor for the local economy in source countries, as they constitute a source of income for many rural households (Medicinal Plant Specialist Group 2007). In Nepal, for example, 15-30% of the total income of the poorer households is provided by selling plants to markets in Delhi (Hamilton 2004).

Although in developing countries those that rely the most on medicinal plants live in rural areas (Hamilton 2004), their use is widespread in urban areas as well (Schippmann et al. 2006. Benefits are also accrued by those living in developed countries where use of medicinal plants is rising and predicted to continue to do so (WHO 2002).

In 2006, based only on information provided by reporting countries (and hence an underestimate), FAO calculated the global removal of “raw material for medicine and aromatic products” to be 121,505 tonnes (FAO 2006a). The total estimated value of the global trade in medicinal and aromatic plants is over $60 billion/y, potentially reaching $80 billion/y (Bodeker & Burford 2007).

4.11.2 What are the overall trends in the provision of wild medicinal plants?

In recent years, there has been a marked increase in the use of complementary and alternative medicine in developed countries, leading to higher demand for these plants, a trend that is expected to continue (WHO 2002, Hamilton 2004). However, the Millennium Ecosystem Assessment indicates a decline in the provision of “biochemicals, natural medicines, pharmaceuticals” due to species extinctions and overharvesting (MEA 2005).

4.11.3 How is the provision of wild medicinal plants affected by changes in wild nature?

The benefits obtained from wild medicinal plants depend on the abundance of the target species. Given than many species are frequently used within one given area, the diversity of the target species is also expected to affect the provision of these benefits. Plants with medicinal value seem to be phylogenetically clumped in some families (Forest et al. 2007), for example Fabaceae, Asteraceae, Poaceae, Solanaceae and Euphorbiaceae (Aguilar-Støen and Moe 2007).

Habitat type and condition are also likely to affect provision of wild medicinal plants. Tropical species are expected to hold the majority of plants with pharmacologically active compounds, given their high plant species diversity. In addition, higher levels of herbivory in species-rich tropical habitats have been suggested as an explanation for the inverse relationship between the average latitude of countries and the proportion of plant species surveyed that tested positive for alkaloid compounds (Voeks 2004). Having said this, many medicinal species of different pharmacopeias have been found more frequently in secondary growth or perturbed habitats than in intact habitats. For example, a review of 18 studies that mentioned the habitat sources for medicinal plants of different tropical countries found that the majority of medicinal plants used were found in disturbance regimes (e.g., secondary forests and successional habitats; Voeks 2004). One possible explanation for this result is that these sites are more accessible and have simpler floristic composition, and so locating particular species when needed is easier (Voeks 2004). Another possible explanation is that open, disturbed habitats are more likely to induce the presence of active secondary compounds as a result of plant defences, for example against herbivory or stress (Aguilar-Støen and Moe 2007). In the former case, intact habitats may potentially have as many or even more useful plants than disturbed habitats, while in the latter case it is expected that disturbed habitats are intrinsically more valuable.

Plant production of secondary metabolites may also vary in response to environmental cues and indeed it has been found that cultivated stocks have lower levels of active ingredients compared to wild populations (Schippmann et al. 2006). The production of these metabolites may therefore be affected by the presence of other species, such as herbivores. Specific wild medicinal plants are likely to be dependent on a broad range of species besides their natural enemies - for pollination, seed dispersal, and growth. For example, *Clusia multiflora* is completely dependent on arbuscular mycorrhizal fungi for their growth and survival (van der Heijden 2008).

Relationship between habitat area and the provision of wild medicinal plants

Everything else being equal, it is expected that the abundance of a particular species of medicinal plant increases linearly with area, except perhaps for areas too small to support viable populations. Richness of medicinal plant species is expected to follow the species-area curve (linear on a log-log scale) which means that richness decays first slowly and then quickly with decreasing area.

4.11.4 What are the main threats to the provision of wild medicinal plants?

The provision of wild medicinal plants is mainly threatened by <u>overharvesting</u> (Beattie et al. 2005), <u>loss of traditional knowledge</u> (Hamilton 2004, Voeks 2004, Beattie et al. 2005), and habitat loss associated with <u>changes in land use</u> (e.g. conversion of forests to plantations, pasture and agriculture) (Beattie et al. 2005, Bodeker & Burford 2007, Medicinal Plant Specialist Group 2007). Estimates for the number of medicinal plants that are globally threatened range from 4,160 to 15,000 (Hamilton 2004, Medicinal Plant Specialist Group 2007).

4.11.5 Are abrupt changes likely in the provision of wild medicinal plants?

The abundance of wild medicinal plants is unlikely to change abruptly, but species richness may be quickly reduced as habitat destruction progresses in highly diverse regions (e.g. in the succulent Karoo of South Africa; Forest et al. 2007).

The abundance of wild medicinal plants is unlikely to change abruptly, but species richness may be quickly reduced as habitat destruction progresses in highly diverse regions (e.g. in the succulent Karoo of South Africa; Forest et al. 2007)

4.11.6 Can we quantify and map the production of wild medicinal plants, and how it might change?

Conceptually, it should be possible to measure of the value of wild medicinal plants for local people, for example by calculating the replacement costs (what would it cost to obtain similar health benefits from conventional medicine). Sustainability would need to be taken into account, with value calculated for a sustained flow of wild medicinal plants that does not result in population depletions. Also conceptually, it should be possible to generate a model that predicts such value from a mix of natural and socio-economic variables. Hamilton (2004) suggested that medicinal plants achieve their highest relative values in societies found in places richest in plant diversity. It is also likely that use of medicinal plants (and therefore its value) depends on income (with poorer people having less access to alternatives), density of human population (with use potentially increasing linearly with human population, particularly rural population), and history of human presence at the site (with older civilisations more likely to have explored the medicinal properties of the local flora) (Hamilton 2004).

This model would not be perfect (as it would only account for local use, not incorporating benefits reaped elsewhere) but would be a first approximation. However, as far as we are aware, no such model has been generated. Furthermore, we found no centralised database on the local value of medicinal plants that could be used to generate the model. We therefore predict that it would not be possible in Phase 2 to map the global production of wild medicinal plants, and how it might change with biodiversity loss and ecosystem change. However, this opinion is based on a superficial overview of the literature, and so we recommend consulting with experts on this subject for a more informed assessment (see key resources below).

What can be done in Phase II? At what cost? By whom?

We predict that current data availability would not allow for the quantification and mapping of the value of wild medicinal plants, and of how it might change with biodiversity loss and ecosystem degradation.

4.11.7 Insights for economic valuation

Accessibility to suitable habitat patches will affect the flow (and therefore the price) of wild medicinal plants. Accessiblity is higher in areas closer to roads and rivers, while in very remote regions the economic value may be zero despite a positive ecological production.

4.11.8 Some key resources

- Alan Hamilton: WWF Co-ordinator of the People and Plants Initiative has worked extensively with medicinal and aromatic plant conservation and sustainable use. Hamilton (2004) presents a comprehensive picture of the current situation of medicinal plants in terms of global harvest, trade and use.

- Medicinal Plant Specialist Group: a global partnership of global institutions (IUCN, SSC, TRAFFIC) that seek to increase awareness of the conservation threats presented by medicinal plants as well as to promote sustainable use and conservation action.

- SEPASAL - Survey of Economic Plants for Arid and Semi-Arid Lands: Database on useful wild and semi-domesticated plants of tropical and subtropical drylands, developed and maintained at the Royal Botanic Gardens, Kew.

- Economic Botany Bibliographic Database: Database maintained by the Centre for Economic Botany, currently containing citations to more than 160,000 references dealing with plants of economic value (including those of drylands). They include the ethnobotany of plant use in traditional societies, medical and industrial uses of plants, and their domestication and history. It is focused on wild plants and minor crop plants.

4.11.9 Participants

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4.12 Outdoors activities related to nature

This section is a fully developed review, led by an expert in the field, but not further reviewed by other experts.

4.12.1 Why is this benefit important for human wellbeing?

Outdoor activities related to nature (nature-based tourism and recreation) can be categorised in numerous ways. For the purposes of this review we consider two means of categorisation; outdoor recreation vs wildlife-based tourism, and local vs. longer-distance travel.

- General outdoor recreational pursuits such as walking, cycling, horse-riding and the general enjoyment of being in green spaces, the countryside and wilderness areas. Beach tourism, in as much as it relies on natural landscapes and natural processes, could also be included here. These kinds of recreation rely on natural or semi-natural landscapes but biodiversity is not the focus of the attraction. They can take place almost anywhere with scenic or wilderness value, including transformed landscapes such as the English (agricultural) landscape, managed woodlands or even urban parks and gardens.

- Wildlife-based tourism (e.g. photographic, birdwatching). This is more closely connected to biodiversity in that it has specific elements of biodiversity (i.e. selected species) as the focus of the recreational attraction. This kind of recreation usually relies on more pristine or untransformed landscapes, and is most often (but not exclusively) associated with protected areas or other designated sites or large-scale wilderness areas, including the marine environment. Note that this section does not cover recreational hunting (covered in Section 4.8), or fishing (covered in Sections 4.6 and 4.7).

- Local recreation, where people travel short distances to enjoy nature, is primarily likely to be general outdoor recreation and enjoyment of nature, as per the first bullet point above, although it will also include visits to nearby nature reserves by local residents. This segment of the market is likely to be more heavily affected by biodiversity loss or the loss of natural spaces, because such loss will reduce access and increase the cost of access for users. As biodiversity is lost locally there will be fewer or no alternative substitutable sites locally.

- Longer distance recreation, where people travel hundreds or thousands of miles to their destination, will primarily be for wildlife viewing as per the second bullet point above although beach tourism and some outdoor pursuits will also be included here. This form of recreation will be less affected by local biodiversity loss, since there will be alternative substitute sites with equivalent access or travel costs elsewhere in the world.

There are many reasons why the provision of outdoor recreational activities is beneficial. Time spent experiencing nature is enjoyable, but also benefits our mental and physical health, fosters well-being and sense of fulfilment (Bird 2005; Brown & Grant 2005), and aids education (Fjørtoft 2001). The psychological benefits of access to nature and natural environments, to a primarily urban, sedentary population, and the resultant improvements in productivity of the workforce, has been recognised since the industrial revolution:

> ‘thousands of tired, nerve-shaken, over-civilised people are beginning to find out that going to the mountains is going home; that wilderness is a necessity; and that mountain parks and reservations are useful not only as fountains of timber and irrigating rivers, but as fountains of life’ (Muir 1898).

There is a body of evidence that strongly suggests that contact with nature is beneficial to mental health and wellbeing (Henwood 2003), aiding patient recovery, reducing stress and anxiety (Ulrich et al. 1991), improving worker concentration and reducing crime and aggression (Bird 2007). Horticulture is widely used as a form of therapy in prisons and hospitals, and BCTV’s ‘green gym’ volunteers gain more benefits than just physical health improvements from manual conservation work (Brown & Grant 2005).

Beyond the psychological are demonstrable physical benefits of outdoor recreation. Many of the outdoor activities that people enjoy rely on nature to provide the rich landscapes and stimuli that underpin these experiences, ranging from gardening and walking in the park to hiking and wildlife watching. Many of these are active pursuits, and as such are extremely valuable in tackling ill-health in the face of a modern cultural environment that encourages overconsumption of food and discourages physical activity (Hill & Peters 1998).

Currently around 1.6 billion adults are overweight worldwide, with in excess of 400 million adults clinically obese, more than 115 million of whom are from developing countries (WHO). This pandemic brings with it not only serious health disorders but also social and psychological problems, all resulting in an enormous economic cost, conservatively estimated at $117 billion annually in the United States alone (Stein & Colditz 2004). However research indicates that the obesity epidemic in the United States could be halted by increasing time spent walking by only 2-3 minutes per day on average (Veerman et al. 2007). Access to green public spaces is important to urban senior citizens’ longevity (Takano et al. 2002), while accessible, large, and attractive public spaces demonstrably increase the frequency of walking, with respondents describing the importance of trees and birds amongst the features important to them (Gilles-Corti et al. 2005).

Although there is currently no evidence of additional physical health benefits from exercising outdoors per se (Henwood 2002), ‘Health Walk’ and ‘Green Gym’ outdoor exercise schemes in the UK have demonstrated that participants are more likely to continue with this type of exercise than indoor gym-based exercise routines (Brown & Grant 2005).

Finally, the tourism industry, incorporating nature-based tourism and outdoor recreation, is one of the world’s largest and fastest-growing industries, supporting millions of jobs worldwide and contributing significant proportions of the GDP of many developing and developed countries (www.worldtourism.org). Almost a century ago the scenic value of Switzerland’s natural landscapes was valued in excess of $200 million annually (Chamberlain 1910), a fact which helped to fuel the massive global expansion of national parks and protected areas and with it the birth of the modern conservation movement (Runte 1987). Nature-based tourism is one of the only non-consumptive uses of protected areas/ecosystems/biodiversity that generates tangible economic values (Ceballos-Lascurain 1996; Gossling 1999), thereby making it an important ecosystem service (MEA 2005).

4.12.2 What are the overall trends in the provision of this benefit?

With increasing ease and affordability of travel, better access and communications, and increasing affluence and leisure time, the global tourism industry is growing rapidly. Travel and tourism account for 10% of world GDP, 8% of jobs and 12% of global investment annually (WTTC 2007). Tourism also has the highest potential for growth of any industry – currently running at more than 4% per annum, with an average of 3% in developed economies and more than 7% in emerging markets" (WTTC 2007). Within this, nature-based tourism and recreation is considered to be an increasingly significant sub-sector, although there are little consistent aggregate data (Goodwin 1996; Eagles 2002). Although there have been numerous studies of the magnitude (Goodwin et al. 1998; Eagles et al. 2000; Wade et al. 2001) and value (e.g. Wells 1993; Moran 1994; Shrestha et al. 2007) of nature-based tourism at local and national scales, to date there has been no attempt at a global assessment. This is in part because there is no consistent approach to recording and reporting visitor arrivals to parks and protected areas (Eagles 2002) and other forms of outdoor recreation go largely unrecorded in any form other than periodic site-based or regional surveys.

Although recent high-profile research suggests that nature recreation is declining per capita in US and Japan (Pergams and Zaradic 2008), this trend is not mirrored in much of the rest of the world where growth in visitation to protected areas is growing at least as fast as international tourism as a whole (Balmford et al., unpublished data). Less data are available for other types of outdoor activity, though it has been estimated that each year over half the population of the UK makes over 2.5 billion visits to urban green spaces (Wooley & Rose 2004), and 87 million Americans participated in wildlife-related recreation in 2006, an increase of 13% over the decade (USFWS 2007).

As people show a universal preference for natural environments over built environments, an increasing amount of outdoor recreation would be expected with projected increases in world population and growing urbanisation (de Groot et al. 2005). This will place an increasing reliance (and therefore value) on remaining natural landscapes.

Two other trends are worth noting. The first is a shifting emphasis in global tourism towards emerging, developing world destinations, and with it an increase in domestic tourism in the developing world as urban incomes rise. The second is an increasing diversification of the agricultural sector in Europe and beyond into recreational service provision. At the same time community-based tourism in rural areas in the developing world is expanding nature-based tourism beyond protected area boundaries (Walpole & Thouless, 2005). Whether these trends are increasing the overall magnitude of nature-based recreation is unclear, but they are undeniably expanding the area and range of ecosystems which provide recreational opportunities.

4.12.3 How is the provision of this benefit/process affected by changes in wild nature?

There are almost no reliable quantitative data on the extent to which changes in biodiversity influence recreational benefits. The value derived from nature by those experiencing and using it for recreational purposes is influenced by a wide range of factors, of which biodiversity is one. Whilst it underlies the provision of key elements of a recreational experience, and of supporting services, its marginal value is difficult to determine and varies depending on the type of recreation and the perspective of the individual user. The following sub-sections review some areas where biodiversity is likely to play a role.

Provisioning of beautiful landscapes/seascapes

Outdoor recreation users rely on the provisioning of aesthetically pleasing landscapes as a stage for their experience (De Groot et al. 2005), landscapes which are created and regulated as part of naturally functioning ecosystems. Examples include the deposition of the calcium carbonate tests of foraminiferans and reef-building organisms which, together with mollusc shells, supply material for many beaches (Yamano 2000).

Studies have shown a clear preference for natural landscapes over urban ones, and people tend to prefer greener, healthy ecosystems, but more detailed judgements of landscape quality such as ‘wildness’ are likely to be more culture-dependent (de Groot et al. 2005). They are also activity-dependent. Much outdoor recreation is physical in nature and relies on the physical (slope, terrain) rather than biological (species and habitats) qualities of the landscape. Moreover landscape ‘beauty’ is in the eye of the beholder and for some can be enhanced by modification (the English countryside being a good example) which may reduce biodiversity or reduce the importance of biodiversity in the recreational experience. A study of forest recreational user preferences in the UK revealed the importance of man-made facilities for enhancing use and enjoyment amongst general and activity-based forest visitors, with only dedicated nature watchers prioritising access to biodiversity (Christie et al. 2007).

The creation of the natural landscapes that make outdoor pursuits attractive is clearly a key benefit provided by biodiversity, yet there is currently insufficient knowledge to be able to define which attributes of the landscape are key to peoples’ experiences, let alone to quantify links between specific aspects of biodiversity and outdoor benefits. Typically such benefits are linked only to very general descriptions of nature such as visual complexity or perceived naturalness (Brown & Grant 2005), which are difficult to translate into biological terms or a concrete metric of biodiversity. However, land use change from natural to intensively cultivated (or urbanised) landscapes will clearly reduce or eliminate the recreational amenity value of an area. This will impact on the provision of local outdoor recreational opportunities where access/distance is a constraining factor.

Provisioning of focal attractions: species diversity or presence of particular species

Humans derive great enjoyment and benefit from interaction with and observation of wild animals and plants, clearly demonstrated in our love of gardening, and the popularity of wildlife watching. E.O. Wilson has argued that the desire for such experiences is the product of an innate love of living things which he terms ‘biophilia’, which drives our need to seek connections with other life-forms (Wilson 1984).

Charismatic species such as gorillas and cetaceans clearly impact demand for outdoor recreation, even generating tourism industries in areas not previously seen as a destination, such as Rwanda. The annual recreational value of wildlife watching in Lake Nakuru National Park, Kenya has been estimated at between US$7-15 million, with flamingos creating more than one third of the value (Navrud & Mungatana 1994). The reintroduction of wolves to Yellowstone has attracted additional tourists to the national park, with consequent economic and social benefits estimated at between US$6-9 million per year (Donlan et al. 2006). Interestingly, the presence of the wolves alone increases the enjoyment of the park experience for tourists, regardless of whether or not they see any wolves (Montag et al. 2005).

Charismatic bird species have a significant influence in generating wildlife tourism in the UK, with an estimated 290,000 people visiting osprey nesting sites each year, bringing an extra £3.5 million to local economies, while the presence of sea eagles on Mull attracts an extra £1.4-1.6 million of spending each year (Dickie et al. 2006). In a study of nature tourists visiting Uganda Naidoo & Adamowicz (2005) found a tangible demand for bird diversity and a willingness to pay for it. Similarly, in Komodo National Park in Indonesia, visitors' willingness to pay increased entrance fees was positively correlated with the number of dragons they had seen during their visit (Walpole, 1997).

Biodiversity is perhaps even more significant in marine tourism. There are around 10 million active scuba divers who generate around $1.2 billion annually, and reef-related tourism is growing rapidly at around 20% per year (Cesar et al. 2004). For these outdoor users the diversity of species found on a coral reef is clearly paramount to the enjoyment of a diving experience. In the aftermath of the mass coral bleaching event of 1997-8 this was demonstrated by the losses to the dive industry in regions hit by coral bleaching and the associated loss of reef species, for example in Palau, where as much as 10 percent of the diving industry's producer surplus was lost (Schuttenberg 2001). A survey of tourists in Bonaire revealed that 80% would not be willing to return in the event of coral bleaching (Uyarra et al. 2005).

These examples all reveal the importance of biodiversity, in terms of species presence (which often relies on broader ecosystem health). It is clear that wildlife-based recreation is generally more sensitive to the availability of pristine environments, species abundance and diversity than broader outdoor recreation. Moreover some of the highest profile attractions, the top predators and other large animals, require intact, functioning ecosystems to persist. Yet the relationship between changes in these elements of biodiversity and the provision (and value) of recreational benefits is complex, case specific, rarely linear and sometimes counter-intuitive.

Special features of tourism as an ecosystem service

There is no linear relationship between magnitude of visitation, expenditure or other measure of value and biodiversity. A reduction in bird diversity in a tropical forest may impact on bird-watchers who are motivated by, amongst other things, species richness. Likewise a decreasing chance of seeing a rare species such as the Komodo dragon, or the disappearance of any of the 'big five' African mammal attractions might deter some from visiting a particular park. However in general there is likely to be significant elasticity of demand, with a change in number of flamingos, or wildebeest, or lions in a park having little immediate impact on visitor numbers (Walpole & Thouless 2005). In other cases increasing rarity and risk of extinction may increase tourism demand for certain sites and species, with the mountain gorilla, tiger and giant panda as obvious examples (Entwistle et al. 2000).

Tourism is still growing and is not currently limited by availability of natural attractions. Increasing availability of nature-based tourism attractions (for example due to the establishment of new protected area destinations) increases choice, substitutability and competition between destinations. It may thus alter the geographic distribution of economic benefits from tourism as new destinations capture a portion of the market. However it will not necessarily increase the overall value of tourism at global scales (Walpole & Thouless, 2005).

The value of tourism as a rationale for ecosystem conservation is limited by the size of the (albeit growing) global market, but in the short term is unlikely to be constrained by a general reduction in biodiversity and availability of recreational sites until such point as access, overcrowding or species disappearance cause a tipping point to be reached. Such tipping points may be reached at the local level, particularly in terms of access to general outdoor recreation where travel cost is likely to be a limiting factor. In theory, if local sites for outdoor recreation decline (with increasing urbanisation or agricultural expansion), there will be less opportunity for people to obtain this benefit and welfare will decline.

Overall welfare will not decline as long as substitutable opportunities are available elsewhere. This is generally more likely to be the case for longer-distance wildlife tourism than for more general outdoor recreation. Most tourists will simply choose an alternative destination. Unlike other ecosystem services, tourism benefits do not flow to the beneficiary but rather the reverse – beneficiaries travel to the point where the service is produced. For international tourism at least, there is not the same distance or access constraint as for some provisioning, regulating or cultural services, and substitutability is possible.

Provisioning of regulatory services

Tourism and recreation rely on a range of regulatory and supporting services provided in part by biodiversity, including climate regulation, water flow, waste management, coastal protection and erosion control. Without the many regulating services provided by wild ecosystems attractive environments for outdoor recreation would not be maintained. This has been seen in some coastal areas in Mauritius, where seagrass beds were removed in the belief that they were not aesthetically pleasing to tourists, resulting in a worse environment for swimmers, with increased turbidity and loss of infaunal biodiversity (Daby 2003). The role of biodiversity in these ancillary services are dealt with elsewhere in this review and thus are not considered further here.

4.12.4 What are the main threats the provision of this benefit?

With increasing global population, urbanisation and land transformation the provision of remaining natural landscapes will increase in importance, particularly for the growing proportion of humanity that does not have any exposure to natural settings. Many natural environments, such as coral reefs, wetlands and montane environments are already becoming significantly degraded and transformed. Coastal areas are also under threat from erosion, over-development and sea level rise. Intensifying conversion of wild habitat to agricultural land will drastically impact outdoor recreation for many, particularly where access to outdoor recreation becomes constrained, although there is insufficient knowledge of landscape preferences to predict the net effects on all of humanity.

At the same time the proportion of the world's surface under designated protection continues to increase (UNEP-WCMC, 2008), and natural areas continue to be added to the UNESCO World Heritage List (http://whc.unesco.org). Although such designation does not guarantee protection (and many protected areas and wildlife populations remain threatened by development pressures, poaching, logging, settlement, etc) it does raise profile and makes such areas more likely to become tourism destinations. This is in effect increasing the provision of relatively pristine natural environments upon which much nature-based tourism is based.

Globally 32% of amphibian species, 23% of mammals, and 12% of bird species are threatened with extinction, while the status of reptiles, plants and fish is uncertain. Within these groups, charismatic taxa such as primates, carnivores and albatrosses are particularly at risk. In recent years, the extinction risk for birds and amphibians has increased (Baillie et al. 2004) and the pattern is likely to be mirrored in other taxonomic groups. WWF's Living Planet Index also suggests that, overall, wild species abundance is also in decline, implying that even the less threatened wildlife may become harder to find.

4.12.5 Are abrupt changes likely in the provision of this benefit?

Abrupt changes in the provision of tourism benefits can occur for a range of reasons. Some of these may be ecological, as systems reach tipping points. Key wildlife populations may collapse through disease or other factors, fire may destroy picturesque landscapes, corals may bleach with sudden temperature shifts, systems may suddenly change from one (attractive) to another (less desirable) stable state. Some of these will be reversible, others may be more permanent.

Abrupt shifts may also (and perhaps more often) be socially instigated. War, terrorism, socio-political disruption, natural disasters and health crises all tend to rapidly and negatively affect international tourism demand, as evinced for example by Kenya, Zimbabwe, Bali, Egypt and Nepal in recent times. Likewise events such as the foot and mouth outbreak in the UK in 2001 may have dramatic impacts as people are prevented from visiting the countryside for recreation.

In some cases it has also been shown that abrupt changes in management of areas, such as degazetting parks or increasing entrance fees, can result in a sudden drop in recreational demand (Goodwin et al. 1998). The current rapid rise in oil prices (and thus aviation fuel costs) and potential carbon taxes on flying, may have similar impacts on international tourism if such changes are too sudden.

4.12.6 Can we quantify and map the global provision of this benefit and how it might change?

We are a long way from being able to map the production of benefits from nature-related outdoor recreation based on changing states of biodiversity. Firstly there is little known about the link between features of biodiversity and outdoor recreation demand on which to base such models. Indeed the analysis of outdoor recreation in general is hampered by a great scarcity of data on outdoor users, and where monitoring does exist is often unreliable and patchy.

Secondly and in such cases where the link has been demonstrated or recreational value estimated, it is not clear how far we can generalise such a relationship from very localised case studies. In contrast to the general terms such as 'perceived naturalness' which are used to describe the desired aspects of landscapes for general outdoor recreation in some literature, valuation studies focussing on the importance of particular species or ecosystems to certain types of outdoor visitor are too context specific to allow generalisation.

It is difficult to tie down universal preferences for biological attributes from such a diverse group as nature-based recreational users, so linking biophysical attributes to amenity value globally is problematic. Part of the difficulty may lie in that such preferences are context-specific, that is, they are dependent on the physical and cultural environment from which people originate. It has been found that tourists' perception of environmental quality is influenced by their past experiences, socio-economic background and culture (Petrosillo et al. 2006).

Thirdly, most studies focus on total economic valuation rather than marginal values and so the application of scenarios to model the impacts of change is constrained.

Already developed integrated models

Existing models of nature tourism demand are few and far between, and are unable to include any aspects of biodiversity amongst their variables. Brainard and colleagues (2001) have produced models predicting demand for recreation in English woodlands based on accessibility, local population, and facilities present. Likewise Jalale (1993) used conventional demand modelling to identify factors affecting visitation to national parks in Malawi and found access and facilities to be important. Balmford and colleagues (unpublished) have been the first to attempt to model visitor numbers to protected areas globally, using a database of site-specific and national data. None of these models were able to include significant variables relating to biodiversity. This may reflect both the importance of non-biotic factors (such as access), as well as the range of motivations for nature-based tourism, including various aspects of naturalness and biodiversity that cannot easily be distilled into adequate quantitative variables (Goodwin & Leader-Williams 2000).

In the UK Carver and colleagues (2002) have developed a method for mapping a subjective judgement of landscape, in this case wilderness. By allowing users to weight the importance of underlying spatial variables such as distance from nearest road, the different preferences of various groups for landscape qualities could be determined, and potentially the relative importance of those qualities produced by biodiversity could be measured. It is possible that a global map of a general subjective descriptor such as 'natural attractiveness' could be pieced together in this manner by allowing local weighting of the underlying landscape qualities. Such a map could then be combined with other recognised variables influencing tourism demand such as accessibility to create a model for general domestic outdoor recreation *potential*.

For wildlife-based tourism it will probably be necessary to create individual models for clearly distinct categories such as scuba divers and bird watchers, and to limit any model extrapolation to those areas where tourism currently exists (e.g within national parks rather than across natural ecosystems more broadly. The challenge is that likely (and relatively simplified) scenarios based on land use change outside protected areas would not trigger marginal changes in protected area tourism that could be valued, unless models included the impact of neighbouring land use change on protected area biodiversity. Even if it did, the likelihood of substitutable alternative destinations as menetioned above would limit the impact on global provision of tourism benefits.

Availability of adequate data

- Further data on the magnitude (and value) of different types of tourism in different ecosystems is a pre-requisite for any modelling exercise.

- Further work examining how nature-based recreation demand (either general outdoor recreation with a primarily local catchment or nature-based tourism with a potentially much broader regional or global catchment) responds to changes in biodiversity (or land use type at the very least) is required before any global scenario models can be created.

Adequacy of scenarios

- None currently available for nature-based tourism and recreational activities.

What can be done in Phase II? At what cost? By whom[7]?

We recommend that in the short term, activities are two-fold, both focusing on the value of specific recreational types in different habitats. First, more in-depth literature-based syntheses of well-studied activity subtypes, namely scuba-diving and bird watching, where the relationship between recreational benefits and biodiversity is more clearly defined and where valuation studies are more advanced.

Second, and of potentially greater value, modelling change in the availability of local outdoor recreation with changing land use/urbanisation and thus increasing distance to recreational sites. The importance and impact of access/distance on tourism demand is already well modelled, and numerous organisations in the UK have expertise in this area.

There is also benefit in modelling tourism demand and marginal value for terrestrial protected areas. This work has recently begun in Cambridge, and investment will be required in two areas: (1) refining the global visitor model for protected areas, and (2) extending it to include economic valuation. This will require improving the coverage and comparability of protected area visitor and finance data, besides a more targeted literature-search for valuation case studies. For this, UNEP-WCMC in partnership with Cambridge University, the International Centre for Responsible Tourism and the IUCN WCPA are well-placed to lead.

These two approaches would need a minimum of 24 researcher-months, and could in theory and in combination provide some kind of partial marginal valuation that could be included in a global scenario modelling exercise.

4.12.7 Insights for economic valuation

The valuation of recreation can be done based on actual financial flows, although distinguishing what proportion of expenditure during a tourism excursion should be attributed to natural attractions can be challenging. Where there is no direct expenditure, other evaluation methods such as travel cost can be used.

Since a propotion of the value of recreation is not captured in markets it is common in recreational valuation studies to use contingent valuation methods to estimate ‘user surplus’, the proportion of the value of the experience retained by the tourist rather than

[7] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

captured in entrance fees, for example. Such methods are problematic, but the issue of where to draw the line in defining what to include in an evaluation of economic benefits is a very important one to resolve at the outset of any global exercise.

4.12.8 Some key resources

Web resources are extremely limited since, apart from UNEP-WCMC and the University of Cambridge (and the Natural Capital Project) no one is exploring this issue at a global scale. The following is broken down into some key organisations and some web-available literature:

Organisations

- Natural Capital Project: www.naturalcapitalproject.org/
- World Database on Protected Areas: www.unep-wcmc.org/wdpa/
- World Commission on Protected Areas: www.iucn.org/wcpa
- International Centre for Responsible Tourism: www.icrtourism.org/
- World Travel Organisation: www.world-tourism.org/

Reports, publications and journal articles

- RSPB. 2007. Wellbeing through Wildlife in the EU. http://www.birdlife.org/eu/pdfs/Wellbeing_EU_final_version_2mb.pdf
- Pretty J, Griffin M, Peacock J, Hine R, Sellens M and South N. 2005. *A countryside for health and wellbeing: The physical and mental health benefits of green exercise*. University of Essex, Colchester. Available online at: http://www.countrysiderecreation.org.uk/pdf/CRN%20exec%20summary.pdf http://195.92.230.85/Images/Hine_tcm2-30031.pdf
- Bird W. 2004. Natural Fit: Can Green Space and Biodiversity Increase Levels of Physical Activity? RSPB. The RSPB. Available at: http://www.rspb.org.uk/Images/natural_fit_full_version_tcm9-133055.pdf
- Collins, S. 2006. The Makuleke model for good governance and fair benefit sharing Steve Collins in IUCN. Policy Matters http://www.iucn.org/themes/ceesp/Publications/newsletter/PM14-Section%20IV.pdf
- Health Council of the Netherlands. 2004. The influence of nature on social, physical and psychological wellbeing. Part 1: review of current knowledge. Report to the Minister of Agriculture, Nature and Food. http://www.rmno.nl/files_content/Nature%20and%20Health.pdf
- UNEP / CMS Secretariat (2006). Wildlife watching and tourism. Bonn. Available at: http://www.cms.int/publications/pdf/wildlifewatching_text.pdf

4.12.9 Participants in this sub-review

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4.13 Regulation of natural hazards

This section is a fully developed review, including contributions by experts in the field, most of whom subsequently reviewed the text.

4.13.1 Why is this process important for human wellbeing?

'Natural hazards' are defined here as infrequent natural phenomena that – during a relatively short period of time – pose a high level of threat to life, health or property. These include seismic events (volcanic eruptions, earthquakes, tsunamis), extreme weather events (hurricanes, floods), avalanches and land slides. This definition excludes what has been termed 'slow hazards' (Cochard et al. 2008), such as gradual beach erosion or sea level rise (although these may increase the intensity and/or frequency of other hazards).

Natural hazards have the *potential* to affect life, health or property, but generally will only do so if they occur in vulnerable areas (areas occupied by humans or property). We define 'natural disasters' as significant personal injury or property destruction caused by natural hazards. A natural disaster is therefore the consequence of natural hazards which take place in vulnerable areas (Blaikie et al. 2004).

Here we focus on the influence of ecosystems in preventing or mitigating the effects of natural hazards. That is, we focus on the biophysical aspects of energy attenuation by biological systems. A proper valuation of the utility of ecosystems for hazard mitigation needs to integrate information on the biophysical aspects discussed here with data on vulnerability (distribution of people and infrastructure).

Note that the influence of ecosystems in regulating water flow is also dealt with in Section 4.9, but there it covers the regulation of seasonal flow. This section refers to the influence of ecosystems in preventing or mitigating the effects of more extreme events.

Natural hazards affect millions of people and cause millions of dollars in property damage every year. In 2000, for example, floods affected 3.5 million people in Cambodia (with associated costs of US$145 million), 5 million in Viet Nam (US$285 million), 5 million in Bangladesh and 30 million in India (FAO & CIFOR 2005). The 2004 Indian Ocean tsunami left more than 270,000 people dead and caused billions of dollars in damage (ISDR 2006). In 2005, Hurricane Katrina hit the gulf coast of the United States, killing an estimated 1,200 people, displacing tens of thousands, and causing damages in excess of $200 billion (Dolfman et al. 2007). Natural hazards can also affect human health indirectly, for example floods can promote waterborne infections and vector-borne diseases, and cause adverse mental and physiological effects that can last months or years (Guenni et al. 2005).

Living organisms can form and create natural barriers or buffers, such as forests (including mangroves), coral reefs, seagrasses, wetlands, and dunes. These have been suggested as valuable in mitigating the effects of some natural hazards such as coastal storms (Wells et al. 2006), hurricanes (Costanza et al. 2006), catchment-borne floods (Bradshaw et al. 2007), tsunamis (Kathiresan & Rajendran 2005), avalanches (Gruber & Bartelt 2007), wild fires (Guenni et al. 2005) and landslides (Sidle et al. 2006). The available evidence for some of these effects is still scarce, and in some cases controversial (see below).

Here we focus on catchment-borne floods, sea-borne hazards (including hurricanes and tsunamis, that also cause floods), and landslides. We chose these because the potential value of ecosystems for mitigating these hazards has received considerable attention in the literature.

4.13.2 What are the overall trends in regulation of natural hazards by ecosystems?

The Millennium Ecosystem Assessment (MEA 2005) considered that there is a high to medium certainty that natural hazard regulation services are declining, due to loss of natural buffers such as wetlands and mangroves. Indeed, all of the ecosystems mentioned above as potentially valuable in providing barriers or buffers mitigating the effects of natural hazards are known to be in decline, either globally or in large parts of the world (see Section 4.13.4 below). For example: Twenty percent, or 3.6 million ha, have been lost from the 18.8 million ha of mangrove forests covering the planet in 1980 (FAO 2007b); 20% of coral reefs have been seriously degraded in only the past two decades (Wilkinson 2006); coastal wetland loss is extremely rapid, reaching 20% annually in some areas (Agardi et al. 2005).

On the other hand, the value of the regulation that is provided by these ecosystems is likely to be escalating, given an increase in human vulnerability to natural hazards. Indeed, not only is human population overall increasing, but people are settling preferentially in flood plains and coastal regions, and concentrating in urban areas where water infiltration and drainage are reduced. Furthermore, excessive water withdrawal is leading to local land subsidence in some areas (e.g. Bangkok is sinking on average 2 cm per year; FAO & CIFOR 2005). Additionally, the frequency of some natural hazards is predicted to increase with climate change (Stern 2007; see Section 4.16); for example, increased sea surface temperature is predicted to increase hurricane activity (Saunders & Lea 2008).

4.13.3 How is the regulation of natural hazards affected by changes in wild nature?

Catchment-borne floods

Floods are the rising of water bodies and their overflowing onto normally dry land (Bradshaw et al. 2007). We refer to ‘catchment-borne floods’ as those that result from extremely high levels of precipitation or rapid snow melt within a catchment. Floods may also happen in coastal areas through sea water inland incursion (see sea-borne hazards). A coincidence between heavy rainfall and strong coastal winds (e.g. during hurricanes) may result in major flooding events, particularly in estuarine regions.

It seems undisputed that geomorphology and rainfall patterns are the main factors affecting the likelihood of catchment-borne flooding in a given region (Bruijnzeel 2004). The question being posed here is whether land use has any noticeable *marginal* effect on flood magnitude.

<u>Upland forests</u> have long been considered to play a key role in the prevention of downstream flooding, and watershed management has often included forest management as a means of regulating flooding (FAO & CIFOR 2005). The proposed mechanism is that forests increase water interception and evaporation from the tree canopy, and further reduce runoff by increasing soil infiltration and storage (a ‘sponge effect’; FAO & CIFOR 2005; Bradshaw et al. 2007).

Evidence indicates that the key factor linking land use and flood regulation is soil condition, rather than the trees, and that much of the soil degradation associated with deforestation results from poor land use practices (e.g., soil compaction during road building, overgrazing, litter removal, destruction of the organic matter, clean weeding; Bruijnzeel 2004; FAO & CIFOR 2005). While at least some of these effects are potentially avoidable through reduced impact logging (Chappell et al. 2006), in practice they do tend to come hand-in-hand with forest loss, and as such it is still pertinent to ask what the impacts of deforestation are on flood regimes.

A recently published global model (Bradshaw et al. 2007) provides the strongest available evidence linking natural forest cover to frequency and severity of flooding, at country level, across the developing world (after controlling for rainfall, slope and degraded landscape area). Their models suggest that a loss of 10% of natural forest cover would result in increases in flood frequency ranging from 4-28% across the countries included. Brujinzeel et al. (2007) have argued that results may be explained by deforestation acting as a proxy for post-forest land use, rather than by forest loss proper.

Bradshaw et al. (2007) also found that while natural forest cover had a negative association with flooding risk, non-natural forest was positively associated. This may perhaps be explained by differences in forest characteristics (e.g. canopy structure) or by differences in the soil (with previously logged forest having recently suffered from soil compaction and erosion).

Considering that Bradshaw et al. (2007) excluded extreme rainfall events such as cyclones and typhoons (Laurance et al. 2007), there does not seem to be any clear published evidence that forests and their soil have a noticeable effect in regulating floods for the extreme rainfall conditions that lead to major flooding events, when the soil becomes saturated and loses its ability to store further water (FAO & CIFOR 2005). However, subsequent analyses by Bradshaw and colleagues (currently in review) indicate that the inclusion of extreme flood events predicts a nearly identical relationship between forest cover and flood severity and frequency.

FAO & CIFOR (2005) quote a literature review (Kiersch 2001) that found that the effects of landcover on floods are only noticeable for small basins (area <50,000 hectares), the interpretation being that for larger areas the effects of flooding tend to be averaged out across the different sub-basins as storms pass over (FAO & CIFOR 2005). Bradshaw et al. (2007) used countries, rather than basis, as a unit of analyses, so their results are not comparable in this aspect.

As for riparian forests, they seem to have an effect in increasing local flooding by increasing water flow resistance, particularly if the vegetation is non-flexible (Darby 1999). It is possible that this effect is not significant for large flooding events (Darby 1999). Presumably, an upstream delay may have downstream benefits, by increasing upstream infiltration and reducing the water speed, particularly in areas prone to flash floods.

Inland (freshwater) systems, such as wetlands (including peat swamp forests; Wösten et al. 2006) and lakes, are considered to have an important flood attenuation effect through energy dissipation of runoff peaks, and by storing excess water (Guenni et al. 2005). Gosselink et al. (1981, quoted by Guenni et al. 2005) found that the storage capacity of forested riparian wetlands adjacent to the Mississippi (USA) declined from 60 days of river discharge during pre-settlement times to less than 12 days discharge (an 80% reduction of flood storage capacity) and that this contributed to the severity and damage of the 1993 flood in the Mississippi Basin. Although conceptually it must be possible to quantify the marginal effects of freshwater wetlands and lakes on flooding in a similar way to the approach employed by Bradshaw et al. (2007) we did not find any such study.

Landslides

Landslides are naturally occurring phenomena in steep terrain, and can pose significant hazards to humans and property. The majority of fatalities and the highest relative costs occur in tectonically-active monsoonal and tropical cyclone affected areas of Asia and the Americas, while absolute economic losses are highest in mountainous developed countries with high levels of rainfall and/or seismicity, notably Canada, the United States, Japan and Italy (Petley et al. 2007). Landslide frequency seems to be increasing, and it has been suggested that land-use change, particularly deforestation, is one of the causes.

There are two main types of landslides: shallow, rapid landslides are episodic processes triggered by individual rainfall events or artificial inputs of water; slower, deep-seated landslides initiate or activate after a longer-term accumulation of water. The major risk to humans occurs with shallow, rapid landslides and the occasional deep, rapid landslides that impact areas where humans settle; conversely, slower, deep-seated landslides rarely cause loss of life, but can inflict extensive property and environmental damage (Sidle et al. 2006).

In steep terrain, <u>forests</u> protect against landslides by modifying soil moisture regime (Sidle et al. 2006). Indeed, tree evapotranspiration reduces groundwater levels, limiting the period of shallow landslide susceptibility as well as the period of deep-seated landslide activity. Deep-rooted vegetation dries soils at greater depths than shallow-rooted species. Tree roots and soil faunal activity contribute to macropore formation and therefore to soil drainage. Additionally, forests contribute to soil shear strength by providing root cohesion to the soil mantle (Sidle et al. 2006). Shallow soils are much more influenced by rooting strength than deeper soil mantles. In shallow soils, roots may penetrate the entire soil mantle, providing vertical anchors into more stable substrate. Dense lateral root systems in the upper soil horizons form a membrane that stabilizes the soil. This membrane is much more significant in protecting against shallow landslides than deep-seated landslides. Tree roots may lend some stability to deeper soils by lateral reinforcement across planes of weakness; however, this beneficial effect would diminish with larger and deeper potential failure sites. Forest loss renders slopes increasingly sensitive to landslide triggers, and increases the mobility (i.e., the run-out velocity and hence distance) of slides once they have been initiated (Petley et al. 2007). It may take up to two decades for secondary regrowth forests to regain comparable levels of root structure (Sidle et al. 2006).

Forests are more effective at preventing shallow (<1m) landslides, while deep-seated (>3m) landslides are not noticeably influenced by the presence or absence of a well-developed forest cover (Bruijnzeel 2004; FAO & CIFOR 2005).

Sea-borne hazards

The most common sea-borne hazards are wind-generated storm surges. In tropical regions (mostly in the belts between 5° and 25° north and south of the equator), these are typically associated with cyclone, hurricane or typhoon storms (here, generally referred to as cyclones) with the effect of wind combining with low atmospheric pressures, and sometimes with high tides, to generate the storm surge. An average of 240 hurricanes in categories 1-5 are recorded every year, about one third of which are very intense (categories 4-5; Webster et al. 2005). Storm surges may happen in some temperate regions as well, such as the Atlantic coast of North America and Argentina and Uruguay; in Argentina, for example, persistent wind caused by 3 to 5 day storms may induce sea level rise up to 4 m, causing extensive floods in coastal areas (Perillo 1997). Tsunamis (generated by earthquakes or submarine landslides) are significantly less common, with an average of a major tsunami per decade, usually in the Pacific Ocean (NOAA 2008). Lander and colleagues (2003) list 157 tsunamis that occurred globally between 1983 and 2001; of these, 30 caused damage and 16 caused fatalities; all but nineteen events were in the Pacific region (including Indonesia).

Wind-generated waves and tsunami waves have quite different characteristics (from Yeh et al. 1994; NERC et al. 2000; Cochard et al. 2008): in general, tsunami waves are faster, higher and contain much more energy than waves associated with cyclone storm surges. Wind-generated waves contain most of their energy near the surface and are only a few dozen meters long, while in tsunamis the wave energy is distributed throughout the entire water column and wave length can reach hundreds of kilometres. As a tsunami approaches land, it takes on the characteristics of a violent, onrushing tide (rather than a typical cresting wave) and the wave increases in height as it approaches the shore. Although storm surges normally hit the coast at lower speeds than tsunamis, the continued driving force of wind may cause them to penetrate deeply inland; large storm surges persisting for several hours during a cyclone may cause greater physical destruction to infrastructure and ecosystems than a few high-energy tsunami waves. On the other hand, tsunamis rarely arrive as a single wave (rather, they typically occur in series known as 'wave trains') and this can have a much greater effect than that predicted on the basis of each wave arriving alone, because the first wave in the train will clear much vegetation and enable following waves to penetrate further than predicted on the basis of the wave height at the coast and the pre-existing vegetation, and because the second and subsequent waves are loaded with debris (Synolakis & Kong 2006).

Given the significant differences between wind-generated waves and tsunamis, it is natural that ecosystems vary in the degree to which they may or not act as barriers in each case.

<u>Coral reefs</u> represent a first important 'line of defence' against open ocean wind-generated waves; by various interacting wave transformation and dissipation processes such as shoaling, refraction, diffraction, bed friction, and energy dissipation through turbulence during the breaking process, reefs may absorb over 90% of the normal wind-driven surface wave energy (see Cochard et al. 2008 and references therein). Reefs also have a significant effect in reducing the impact of storm waves. For example, during tropical cyclone Aivu in North Queensland in 1989, wave heights of 10 m were reduced to about 6 m after passage over coral reefs (Young and Hardy 1993). The effects of reefs on wind-generated waves depend on aspects such as: reef morphology (determined by species composition), reef structure (elevation, reef slope and reef flat width), relative water depth at reef edge; reef natural continuity and condition (dissipation is lower for fragmented and/or degraded reefs); distance between the reef and the shore; tidal height (wave dissipation higher in the low tide) (Cochard et al. 2008 and references therein). Coral mortality results in the gradual loss of the three-dimensional structure of the reef as the dead coral is eroded by wave action, which both reduces it effectiveness as a barrier (by increasing the depth of water above the reef flat) and diminishes its wave dissipation properties (by reducing roughness; Sheppard et al. 2005).

The effects of coral reefs on tsunami waves are much less straightforward. The first studies after the 2004 Indian Ocean tsunami suggested that fringing around small island reefs had a protective role (UNEP 2005), subsequently supported by model simulations (Kunkel et al. 2006). However, this may be due to refraction of the tsunami wave around small islands rather than by the reefs (Yeh et al. 1994). The only multifactorial study of the tsunami impact to incorporate bathymetry, geomorphology, distance from source, and coastal ecosystems, found that areas covered by coral reefs were *more* affected by the tsunami (Chatenoux & Peduzzi 2005, 2007). It has been suggested that this could be explained by channelling of the wave flow through gaps in the reef, although this is not supported by model simulations (Kunkel at al. 2006). Other numerical simulation studies (Lynett 2007) concluded that an obstacle, such as a reef, will always reduce run-up and maximum overland velocity – but that when the obstacle is small relative to the wave length of the tsunami, the reduction will be “practically inconsequential”. The fact that overall coral reefs suffered relatively little direct wave damage from the Indian Ocean tsunami (Wilkinson et al. 2006), even in the areas where it was most ferocious (Baird et al. 2005), suggest that coral reeds had little interaction with (and therefore caused little attenuation of) the tsunami wave.

While seagrasses can occur in locations exposed to fairly high wave energies, extensive seagrass beds are predominantly found in rather sheltered locations of intermediate to calm waters, such as on extensive intertidal and permanently submerged sand and mud flats in bays and estuaries, or in lagoons behind coral reefs and sand bars. Many seagrass beds, therefore, only act as secondary wave buffers, keeping wind-driven waves at reduced heights (at least at about the local water depth) behind reefs and other ‘frontline’ buffer features. Very broad, dense seagrass meadows can substantially contribute to the dissipation of wave energy on shallow tidal flats. The effectiveness of wave attenuation depends on the water depth and the length, density and flexibility of seagrass blades (Cochard et al. 2008 and references therein).

Seagrass beds were found to be negatively associated with the magnitude of tsunami impacts in the study by Chatenoux & Peduzzi (2005, 2007), but the authors note that the shallow, gently sloping coastal bathymetry associated with this ecosystem could be a confounding factor. Like mangrove forests (see below), seagrass areas occur chiefly in bays where bathymetry is likely to produce a high tsunami wave flow depth, and so they may function as ‘offshore land’, diminishing some of the wave energy before the waves wash over the actual shoreline (Cochard et al. 2008). In this case, however, any protective effect is attributable to bathymetry, and not to the presence of the seagrass ecosystem.

Mangrove forests are composed of salt-tolerant species and occur ubiquitously as a relatively narrow fringe between land and sea, between latitudes 25°N and 30°S (Valiela et al. 2001). They thrive in areas of relatively calm waters such as in estuarine environments, on accreting shores with fore-lying mudbanks, in bays and in lagoons, and areas protected by sand bars, islands, coral reefs and/or seagrass beds (Cochard et al. 2008 and references therein). It is generally assumed that mangroves have a protective role against cyclone storm winds and associated wind-generated wave impacts (e.g. Badola & Hussain 2005). Indeed, mangrove stands have been shown to reduce wave energy through the increased drag caused by their complex root (pneumatophores) systems and dense branch and leaf cover (Mazda 1999). Not all mangroves are the same though, and the degree of wave attenuation depends upon the density, size, and species composition (Mazda 1999). The value of mangroves as a protection buffer against very large cyclonic storm surges is not known exactly, but likely to be more significant for large stands of old-growth mangroves (Cochard et al. 2008).

The value of mangroves for protection from tsunamis has been hotly debated. Many initial observations suggested that mangroves dissipated much of the energy of the Indian Ocean 2004 tsunami (e.g. EJF 2006; UNEP 2005). These impressions were supported by several published studies (Dahdouh-Guebas et al. 2005; Danielsen et al. 2005; Kathiresan & Rajendran 2005; Iverson & Prasad 2007) which were extensively quoted as evidence for the value of tsunami 'greenbelts'. However, the adequacy of the analyses in each of these studies has subsequently been questioned (e.g. Kerr et al. 2006, Kerr & Baird 2007, Baird & Kerr 2007, 2008) and it has been suggested that the effect of tsunami greenbelts is a myth (Baird & Kerr 2007). Chatenoux and Peduzzi (2007) found less tsunami impact with mangrove cover but were unable to tease out any effect from the influence of coastal topography, as all their test sites with mangrove cover were situated in sheltered areas. Given the long period of tsunami waves, mangroves (a 'permeable barrier') cannot be expected to prevent flooding altogether, but potentially they can contribute to dissipating wave energy by increasing land roughness. Indeed, in gently inclined coasts, the extent of area inundated by the tsunami is limited not by the maximum height that the tsunami can reach but rather by the dissipation of the wave by drag forces as it flows over the more or less rough surface of the land (NERC et al. 2000). Vegetation affects roughness: models estimate that the inland inundation distance for a 10 m high tsunami on a flat area (mudflats, ice, open fields without crops; roughness coefficient 0.015) is 5700 m, while on forest or jungle (roughness coefficient 0.07) the inundation distance would be 260 m (NERC et al. 2000). The current situation is that insufficient studies have been done to either prove or disprove the value of mangrove forests for tsunami mitigation. Future tests are needed that control for key confounding factors such as bathymetry and topography (Cochard et al. 2008). It is likely that for the most violent tsunamis the buffering role is fairly negligible, while in less extreme situations the protective role of forests cannot be dismissed (Cochard et al. 2008).

Coastal dunes are less well developed in the tropics than in temperate zones. Dunes form barriers that can prevent waves (up to a certain height) from flowing inland and absorb energy from wave impact. Models and empirical evidence suggest that a tall sand dune barrier can significantly contribute to buffering of lower tsunami waves (Liu et al. 2005; Cochard et a. 2008). Dune vegetation does not normally play an important role in buffering wave impacts directly, but it is crucial for providing stability to sand dunes and the coastline (Cochard et al. 2008 and references therein). In temperate systems, dunes are often dominated by grass (such as *Ammophila arenaria*) that actively contribute to dune formation and growth by attenuating wind speed and trapping sediments. Higher dunes are more effective barriers. As *A. arenaria* density increases over time, dune height also increases due to sand deposition, and so does the height of waves that can overtop the dune (Barbier et al. 2008).

In tropical areas, dune development is suppressed by rapid colonisation by beach forest. This may have a significant buffering effect against storm surges and, as with mangroves, may in principle attenuate the effects of tsunami waves by increasing land roughness (Cochard et a. 2008). Though coastal tsunami protection forests have been planted and maintained in Japan for many years, little field data on potential damage mitigation by coastal ecosystems was available until the devastating tsunami of December 2004. Mature, single-species stands of planted trees (particularly coconuts, but also *Casuarina*) seem to offer less hydraulic resistance to small tsunamis than mangroves, since they often lack both dense vegetation undergrowth and a web of stilt roots. In contrast, multiple-layer beach forests may be more effective tsunami buffers (Cochard et al. 2008).

Wetlands such as salt marshes in estuaries and deltas are considered highly valuable for coastal protection, including from high intensity hurricanes such as Katrina (e.g. Costanza et al. 2006). A recent analysis (Costanza et al. 2008) found that differences in wetland area explain 60% of variation in hurricane-related damage in the US since 1980, estimating that coastal wetlands in the US currently provide $23.2 billion/yr in storm protection services. Coastal wetlands have been suggested to reduce the damaging effects of hurricanes on coastal communities by absorbing storm energy, including by: decreasing the area of open water for wind to form waves; increasing drag on water motion (and hence reducing the amplitude of a storm surge); and directly absorbing wave energy (Costanza *et al.* 2006).

What is the relationship between habitat area and the impact of ecosystems on the regulation/mitigation of natural hazards?

There is evidence, for at least some natural hazards, that the relationship between regulatory function and area/width is an inverse asymptotic relationship, with a small ecosystem area/width providing substantial protection, and subsequent added value increasing slower with increasing area/width. This is supported by empirical analysis of the effect of forests in reducing the frequency of catchment-borne floods (Bradshaw et al. 2007), and by measurements of attenuation of wind-generated waves as they cross mangroves and marshlands (Barbier et al. 2008).

We found no study on the relationship between the area covered by forest and landslide attenuation. It may be that in this case a small difference between nearly forested and 100% forested is actually more important than the same small difference between nearly deforested and 0% forest cover.

We found no study that investigated the relationship between habitat area (e.g. mangrove width) and impact reduction for tsunami waves. If any such effect does exist, it is likely that for smaller areas/widths it is negligible.

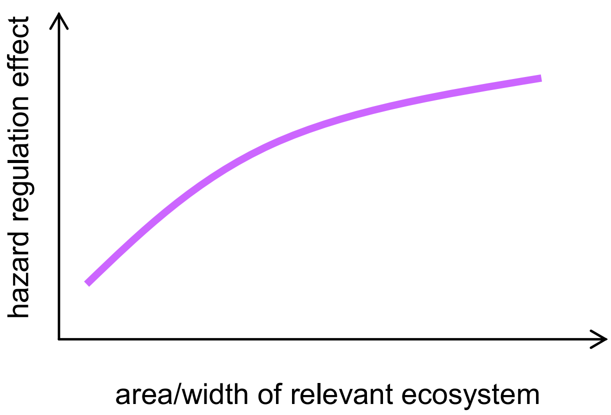

Figure 19. Relationship between the area/width of relevant ecosystem and the effect on the regulation of natural hazards (e.g., relationship between width of mangrove stand and the overall effect of the mangrove forest on regulation of storm surge).

4.13.4 What are the main threats to the regulation of natural hazards?

The main threats are those leading to a decline in coverage and/or degradation of the relevant ecosystems:

- Forests worldwide, and tropical forests in particular, are threatened by logging and agricultural expansion (Mayaux et al. 2005).

- Inland freshwater systems are threatened by: changes in land use (particularly vegetation clearance, drainage, and infilling); agricultural expansion; urbanisation; invasive species; hydrologic modification to inland waters; over-harvesting; pollution, salinisation, and eutrophication; and, global climate change (Finlayson et al. 2005).

- Threats to coral reefs include climate change, ocean acidification, direct destruction (e.g. destructive fishing methods; mining), overfishing, pollution, sedimentation, and disease (Wilkinson 2006).

- Seagrass beds global declines are attributed to dredging and anchoring in seagrass, meadows, coastal development, eutrophication, hypersalinization (resulting from changes to inflows), siltation, habitat conversion (e.g. for algae farming), and climate change (Agardy et al. 2005).

- Mangroves have been degraded by conversion to aquaculture, timber extraction, use of wood for fuel and charcoal production, diseases and storms (Valiela et al. 2001). Sea level rise poses another threat, but in the longer term mangroves are resistant to shoreline evolution (Alongi 2008).

- Beaches and dunes have undergone massive alteration due to coastal development, pollution, erosion, sand mining, groundwater use, and harvesting of organisms (Agardy et al. 2005).

- Coastal wetlands have undergone widespread change and destruction, particularly through draining for land reclamation, and are one of the global ecosystems predicted to suffer the most from climate change (Agardy et al. 2005).

These ecosystems vary in their resilience to disturbance and in their capacity to recover following destruction.

4.13.5 Are abrupt changes likely in the regulation of natural hazards?

If the relationship between hazard regulation and ecosystem extension is indeed an inverse asymptotic relationship (see above), then it may happen that in regions where past ecosystem loss has been extensive, the next losses may have a disproportionate effect in the provision of this service.

Coral reefs are subject to rapid changes which are not easily reversible (see Section 4.6). Hence, if they are found to have an important role in mitigating sea-borne natural hazards, this benefit may be at risk. Overfishing may trigger a transition to an algal-dominated system (Mumby et al. 2007) while climate change can cause sudden and/or widespread mortality due to increasing coral bleaching and consequent mortality, as well as weakening of the coral skeletons as a result of ocean acidification (Hoegh-Guldberg et al. 2007). Dead coral reefs retain their sheltering value for a few years, but eventually the skeletons begin to disintegrate and the reef crest starts eroding, reducing its dissipating effect on incoming waves (Sheppard et al. 2005).

Coastal ecosystems are often linked by physical as well as biological interactions (Cochard et al. 2008). Coral reefs' protection of shorelines facilitates the growth of mangroves and seagrasses, and so their degradation may have further repercussions for the capacity of ecosystems to mitigate natural hazards. On the other hand, mangroves and seagrasses bind soft sediments, facilitating reef development in areas that would otherwise have too much silt, and therefore preventing coral reef sedimentation.

Climate change, ocean acidification and sea-level rise are the conditions that are more likely to result in rapid, non-linear, and difficult to reverse changes in the value of ecosystems for mitigating some natural hazards. For example, climate change may increase the frequency of hurricanes or fires to such levels that natural vegetation has no the time to recover between events, while ocean acidification and bleaching may result in the quick demise of large coral reefs.

In many cases, the effect of ecosystems on natural hazard mitigation is not properly established yet. Assuming such an effect is established, abrupt changes in this service may be associated with abrupt changes in ecosystem extension and condition, for example the degradation of coral reefs or forests due to climate change. If the relationship between hazard regulation and ecosystem extension is an inverse asymptotic relationship, then it regions where past ecosystem loss has been extensive may suffer a disproportionate future decline in the provision of this service.

4.13.6 Can we quantify and map the global regulation of natural hazards and how it might change?

Catchment-borne floods

A near-global first-cut model is already available: Bradshaw and colleagues (2007) developed a generalized linear and mixed-effects model that controls for rainfall, slope and degraded landscape area to investigate the relationship between flood frequency and amount of remaining natural forest in each country. The model was developed and tested using data on flood events collected by the Dartmouth Flood Observatory between 1990 to 2000, from 56 developing countries, based on remote sensing as well as press reports.

This model would benefit from further developments and tests (Bruijnzeel et al. 2007[8]), including using catchments (a finer and more logical unit) as the unit of analysis and being

[8] Note that this comment by Bruijnzeel et al. 2007 is not currently published as a peer-reviewed article.

expanded to the global scale. Additionally, it would be desirable to test if the model would be improved if the presence of freshwater wetlands was also taken into consideration, possibly using data from the Global Database of Lakes, Reservoirs and Wetlands (Lehner and Döll 2004).

Better underlying data on floods would be desirable, as current data are prone to reporting bias in areas with more human population (Bruijnzeel et al. 2007[8]), but it is unlikely that better data will be made available within the next year. It would in any case be important to test explicitly if the current results are an artefact of such bias.

The model by Bradshaw and colleagues (2007) does not account for extreme rainfall events (e.g. during cyclones and typhoons). It would be desirable to investigate if this means that natural forest cover does not have an effect on flooding in these circumstances or if a model could be expanded which also accounted for these extreme events.

Landslides

There seem to be a wide diversity of modelling approaches for predicting landslide risk given information on topography, soil, geology, forest cover and landcover (ranging from logistic regression, to likelihood ratio models, to neural networks; e.g. Lee et al. 2007b; Morrissey et al. 2008). Presumably, for any of these models it should be possible to isolate the specific effect of forest cover, as well as to contrast different scenarios with different land uses. We did not review these models exhaustively, and did not contact experts specifically on this subject, so we are unable to advise what type of model would be most indicated.

Our recommendation is that, in Phase 2, Prof. David Petley, at Durham University, is contacted for advice. He is the founder and Director of the International Landslide Centre, which is compiling a worldwide landslide fatality database aiming at analysing global landslide risk (Petley et al. 2005).

Sea-borne hazards

There are diverse models investigating the effects of particular ecosystems on physical wave attenuation (e.g. Barbier et al. 2008). These produce useful information, but cannot be easily extrapolated to modelling coastal protection across entire regions. Furthermore, they do not account for the interaction between different ecosystems (e.g., mangroves sheltered by fringing coral reefs) or for the spatial configuration of barriers (e.g. gaps in coral reefs).

While a number of models have been developed on the effects of coastal vegetation on the impacts of the 2004 Indian Ocean tsunami, these have important limitations (see references above) and as such are not proposed as adequate for Phase 2. It is still a hypothesis that coastal vegetation has a significant effect in reducing impacts from tsunamis. Only if this hypothesis can be verified, and only if such effect can be quantified and predicted, can the value of ecosystems for mitigating tsunamis be estimated. Hydrodynamic models of tsunami propagation and inundation have been developed which take into account bathymetry and topography (MOST model: method of splitting tsunami; Titov et al. 2005; Borrero et al. 2006), and these would need to be refined to account for detailed information on how coastal landcover affects the propagation and the impact of tsunami waves. Such models could be calibrated using the vast amount of data collected in the aftermath of the 2004 Indian Ocean tsunami (Cochard et al. 2008), including high-quality aerial photographs as well as information collected from field surveys. This would provide a basis for substantially more reliable tests of the effects of ecosystems and – if an effect is found – a much more solid basis for generalisation across other tsunami-prone tropical coasts.

Regarding storm surges, such as those caused by cyclones and hurricanes, although they are more frequent, suitable data are not necessarily more available. A promising approach is the one developed by Costanza et al. (2008) to model the effects of wetlands for hurricane protection in the US. They used a regression model based on records of 34 major hurricanes since 1980, using the economic damage per unit GDP in the hurricane swath as the dependent variable and the wind speed and wetland area in the swath as the independent variables. GDP was mapped in a spatially-explicit way by using a linear allocation of the national GDP to the light intensity values of the night time image composite. In principle, this approach could be used to model several ecosystem types simultaneously (e.g., mangroves, corals and seagrasses) and their interactions.

Costanza et al. (2008) considered extending their approach to the global scale, but were prevented from doing so by the lack of good wetland maps. It is unlikely that such maps would become available within the next year. Data for other coastal systems may be more accessible though, through the World Atlas of Coral Reefs (Spalding et al. 2001), the World Atlas of Seagrasses (Green & Short 2003) and the ongoing update to the 1997 World Mangrove Atlas (Spalding et al. 1997) due in 2009. The Hawai'i Solar Observatory has data on tropical storm paths worldwide stretching back to 1995. If data on economic damage of different hurricanes can be obtained, the model developed by Costanza and colleagues can potentially be used as a first-cut model of hurricane damage considering the presence or absence of coral reefs, seagrasses and mangroves.

If it is not possible to obtain the necessary data at the global scale, another option would be to develop regional models based on finer data and use the results obtained in these to extrapolate globally.

What can be done in Phase 2? At what cost? By whom[9]?

Catchment-borne floods: we recommend developing the model by Bradshaw et al. (2007) as a first-cut basis for an economic evaluation. We do however recommend investing 12 researcher-months in further developments and testing of the model. This could be best done through a collaboration between the research team of Dr. Corey Bradshaw at the Charles Darwin University (Australia) and L.A. Bruijnzeel at the University of Amsterdam (Netherlands).

Tsunamis: we recommend investing 12 researcher-months in the development and test of a model based on the detailed data collected in the aftermath of the 2004 Indian Ocean tsunami. This would probably be best done through a collaboration between experts on tsunami hydrodynamics (e.g. Dr. Vasily Titov at the NOAA Center for Tsunami Research) and ecologists (e.g. Dr Roland Cochard at the Institute of Integrative Biology, Switzerland).

Storm surges: Provided the relevant data are available, a correlational model as the one developed by Costanza and colleagues (2008) could be used as a first-cut global model. If this is not possible, models could be developed for regional scales and results then extrapolated to the global scale. An investment of 12 researcher-months is suggested, based with Prof. Robert Costanza at the University of Vermont.

Landslides: it should be possible to model the specific effect of forest cover to landscape risk. We suggest contacting David Petley, at Durham University, Director of the International Landslide Centre.

Adequacy of scenarios

We need the scenarios to produce:

- global maps of forest cover
- global maps (or regional, if landscapes are analysed) of coral reefs, of mangroves, of seagrasses and of coastal wetlands.

4.13.7 Insights for the economic valuation

If using the Costanza et al. (2008) model for coastal storms, the results would already be produced in economic terms.

For the other models suggested, what would be modelled is the physical reduction in damage (flooding, tsunamis, and landslides). These results would then be crossed with data on vulnerability (e.g. nightlights as a spatially-explicit measure of relative GDP; Costanza et al. 2008) and on observed economic damage when hazards to happen to estimate the spatially explicit economic consequences of different scenarios (with higher or lower coverage of particular ecosystems).

[9] This is our recommendation for Phase 2, based on the results of this review. It does not commit the leaders of Phase 2 to follow it, and it does not commit the recommended research group to actually do such work.

4.13.8 *Some key resources*

- Dartmouth Flood Observatory keeps a Global Active Archive of Large Flood Events (1985-present) using information derived from a wide variety of news, governmental, instrumental, and remote sensing sources.
- International landslide centre is maintaining a worldwide landslide fatality database.
- The Hawai'i Solar Observatory has data on tropical storm paths worldwide stretching back to 1995, and a facility to calculate probability of strike given the co-ordinates of the location.
- EM-DAT database, a global database on natural and technological disasters that contains essential core data on the occurrence and effects of more than 17,000 disasters in the world from 1900 to present; maintained by the Centre for Research on the Epidemiology of Disasters (CRED), School of Public Health, Université Catholique de Louvain, Brussels, Belgium.
- Reliefweb records past natural disasters worldwide, including the extent of destruction and human suffering (but good maps of the area affected are not always available).
- UNEP-WCMC has compiled a *World Atlas of Seagrasses* (Green & Short 2003), and a *World Atlas of Coral Reefs* (Spalding et al. 2001). A *World Mangrove Atlas* (Spalding et al. 1997) is currently being updated.
- Global Lakes and Wetlands Database developed through a partnership between WWF and the University of Kassel in Germany (Lehner and Döll 2004).
- NOAA Center for Tsunami Research does research and development of methods to predict tsunami impacts on the population and infrastructure of coastal communities. Chief Scientist: Dr. Vasily Titov.

4.13.9 *Participants*

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4.14 One-time use benefits

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.14.1 Why are one-time use benefits important for human wellbeing?

We use the term "one-time use benefits" to refer to those benefits that are initially copied from, or inspired by, wild nature, but subsequently propagated outside of it. In this section, the term "natural products" refers to the biodiversity elements (e.g. species, genes, habitats) from where one-time use benefits are initially obtained.

One-time use benefits include those benefits obtained through:

- Bioprospecting: the search for new chemical compounds in organisms, particularly for pharmaceutical use (e.g. Newman & Cragg 2007).
- Biomimicry or bionics: "innovation inspired by nature" (Benuys 1997). Ideas for new materials or systems that are copied from, or inspired by, natural designs and processes, and used for solving human problems, for example in engineering, computational science, or architecture.
- Nature art: photography, films and art based on/inspired by nature (e.g. Attenborough 2007).
- Domestication: when a species that provides benefits to humans is propagated anthropogenically, such that those benefits are reaped from domesticated – rather than wild – populations. This includes, for example, the domestication of: pollinators (see also Section 4.1); crop and livestock species (see also Section 4.3); new agents for biological control (see also Section 4.2); fish species for aquaculture (see also Section 4.7); and species used to improve water quality in water treatment plants (see also Section 4.9).

This may correspond to a seemingly disparate set of benefits, but they have in common the fact that their production is directly dependent on nature's diversity (e.g. of species, genes), rather than on the abundance of particular species. Indeed, these benefits require a single use of nature, which can be consumptive (e.g., a plant is harvested to extract a chemical compound or to establish a domestic population) or not (e.g., a documentary is filmed in a natural area), but does not rely on a continuous supply from wild nature. After this one-time use, the benefits are propagated through anthropogenic means (e.g., propagation of chemical compounds through the pharmaceutical industry; propagation of images through television) and continue to aggregate economic value even if the original natural product is lost (e.g. if the original plant goes extinct or the natural area is destroyed). "One-time use" does not necessarily mean "instantaneous" – in the case of domestication, in particular, a long time may be needed for the benefit to become independent of the original natural product (i.e., from harvesting of the wild relatives of the domesticated species) and there may have been multiple origins for the same benefit (Olsen & Gross 2008).

Here, we focus on pharmaceutical compounds obtained from natural products, as more information is available on the benefits derived from these than from other types of one-time use.

Table 4. Examples of one-time use benefits, all featured as recent BBC News articles. See Beattie et al. (2005) for many other examples.

Species (reference)	**Benefit**	**BBC News article**
Sea cucumber (Yoshida et al. 2007).	Health: malaria treatment	Sea cucumber 'new malaria weapon' Sea cucumbers may provide a potential new way to block transmission of the malaria parasite.
South American Paradoxical Frog (Adbel-Wahab et al. 2008)	Health: diabetes treatment	Frog skin diabetes treatment hope Skin secretions from a South American "shrinking" frog could be used to treat type 2 diabetes.
Sea cucumber (Capadona et al. 2008)	Health: brain implants	'Sea slug' inspires brain implant Sea cucumbers inspire a novel material that could be used in brain implants.
Geckos and mussels (Lee et al. 2007a)	New materials: adhesive	Gecko glue exploits mussel power The remarkable adhesive abilities of geckos and mussels have been combined to create a super-sticky material.
Geckos (Mahdavi et al. 2008)	Health/new materials: waterproof medical bandage	Gecko inspires internal bandage The gecko lizard has helped inspire scientists in Massachusetts to develop a waterproof adhesive bandage for surgical wounds and internal injuries.
Southeast Asian beetle (Vukusic et al. 2007)	New materials: extremely white materials	White beetle dazzles scientists A dazzling insect could help the development of brilliant white, ultra-thin materials.
Bees (Jeong et al. 2006)	New technology: ultra-thin cameras	Insect eye inspires future vision An artificial insect eye that could be used in ultra-thin cameras has been developed by scientists in the US.
Spiders and geckos (Pugno 2007)	New materials: adhesive	'Spider-man' suit secret revealed A "Spider-man" suit that enables its wearer to scale vertical walls like the comic and movie superhero could one day be a reality.
Sea squirt (Grosso et al. 2007)	Health: cancer treatment	Sea squirt drug 'treats cancer' A drug made from the sea squirt may help those with a form of cancer.
Bats (Hedenstrom et al. 2007)	Ideas: fly engines	Flexible secrets of how bats fly Researchers use honey and a wind tunnel to image the amazingly flexible flap of a bat's wings.
Water lily	Ideas: solar panels	Water lily plan for solar power Large lily-shaped discs which harness solar power could soon be seen floating on the River Clyde.
Rattlesnake (Attenborough 2007)	Recreation: nature documentary	Wild rattlesnake hunt filmed A BBC crew has managed to film a rattlesnake hunting in the wild for the first time.
Caecilians (Attenborough 2007)	Recreation: nature	'Yummy mummy' caught on film A BBC crew has filmed a worm-like amphibian

	documentary	allowing her young to peel off her skin and eat it.
African Elephant (BBC 2007)	Recreation: nature photography	Elephant stomps to photo victory A picture of an elephant in a water-hole wins the Shell Wildlife Photographer of the Year award.

Benefits from one-time biodiversity use impact all aspects of human wellbeing (

Table 4; Beattie et al. 2005), including for example: food production (e.g. crop domestication), water provision (e.g. domestication of species used in water treatment), development of raw materials (e.g. new adhesive materials), health care (e.g. new pharmaceutical drugs), industrial applications (e.g. new sonar), energy production (e.g. solar panels), and recreation (e.g. nature documentaries).

One-time use does not include the extraction of benefits from nature through continuous harvest (e.g. as with medicinal plants, Section 4.11, or wild animal products, Section 4.8). Instead, the value of wild nature for the production of these one-time use benefits is wholly about option value (Pearce & Moran 1994). Indeed, given that by definition one-time use benefits, once extracted, become independent from nature, the current value of wild nature is not related to those benefits that have already been extracted, but to those that will be extracted in the future. That is, we value wild nature for one-time use benefits because it keeps options open for future (currently unknown) one-time uses.

Given that future one-time use benefits are still unknown, the economic value of the natural products from where they are derived is also unknown. However, judging by how much humanity has benefited in the past from just a few species, the future value for one-time use benefits is predicted to be very high. Indeed, of the estimated 5-30 million of living species on Earth, fewer than two million have been described and less than 1% of these have provided the basic resources for the development of all civilizations thus far (Beattie et al. 2005). Benefits obtained from some specific species have been extremely valuable to human wellbeing. Crop domestication is the prime example: most of mankind lives off no more than 12 plant species (Esquinas-Alcázar 2005) with 30 crops supplying 90% of the global calorie intake (Wood et al. 2005). A broad set of freshwater species are still being domesticated/selected for aquaculture production (FAO 2006b). About half of approved antitumor drugs worldwide are either natural products or directly derived from them (Newman & Cragg 2007); for example, two important cancer-fighting medicines have been extracted from the rosy periwinkle (*Catharanthus roseus*) – vinblastine, which has helped increase the chance of surviving childhood leukaemia from 10% to 95%, and vincristine, used to treat Hodgkins' Disease – generating about US$100 million per year in revenue to pharmaceutical companies (Karasov 2001).

It is therefore expected that the application of new technologies to the exploration of the currently unidentified species will yield many more future benefits for humanity (Beattie et al. 2005). However, attributing an economic value to the potential of species to yield valuable natural products is less straightforward. Given the low probability that individual species or sites will produce important discoveries, disappointingly small marginal values are typically obtained per species or per hectare of natural habitat. For example, Naidoo & Ricketts (2006; using calculations from Simpson et al. 1996) valued tropical forest in Paraguay for bioprospecting at $US 2.21/ha; and the celebrated bioprospecting contract

Merck Pharmaceutical and Costa Rica's Instituto Nacional de Biodiversidad, amounted to only US$1.1 million (Barrett & Lybbert 2000).

4.14.2 What are the overall trends in the provision of one-time use benefits?

We predict the trends are overwhelmingly positive, given increased scientific and technological developments that create the opportunities for these benefits to be explored (Beattie et al. 2005; Patterson & Anderson 2005).

On the other hand, an increase in the reliance of artificially designed molecules may result in a decline in the use of natural products by the pharmaceutical industry. That said, a "renaissance" of the use of natural products as drug candidates is taking place with the recognition that the development of purely synthetic molecules is not as efficient as the adaptation of natural molecules or the recreation of their analogues through synthetic pathways (Patterson & Anderson 2005).

Loss of species and of intraspecific genetic diversity are also likely to impact negatively on the provision of one-time use benefits. The Millennium Ecosystem Assessment reported a decline in the production of "biochemicals, natural medicines, pharmaceuticals" due to extinction and overharvest (MEA 2005), however this includes not only one-time use benefits but also natural medicines that rely on continuous harvest from natural ecosystems (and are vulnerable to overharvesting).

4.14.3 How is the provision of one-time use benefits affected by changes in wild nature?

The diversity of life provides the raw material for one-time use benefits. Given that they are not dependent on continuous harvest from wild populations, abundance (e.g. biomass, population size) is not directly relevant to the maintenance of these benefits, although it can be indirectly important in increasing resilience against diversity loss (extinction of species or populations). Abundant species (or, more specifically, those that are widespread) may however have a higher likelihood of being screened for valuable natural compounds or of becoming the inspiration for particular ideas or nature art. However, rarity is valued on its own right in some types of benefits (e.g., in nature documentaries).

It is not possible to predict which genes, species, or ecosystems will become valuable for one-time use benefits in the future. In the past, a wide variety of species (microbial, plant, and animal and their genes) have provided services, products, blueprints, or inspiration for products or the basis of industries. Species-rich environments such as tropical forests and coral reefs are expected to supply many benefits in the long term, but valuable natural products have been obtained from many diverse ecosystems, including temperate forests and grasslands, arid and semiarid lands, freshwater ecosystems, and montane and polar regions, as well as cold and warm oceans. In this context, the conservation of all biodiversity in all ecosystems would provide the most opportunities for one-time benefits in the future (Beattie et al. 2005).

Species, however, are not all the same. Those that are more phylogenetically distinct are more likely to contain irreplaceable information and therefore to be of higher economic value (Forest et al. 2007). Species that are simultaneously highly unique and highly threatened (e.g. EDGE species – evolutionarily distinct and endangered; Isaac et al. 2007) correspond to a higher risk of the irreplaceable loss of future one-time use benefits.

Species also differ in their likelihood of being sources of useful compounds. Species in highly diverse ecological communities tend to have more bioactive secondary compounds (e.g. chemical compounds produced by plants for protection against herbivory or for attracting pollinators,) than do species from simple communities (Roughgarden 1995). This may be because species in highly complex communities have evolved with many more inter-specific interactions that favoured the evolution of such compounds. If so, community species richness may be a good indicator of the value of species for the pharmaceutical industry and other sectors that make use of biological compounds.

Relationship between habitat area and one-time use benefits

Given their tight relationship with diversity, one-time use benefits are likely to vary with area following the species-area relationship (McArthur & Wilson 1967), in which a reduction in habitat results at first in a slow decline in diversity, but then diversity (and, presumably, one-time use benefits) declines very rapidly as the remaining area becomes quite small. This relationship may be even steeper given Roughgarden's (1995) suggestion that the incidence of potentially beneficial compounds is a positive non-linear function of diversity.

4.14.4 What are the main threats to the provision of one-time use benefits?

Given their dependency on biodiversity, one-time use benefits are likely to be very affected by all the main threats that are associated with the extinction of species including: habitat loss and degradation, over-harvesting, climate change, pollution, invasive species, and disease (Baillie et al. 2004). These same threats lead not only to the loss of species, but also to the loss of populations and broader erosion of genetic diversity (Ceballos & Ehrlich 2002).

One-time use benefits are also likely to be particularly affected by the loss of species that are more phylogenetically unique (with few living relatives), and whose extinction results in a disproportionate loss of phylogenetic diversity. Unfortunately, current extinction risk is not phylogenetically random and so phylogenetic diversity is being lost faster than species diversity (Purvis et al. 2000).

Species used in traditional medicine are often good candidates for screening in search of valuable pharmacological compounds. Indeed, not only are they more likely to have useful compounds, their long-term use by humans (often hundreds or thousands of years) also means that they are likely to have low human toxicity (Fabricant & Farnsworth 2001). The loss of traditional knowledge is therefore a serious threat to our capacity to obtain one-time use benefits from natural resources (Beattie et al. 2005).

4.14.5 Are abrupt changes likely in the provision of one-time use benefits?

Given the non-linear shape of the species-area relationship (McArthur & Wilson 1967), in areas where there has already been extensive habitat loss, a small additional loss of habitat will lead to a disproportionate loss in species diversity. Of particular concern are those regions of the world where there is high species endemism and that have already suffered substantial habitat loss (e.g. in biodiversity hotspots, such as Madagascar and the Philippines, which have at least 1,500 endemic plants and have already lost > 70% of their habitat; Myers et al. 2000; Mittermeier et al. 2004). Here we may see in the near future abrupt declines in species numbers, resulting in the irreversible loss of global species diversity, and therefore of potential future one-time use benefits.

Climate change and ocean acidification are already affecting biodiversity (Parmesan 2003) and some have predicted that these may become a threat as important as, or even more important than, habitat loss (Thomas et al. 2004). Some taxa, such as corals (Carpenter et al. 2008) are particularly at risk.

Loss of traditional knowledge is also likely to be happening at increasing rates given loss of traditional knowledge, for example on the use of medicinal plants (see Section 4.11).

Overall, we predict that there is a high probability that potential for one-time use benefits is being lost abruptly, particularly in high-biodiversity regions that have already suffered extensive habitat loss, for taxa that are particularly sensitive to climate change, and for regions where cultural knowledge is being lost very quickly.

4.14.6 Can we quantify and map the global provision of one-time use benefits and how it might change?

We believe it would be possible to create a first-cut model estimating how changes in land use would affect the global provision of one-time use benefits. At the very least, it would be possible to use methods developed in previous studies for calculating values per hectare of land. For example, Pearce & Moran (1994) obtained estimates of the value of plant species for future drugs, based on the probability of discovering a commercially valuable product, obtaining estimates of US$0.01 to US$21 per hectare of tropical forest. Simpson et al. (1996) used a more sophisticated approach, calculating the value of marginal species of higher plant at US$ 9431 (based on the incremental probability of making a commercial discovery from each plant, acknowledging the possibility of redundancy between species), and finding values between US$ 0.20 and US$21 per marginal hectare of tropical forest in biodiversity hotspots (Myers et al. 2000).

Like Simpson and colleagues (1996), we acknowledge that the marginal value of particular areas for the provision of future one-time use benefits depends on the uniqueness of its biota. If the same species occur elsewhere, then the loss of that particular area on its own will not result in the loss of any species, and therefore the options for obtaining future one-time benefits remain open. Ideally, we would need a method for estimating how many species are lost *globally* when a particular area is converted into a different form of landuse. Detailed information on species distributions – which could be used to calculate species extinctions directly – is not available, so a great deal of inference is needed.

We recommend that a global model of the provision of one-time use benefits is done using ecoregions (Olson et al. 2001) as the spatial unit, focusing on a first instance on plants as sources of compounds for the pharmaceutical industry.

A previous study estimated the number of plant species per ecoregion (Kier et al. 2005). Landcover data derived from satellite imagery (e.g. Bartholome & Belward 2005) can be used to estimate the area of remaining natural habitat in each ecoregion. Scenarios of future land use (Section 4.18) would indicate the predicted future area of natural habitat per ecoregion. The species-area relationship (McArthur & Wilson 1967) could then be used to estimate the number of species that are lost when a given area of habitat is lost within each ecoregion. Assuming a reasonable estimate is available for the overall global future value of plant compounds for the pharmaceutical industry, the marginal value per plant species can be obtained (e.g., Simpson et al. [1996] estimated this to be ~ US$ 10,000 per species) and so the economic value associated with a particular scenario of land use change could be estimated. This, however, is assuming that all plant species have the same probability of being the sources of valuable compounds. Another option is to assume, following Roughgarden (1995), that the probability of finding useful compounds is higher for species that have evolved in the most diverse communities. In the latter case, a higher economic value could be attributed to species in more diverse ecoregions. One option is to assume that the probability of a useful compound being present depends on the number of interspecific interactions in the community, which can be estimated as the square of species diversity. Different monetary values could then be attributed to species from different ecoregions, proportional to the square of ecoregional species diversity.

There are, however, two main complications with this overall approach:

a) Species may be lost from an ecoregion but retained somewhere else, and so remain available as potential natural sources for one-time use benefits. To avoid this from happening, the analysis would need to be restricted to endemic species, which only occur in each ecoregion. This would underestimate global losses (as it would miss species that occur in multiple ecoregions and become extinct in all of them) but it is likely to give a much more realistic estimate than by assuming that all species lost from an ecoregion are lost globally. Unfortunately, though, numbers of endemic plant species per ecoregion are not yet available.

b) The species-area relationship assumes that when natural habitat is lost, the area available to species effectively shrinks. In practice, the total area remains the same, but natural habitat is replaced with other habitats (e.g. forests replaced with shrubland or pastureland). While for some plant taxa, such as trees and lianas, the transition from natural forest to some types of transformed habitat (e.g. pastureland, agriculture) does generally result in the elimination of all species, for many taxa transformed habitats still retain some species (e.g. Barlow et al. 2007). Hence, if all natural habitat is lost, there are not necessarily 100% species extinctions. It is possible to adapt the species-area relationship to account for this, but this requires information on the fraction of native species that use transformed habitats. This information is not readily available.

It is unlikely that data on plant species endemism per ecoregion will be collected within the next year, but perhaps it could be inferred through modelling, using information on ecoregions for which data exists. At very least, the approach could be applied to the biodiversity hotspots (which are groups of ecoregions), for which numbers of endemic plants have been estimated (Mittermeier et al. 2004). Given that biodiversity hotspots contain about 50% of the world's plant species as single hotspot endemics, that most of them are also highly species' diverse, and that these regions face high levels of habitat loss, they are likely to include a very large fraction of the future losses of species that are valuable for the pharmaceutical industry. Another option is to apply this method to taxa for which distribution and endemism data are available at the ecoregional scale, such as birds, mammals and amphibians (Lamoreux et al. 2006), but this would assume that these groups are good surrogates for plants, which is unlikely to be the case.

Estimating the fraction of native plants that subsist in transformed habitats would not be straightforward, but we feel that a first approximation could be obtained through a comprehensive literature review of studies that compared plant species composition in natural and transformed habitats (e.g. Bhagwat et al. 2008).

> *What can be done in Phase II? At what cost? By whom?*
>
> We predict that it will be possible to produce within one year a first-cut global model to evaluate how the pharmaceutical industry would be affected by biodiversity loss (through modelling changes in land cover leading to species' extinction).
>
> We estimate that a dedicated post-doc would be required (12 researcher-months) to review the appropriate literature, build the model, and compare scenarios.
>
> This research could be conducted at the Conservation Science Group, Department of Zoology, University of Cambridge, in collaboration with the Centre for Social and Economic Research on the Global Environment (CSERGE), at the University of East Anglia[10].

4.14.7 Insights for economic valuation

The section describes above would provide information on how the global value of plants for the pharmaceutical industry could be modelled across space. However, the evaluation of such global value would still be needed, either based on existing approaches (e.g. Simpson et al. 1996) or others to be developed by economists.

4.14.8 Some key resources

- Gordon Cragg and David Newman at the Natural Products Branch, National Cancer Institute (Frederick, MD, USA) have compiled very useful data on the use of natural products in the pharmaceutical industry (particularly in drugs for cancer treatment). Key references include Cragg and Newman (2001, 2003) and Newman & Cragg (2004, 2007).

- The IUCN Red List of Threatened Species database has information on species that are threatened with extinction.

- The EDGE of Existence programme of the Zoological Society of London keeps information on species that are evolutionarily distinct and which are endangered with extinction.

- The Biomimicry Institute aims to "nurture and grow a global community of people who are learning from, emulating, and conserving life's genius to create a healthier, more sustainable planet".

[10] This is our recommendation for Phase II, based on the results of this review. It does not commit the leaders of Phase II to follow it, and it does not commit the recommended research group to actually do such work.

4.14.9 Participants

Authors

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Contributors

Bernardo Strassburg (University of East Anglia, UK)

Acknowledgements

Joan Roughgarden (Stanford University, USA) has provided information and references.

4.15 Non-use benefits

This section has not been developed.

Non-use benefits are derived simply from the knowledge that the natural environment is maintained. Economists recognise three main components (Defra 2007):

- Bequest value: where individuals attach value to the fact that a given natural resource is passed on to future generations;

- Altruistic value: where individuals attach value to the availability of a given natural resource to others in the current generation;

- Existence value: where individuals gain value simply from the existence of a resource, even though the individual has no actual or planned use of it.

One might expect that these values are declining, with the decline in traditional practices and beliefs (e.g. sacred species, sacred groves; McIvor & Pungetti 2008). However, this trend may to some degree be offset by the growing dissemination of biodiversity images (e.g. nature documentaries such as BBC 2006) and conservation values (e.g. increase in membership in conservation organisations; BirdLife International 2004).

These values are difficult to quantify (Turner 1999), and perhaps even more so it is to model the processes that generate and maintain them, and how they are expected to change under different scenarios. We believe that it is unlikely that the value of these benefits can be quantified in Phase 2, and so do not develop this theme any further.

This should not be interpreted to mean that these values are irrelevant in monetary terms; indeed, they may be more valuable than use values. For example, a study on the value of Natura 2000 sites in Scotland found that 99% of the overall value of such sites was non-use.

4.16 Global climate regulation

This section is based on a quick literature review, and did not receive contributions from, nor has it been reviewed by, experts in this field. We expect some of the uncertainties we identify below could be resolved by such a review process.

4.16.1 Why is global climate regulation important for human wellbeing?

There has been extensive work on the ways in which climate change influences human wellbeing (and therefore how limiting change or its effects may improve wellbeing). It is very well-established that climate change will have profound effects on wellbeing including through effects on food and water provision, health, property and infrastructure, global security and impacts on ecosystems and the benefits we derive from them. A disproportionate burden of these impacts falls on poor regions of the world, and so climate change is predicted to affect global development goals very significantly. Key references include:

- Stern (2007) – *Stern Review on the Economics of Climate Change*. This review has collated information on how climate change affects: water provision, food provision (including marine fisheries), health, land (including effects of sea-level rise), property/infrastructure, and the environment (including species extinctions). It further analysed how climate change affects development at global and regional scales.

- House et al. (2005) – the section of the *Millennium Ecosystem Assessment: Current State and Trends* on *Climate and Air Quality*, particularly the Section 13.6 on *Impacts of Changes in Climate and Air Quality on Human Well-being.*

- IPCCWGII (2007) – *Climate Change 2007 – Impacts, Adaptation and Vulnerability* (Working Group II contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change), including an assessment of impacts on fresh water resources, ecosystems, food, coastal systems, human health, and industry and an overview of how impacts might vary across continents.

4.16.2 How is global climate regulation affected by changes in wild nature?

There has also been extensive work on the mechanisms driving climate change, including the ways in which natural ecosystems affect the concentration of greenhouse gases, and the ways in which human activities are impacting those processes. It is well-established that terrestrial and aquatic ecosystems play key roles in the global geochemical cycles affecting the concentration of greenhouse gases in the atmosphere. Ecosystems also influence climate at a regional scale by affecting radiation (e.g. albedo), cloudiness, and surface temperature. Hence anthropogenic changes on ecosystems have significant impacts on climate change. Land use changes affecting forest ecosystems (deforestation, reforestation, afforestation) have a particularly strong influence on the balance of carbon emissions, sequestration, and storage. Forest fires also contribute to climate change by releasing aerosols. Other ecosystems play important roles, including: wetlands and reservoirs (with net methane emissions); grasslands (with variable degrees of underground carbon storage); peatlands and mires (important carbon storages); and agricultural lands (particularly nitrous oxide emissions, also methane). Oceans are substantial carbon reservoirs and sinks, although some studies suffest that these effects may be weakening with climate change itself (e.g., Zondervan et al. 2001; Le Quere et al. 2007). There has been plenty of work on the quantification of carbon storage, sinks and emissions for different ecosystems (including above and below ground) and these can be mapped at the global scale.

Key references include:

- SCBD (2003) *Interlinkages between biological diversity and climate change: Advice on the integration of biodiversity considerations into the implementation of the United Nations Framework Convention on Climate Change and its Kyoto protocol* discusses ways in which climate change affects biodiversity and also ways in which changes in natural systems affect greenhouse gases emissions.

- IPCCWGI (2007) – *Climate Change 2007 – The Physical Science Basis* (Working Group I Contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change), in particular chapter 7 on *Couplings Between Changes in the Climate System and Biogeochemistry*.

- IPCCWGIII (2007) – *Climate Change 2007 – Mitigation of Climate Change* (Working Group III Contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change) in particular chapter 9 on *Forestry*.

- Foley et al. (2005) includes a discussion of the ways in which land use change affects global climate.

- IPCC (2001), the *IPCC Special Report on Emissions Scenarios*, particularly Chapter 3: *Scenario Driving Forces* which discusses agriculture and land-use emissions and how these affect climate change scenarios.

- Pregitzer & Euskirchen (2004) discuss the role of forests in carbon cycling and storage.

- McGuire, A. D., Sitch, S., Clein, J. S., Dargaville, R., Esser, G., et al. (2001) Carbon balance of the terrestrial biopshere in the twentieth century: Analyses of CO_2, climate and land use effects with four process-based ecosystem models. Global Biogeochem Cycles 15, 183-206.

- Tian, H., Melillo, J. M., Kicklighter, D. W., McGuire, A. D. & Helfrich, J. (1999) The sensitivity of terrestrial carbon storage to historical climate variability and atmospheric CO2 in the United States. Tellus Ser B Chem Phys Meteorol 51, 414-452.

4.16.3 Can we quantify and map the value of wild nature for global climate regulation, and how it might change?

As decribed above, there has been extensive research on how human activities are affecting global climate, including on the consequences of changes in land use (e.g., McGuire et al. 2001; Foley et al. 2005). Land use maps coupled with models can be used to estimate differences in carbon storage, emissions and sequestration under different scenarios (e.g.

McGuire et al. 2001). We are therefore confident that it is possible to map the value of wild nature for global climate regulation, and how it might change under different scenarios.

What can be done in Phase 2? At what cost?

We are very confident that the effects of particular policy actions on global climate regulation can be quantified, and the corresponding economic consequences assessed, through the effects of the policy on landuse. We predict this could be done with an investment of 12 researcher-months.

4.16.4 Insights for the economic valuation

There has been important work on the calculation of the economic implications of climate change and the costs of climate change mitigation. This information is fundamental for deriving a price for carbon, which can be created through both tax and trading. The first international emissions trading scheme was created by the European Union in 2006. A carbon price can be used to quantify the economic consequences of different policy actions by assessing how they differ in terms of carbon storage, emission and sequestration by different ecosystems.

Key references include:

- Stern (2007) – *Stern Review on the Economics of Climate Change*, particularly its Section 15: *Carbon Pricing and Emissions Markets in Place*.

4.16.5 Participants

Authors

Hannah Peck, Ana Rodrigues, Andrew Balmford (University of Cambridge, UK).

4.17 Unknown benefits or processes

This section has not been developed.

Wild nature may potentially contribute to our future welfare in ways currently not realised, including both through ecosystem processes not currently known or valued (e.g., carbon storage and sequestration has only recently been identified as a valuable ecosystem process) and through benefits currently not predicted (e.g., algae are just starting to be considered as a promising biofuel). While these benefits are difficult or even impossible to measure, they may be very substantial.

This theme is not further developed, and the value of these benefits cannot be quantified in Phase 2.

4.18 Scenarios

Scenarios provide a methodology for exploring alternative future environments in which today's decisions might be played out. In practice, scenarios are stories, written or spoken, built around carefully constructed plots often termed narrative storylines. Scenarios are not predictions; instead, scenarios are an approach to help manage the inherent uncertainties of decisions by examining plausible, internally consistent alternatives of how the future might unfold and comparing the potential consequences of decisions in different future contexts (SEI, n.d.; Verbug et al. 2006). Scenarios may be purely qualitative, or may generate quantitative results through the use of one or more simulation models.

The generation of appropriate scenarios is a key component of Phase 2. In order to be useful in guiding policy, the assessment of economics of ecosystems and biodiversity needs to compare two future states of the world (see Section 2.3.2): one reflecting the continuation of business-as-usual, with the associated loss of ecosystems, biodiversity, and the benefits they bestow on people; and one in which in which such losses have been slowed. The contrasting scenarios should describe the losses of biodiversity and ecosystems, and the interventions needed to reduce them. These alternative states of the world can then be combined with models of ecosystem services to assess how the value of benefits foregone (under business-as-usual) compares with the costs of conservation (under the alternative scenario; see separate report on "Review of the Costs of Conservation and Priorities for Action", by Bruner, Naidoo and Balmford, also an output of this Scoping the Science project). Without such scenarios, all results obtained will be of questionable relevance.

Throughout we refer to two scenarios, but many more can and should be generated to assess specific policy packages. The scenarios compared need to be identical in everything apart from the specific policy package(s) being tested, and its effects; without this the economic results cannot be directly attributed to a difference in biodiversity or ecosystems. For example, an appropriate, specific contrast would be between the state of the world by 2025 generated by a business-as-usual (BAU) scenario, and an alternative state of the world generated by an otherwise identical scenario that includes the policy option of "a comprehensive global network of marine protected areas". These two scenarios could be contrasted to evaluate the specific economic consequences (both the costs and the benefits) of implementing the marine protected area network, as everything else (e.g., population, technology, consumption) would be the same. It would not be appropriate to try to evaluate the consequences of establishing a comprehensive marine reserve network by contrasting a 2008 state of the world (with its incipient network of marine protected areas) with a 2025 state of the world generated by the policy option scenario. Such a contrast would not take into account the changes in many other variables, including total human population, demand for marine fish, technological changes in fishing methods, etc.

In order to be useful, the scenarios need to have the right level of information at the right spatial scale. For example, if the action is to implement new protected areas, the scenario needs to state where they are; if the action is to manage fisheries appropriately, the scenario needs to spell out how that would be done, for example by creating no-fishing zones and setting sustainable fisheries quotas. If the scenarios do not have the necessary detail, they will be inappropriate to test how changes in wild nature affect the benefits we derive from them.

Scenario development has become a popular tool for the assessment of policy options, at scales ranging from national and regional (e.g. Ferreras et al. 2001; Green et al. 2005; Johansson et al. 2007; Rounsevell et al. 2006; Sandker et al. 2007; Soares-Filho et al. 2006) to global (e.g. House et al. 2002; Kindermann et al. 2006; Msangi et al. 2007; Sathaye et al. 2003; Ten Brink et al. 2007) and there is a growing literature on scenario generation and implementation. Recent large-scale scenario exercises likely to be relevant to the present assessment include: ATEAM (for Europe); the global and regional scale Millennium Ecosystem Assessment; the Global Environment Outlook exercises; the International Assessment of Agricultural Knowledge, Science and Technology for Development; the Intergovernmental Panel on Climate Change fourth assessment; the World Energy Outlook; and the OECD Outlook series (environment, agriculture, economics).

It is beyond the capacity of the current project to review the extensive literature on scenario development, to recommend which types of methods/approaches for generating scenarios are the most suitable for the current assessment, or to propose in any detail which specific scenarios should be contrasted. We emphatically recommend that this is a major priority for Phase 2 of the Review: indeed, this is the task that underlies the development of all other work.

For each review theme, we have made a general assessment of whether the scenario results currently being generated in global to regional scale exercises are likely to produce the level of information needed to allow modelling of the likely changes in benefits and services (see sections 4.1 - 4.17). In addition, we provide below a brief overview (but not an exhaustive list) of the main scenarios, which we hope covers the major models used in global-scale exercises.

Table 5. Main scenarios used in policy evaluation

Model / approach	**Example scenario exercises***	**Output variables**	**Spatial scale of future simulation**	**Theme relevance**	**Scope to improve**	**Reference**
Global climate models – Hadley Centre, CSIRO, etc.	IPCC (Nakicenovic & Swart 2000)	Climate variables such as temperature, precipitation, relative humidity	Multiple degrees - downscaling approaches permit finer scale results	All	Ongoing scope to improve integrated land cover models	Parry *et al.* 2007; Solomon *et al*. 2007
AIM	GEO, MA	Land use including biofuel production; agricultural production	5 degree grid – Asia-Pacific region	(Land use / cover)	Finer resolution	NIES 2008
		Surface runoff and river discharges	5 degree grid – Asia-Pacific region	(Freshwater)		

		Air quality – pollutant emissions (NOx, SO2, CO)	Grid (scale ?) – Asia-Pacific region	?		
CLUE / CLUE-S	GEO; regional exercises	Land use demand, land cover	Flexible – depends on inputs	(Land use / cover)		Verburg *et al*. 2002
EcoSim / EcoPath/ Ecospace	GEO, MA	Fish production; biomass	Regional	Fisheries	Freshwater fisheries simulation	Pauly et al. 2002
GLOBIO	GEO, GBO, OECD Environment Outlook	Biodiversity indicator 'mean abundance of original species'	0.5 degree	-	Additional indicators may be developed	Alkemade et al. 2006
		Change in ecosystem extent	0.5 degree (fractional cover per cell)	(Land use / cover)	Modelling approach is relatively basic	
GUMBO	'Big Gov / Eco-topia / Mad Max / Star Trek' set – Costanza (2000)	Land use	Global (11 biomes)	(Land use / cover)	Add spatial component	Boumans et al. 2002
		Ecosystem services (see model description below)		Direct simulation of services and their marginal value		
IFs	GEO	Multiple socio-economic variables (economics, food demand, population, poverty etc)	National scale	Useful to provide internally consistent sets of drivers for other models	More direct environmental limits	Hughes 2001; Hughes & Irfan 2007

		Environment variables: remaining fossil fuels, area of forested land, water usage, and CO2 emissions		Various; but other models typically have finer resolution	Finer resolution	
IIASA forest model	Carbon price scenarios	Forest area & biomass	0.5 degree	(Land use / cover); carbon	Finer resolution	Kindermann et al. 2006
IMAGE	GEO, GBO, IPCC, MA, IAASTD	Land cover – including cropland, pastureland, forest, urban…	0.5 degree	(Land use / cover)	Finer resolution	Bouwman et al. 2006
		Soil degradation	0.5 degree	(Land use / cover)		
		NPP	0.5 degree	Carbon storage		
		Timber production	24 regions of world (?)	Forest services		
		Nitrogen deposition	0.5 degree			
		Risk of water erosion of soils	24 regions (?)	(Land use / cover)		Potting & Bakkes 2004
IMPACT	GEO, MA	Crop, livestock consumption and production	69 regions of world	(Land use / cover)	Finer resolution	Rosegrant et al. 1995
		Fish consumption and production		Fisheries		
Macro-PDM	IPCC	River runoff	0.5 degree	(Freshwater / marine)	Integrate effects of land cover change	Arnell 2003
NEWS model suite	MA	Exports of nutrients from rivers to coastal zones	River basin mouths	Fisheries; oceanic carbon sequestration	Linkage with oceanic transport model	Seitzinger et al. 2005

WaterGAP	GEO, MA	Water flow and storage (surface runoff, groundwater recharge, river discharge, water storage in soil, groundwater and surface water bodies)	0.5 degree	Freshwater		Döll *et al.* 2003
		Water use (withdrawals for irrigation, livestock, households, thermal power plant and manufacturing)	0.5 degree	Freshwater		

***Acronyms – scenario exercises**

GEO – Global Environment Outlook (4)

GBO – Global Biodiversity Outlook (2)

IAASTD - International Assessment of Agricultural Knowledge, Science and Technology for Development

IPCC - Intergovernmental Panel on Climate Change assessment reports (4^{th})

MA – Millennium Ecosystem Assessment – Carpenter et al. 2005

Model descriptions

(Quotation marks indicate text directly from paper or webpage describing model.)

Global climate models.

Atmosphere-Ocean General Circulation Models designed to reproduce current and historical climate, and project the response of future climate to changes in variables such as the Earth's atmospheric composition, land cover and solar forcing. Confidence is higher for changes in temperature than in precipitation. See IPCC reports for further details.

AIM – Asia-Pacific Integrated Model

An integrated assessment model, designed to assess policy options for stabilising and adapting to changes in global climate, particularly in the Asian-Pacific region. Developed at National Institute for Environmental Studies, Japan.

CLUE / CLUE-S

Developed for the analysis of land use at a national to continental scale. CLUE-S is a variant for smaller regional scale applications (e.g., a watershed or province) at a fine spatial resolution. Demand is simulated with one module, and spatial distribution with another – it is possible to supply this second module with demands from a different external scenario. Model uses decision rules such as conversion probabilities between land uses. Developed at Wageningen University.

EcoSim / EcoPath / EcoSpace

Simulates biomass within marine food web models under the impacts of competing fishing fleets. EcoSim with EcoPath combines software for ecosystem trophic mass balance analysis (Ecopath), with a dynamic modeling capability (Ecosim) for exploring past and future impacts of fishing and environmental disturbances as well as for exploring optimal fishing policies. EcoSpace simulates the influence of protected area networks. Developed at University of British Columbia.

GLOBIO3

Developed for the analysis of impact of policy options on biodiversity. Indicator is 'mean abundance of original species of ecosystem' – hence a relative indicator of biodiversity compared to a reference 'natural' state, rather than an absolute indicator of biodiversity compared between locations. Based on a set of cause-effect relationships grounded in the literature on the impacts of pressures on biodiversity. Developed at the Netherlands Environment Assessment Agency / UNEP/GRID Arendal / UNEP-WCMC.

GUMBO - Global Unified Metamodel of the Biosphere

"Gumbo consists of five sectors or spheres: Atmosphere, Lithosphere, Hydrosphere, Biosphere, and Antrophosphere (human systems). It is also divided into 11 biomes or ecosystem types which encompass the entire surface area of the planet: Open Ocean, Coastal Ocean, Forests, Grasslands, Wetlands, Lakes/Rivers, Deserts, Tundra, Ice/rock, Croplands, and Urban. Global energy, carbon, nitrogen, and water dynamics are incorporated in Gumbo, along with soil erosion and formation, fossil fuel formation and use, plant and animal productivity and harvest, human population, energy, biomass, and water use, economic production and welfare, and changes in natural, built, human, and social capital". Simulates ecosystem services on a global scale, based on flows and storages in the model. The services are: gas regulation, climate regulation, disturbance regulation, water use, soil formation, nutrient cycling, waste treatment, food production, raw materials, and recreation/cultural. Developed at University of Vermont.

IFs - International Futures

"The broad purpose of IFs is to serve as a thinking tool for the analysis of near through long-term country-specific, regional, and global futures across multiple, interacting issue areas". The overall model incorporates different sub-models, including: Population, Economics, Agriculture, Education, Energy, Socio-Political, the International Political, Environment, Technology, and Health. Model relies on simulation of macroagents (e.g. governments) and markets. Developed at University of Denver.

IIASA forest model [precise name unclear]

"Calculates differences in net present value of different land uses using a spatially explicit integrated biophysical and socio-economic land use model. Key model parameters, such as agricultural land use and production, population growth, deforestation and forest product consumption rates were calibrated against historical rates. Land use changes are simulated in the model as a decision based on a difference between net present value of income from production on agricultural land versus net present value of income from forest products. Assuming fixed technology, the model calculates for each 0.5° grid cell the net present value difference between agricultural and forest land-uses in one-year time steps"

IMAGE - Integrated Model to Assess the Global Environment

The IMAGE model simulates direct and indirect pressures on human and natural systems resulting from industry, housing, transport, agriculture and forestry. Socio-economic activities and drivers of change are allocated to 24 regions of the world, and the climate, land-cover and land-use change-related processes are represented on a 0.5 degree grid. Developed at the Netherlands Environment Assessment Agency.

IMPACT - International Model for Policy Analysis of Agricultural Commodities and Trade

IMPACT has as its basis a set of national supply and demand equations for 32 commodities, with each country model being linked to the rest of the world through trade. "To explore food security effects, IMPACT projects the percentage and number of malnourished preschool children (0 to 5 years old) in developing countries as a function of average per capita calorie availability, the share of females with secondary schooling, the ratio of female to male life expectancy at birth, and the percentage of the population with access to safe water. A wide range of factors with potentially significant impacts on future developments in the world food situation can be modeled based on IMPACT. They include: population and income growth, the rate of growth in crop and livestock yield and production, feed ratios for livestock, agricultural research, irrigation and other investments, commodity price policies, and elasticities of supply and demand. For any specification of these underlying factors, IMPACT generates projections for crop area, yield, production, demand for food, feed and other uses, prices, and trade; and for livestock numbers, yield, production, demand, prices, and trade". Developed at International Food Policy Research Institute.

Macro-PDM

Global scale river runoff model designed for use with IPCC scenarios. Changes in runoff are largely driven by change in precipitation, offset by increases in evaporation. Each 0.5 degree cell is treated as an independent catchment. Developed at Tyndall Centre.

NEWS model suite - Global Nutrient Export from Watersheds

Simulates river nutrient export from watersheds. "The Global NEWS system includes models that can be used to predict export of sediments, DIN, DIP, DOC, DON, dissolved organic P (DOP); particulate organic C (POC), PN, and PP" – based on factors such as inputs of fertiliser, manure and sewage. Developed through a workgroup of UNESCO's International Oceanographic Commission.

WaterGAP

"The overall aim is to investigate current and future world-wide water availability, water use and water quality. Whereas water availability and water use have been implemented, a water quality module is currently under development and constitutes the next major goal. In particular, Water-GAP is concerned with the various impacts of global change on water availability and water demand, and to determine the development of water stress conditions on different spatial and temporal scales. Further research subjects are the variations of the water balance components on the hydrological large scale, and the future development of extreme conditions - such as floods and droughts". Developed at University of Kassel / University of Frankfurt.

Towards Phase 2

One or more of these scenarios may provide a useful framework for scenario development for Phase 2. However, as the theme-specific reviews make clear, properly modelling the consequences for service provision of losing wild nature will typically require much more detailed outputs than those currently provided by global scenarios – with spatially-explicit data on the extent and condition of all major land-cover types, climate, settlement patterns, demands for different services, resource-harvesting regimes, and so on. Hence scenario development, linked to the identification of specific interventions needed to slow the loss of wild nature, should in our view be a major activity during Phase 2.

4.18.1 Participants

Author

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4.19 Prioritisation of recommendations for future work

This section builds from the results of the thematic reviews (sections 4.1 to 4.17) to prioritise amongst the analyses recommended those that are most important and most feasible for Phase 2.

Scenario development

The first priority for Phase 2, underlying all subsequent work, is the development of appropriate scenarios, built as counterfactuals to test the specific policy package that is being tested.

As emphasised in sections 2.3.2 and 4.18, it is fundamental that in the evaluation of the economic consequences of biodiversity loss and ecosystem degradation, two scenarios are compared. These need to be identical in everything else but the specific policy package being tested. Otherwise, the economic results cannot be directly attributed to a difference in the state of biodiversity/ecosystems. Key steps in scenario development include spatially-explicit quantitative descriptions of the likely state of biodiversity under business-as-usual and under a set of policy interventions deemed to be sufficient to conserve biodiversity; and the identification and costing of those interventions. Care must be taken to ensure that the quantitative, mapped descriptions of the alternative states of the world are fit-for-purpose – that is to say that they address all of the key biophysical and socioeconomic input variables needed to feed into the models of the production and flow of each process and service of interest.

Prioritising recommendations for Phase 2

The thematic reviews (sections 4.1 to 4.17) present an assessment of the feasibility, for each particular theme, of quantifying the provision of ecosystem processes or benefits in a spatially-explicit way as a basis for the economic valuation in Part 2. Recommendations are presented for how such quantification can be done, and the resources estimated to be necessary for each analysis. Here, we prioritise amongst those recommended analyses, based on two criteria (Figure 20):

- **Importance to human wellbeing**: The degree to which these processes/benefits are likely to affect the overall results of the Review. This is based on a qualitative assessment of the predicted magnitude of the overall economic value of each process or benefit. Recommended analyses were coded, in decreasing order of importance, as A, B, or C. We have marked with an asterisk (*) those benefits that are particularly valuable for local livelihoods (which may have low overall economic value but may directly influence the wellbeing of millions of people).
- **Feasibility in Phase 2**: The likelihood that the particular analysis recommended can be successfully undertaken in one year. Recommendations were coded, in decreasing order of feasibility, as 1, 2 or 3.

The combination of these two criteria produces an overall priority ranking (Figure 20), from very high priority, when the analysis is both highly important and highly feasibly (A1), to very low priority, when the analysis is of lower importance and not particularly feasible in Phase 2 (C3).

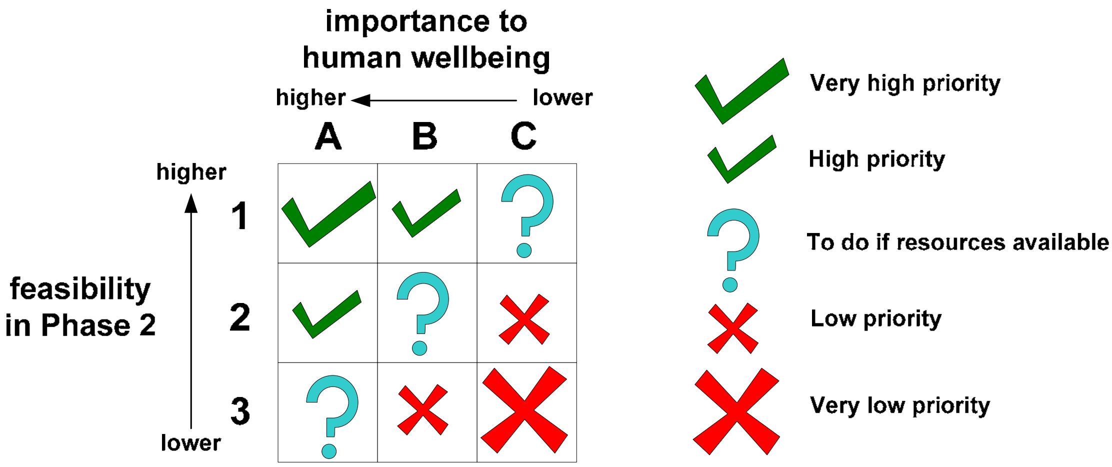

Figure 20. Prioritisation of future research lines according to their importance to human wellbeing (A to C) and feasibility in Phase 2 (1 to 3). Priority levels range from very high priority (A1) to very low priority (C3).

Table 6 presents the results of the prioritisation for each of the analyses recommended for the benefits/processes considered in this project.

Prioritising recommendations for research in the longer-term

For the longer-term, priorities are defined based on importance alone. Indeed, all of the analyses proposed are feasible, and so should all be done if time is less of a limitation. In this case, the focus should become to ensure that the most important benefits or processes are covered first.

The priority then becomes to tackle analyses coded as A first, followed by B and by C.

Table 6. Prioritisation of the recommended analyses for each category of benefit or process considered in this review.

Benefit/process	**Recommended analysis (see corresponding section for more detail)**	**Priority code**	**Priority level**
All	• Scenario development	**Essential**	**Essential**
Wild crop pollination	• Global pollination model building from landscape-scale assessments	B2	
Biological control of crop pests	• Global crop biological control model building from landscape-scale assessments	B3	

Genetic diversity of crops and livestock	• Global risk assessment of crop/livestock disease from loss of genetic diversity	B2?-3?	?
Soil quality for crop production	• Global valuation of improvement in soil quality from changes in soil biota (internal effects)	B3	✗
	• Global valuation of soil subsidies when agriculture expands into natural ecosystems (conversion effects)	B1*	✓
	• Global model of the protective effects to crop soils of neighbouring natural habitat (neighbouring effects)	C2*	✗
	• Global model of value of wildlife fertilisers.	C3*	✗
Livestock	• Global model of rangeland contribution to livestock production	B1*	✓
Marine fisheries	• Global model of marine fisheries provision	A1*	✓
Inland fisheries	• Global model of inland fisheries provision including aquaculture	B2*	?
Wild animal products	• Pantropical model of wild meat provision	B1*	✓
	• European and North-American model of domestic recreational hunting	B1	✓
Fresh water provision and regulation	• Global hydrological model for water provision (quantity)	A1*	✓
	• Global hydrological model for water regulation (timing)	A2*	✓
	• Global hydrological model for water purification (quality)	A2*	✓
Wild harvested fibres	• Global model of sustainable timber production from natural forests	B2?-3?*	?
Wild medicinal plants	• Global model of provision of medicinal plants (harvesting)	B3*	✗

Nature-related outdoor activities	• Global model of tourism in protected areas	B3	✗
	• Global model of the value of green areas for local recreation	B2	?
Natural hazard regulation	• Global model of the value of forests (and possibly wetlands) for regulating catchment-borne floods	B2*	?
	• Global model of the value of coastal ecosystems for mitigating the effects of tsunamis	C2*	✗
	• Global model of the value of coastal ecosystems for mitigating the effects of storm surges	B2*	?
	• Global model of the value of forests for preventing landslides	C2*	✗
One-time biodiversity use	• Global model of loss of option values for benefits derived from single species	B1	✓
Non-use values	• Global valuation of non-material benefits from biodiversity	B3	✗
Global climate regulation	• Global terrestrial model of carbon storage and sequestration	A1	✓
Unknown benefits or processes	• Non-quantifiable		

4.19.1 Participants

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5 INVENTORY OF RESEARCH ORGANIZATIONS, PROGRAMMES AND RECENT LITERATURE DEALING WITH ECONOMICS OF BIODIVERSITY LOSS

As a contribution to the Scoping the Science project, an inventory was made of the main current research programmes, organizations and literature dealing with economics of biodiversity loss. This Chapter presents the main results of these reviews and Annexes 6 and 7 list more detailed information on the results. It must be stressed that both the literature search and inventory of organisations and programmes is not exhaustive, thus to obtain a more complete overview it is foreseen that this identification tasks will be continued as a part of the upcoming EU initiatives on economic of biodiversity loss.

5.1 Literature review

An inventory was made of the main publications since 2005 (i.e. post the release of the Millennium Ecosystem Assessment). The research has been realised using the following search engines: Science direct, Scopus and Google scholar (succinctly), as well as direct search on the home pages of the Journals Ecological Economics, Nature and Science.

A combination of key words was used during the research for each search engine. These included “Biodiversity + Economic Loss”, “Biodiversity + Economic Cost” and “Biodiversity + Economic Valuation”.

From these key words the articles including all the terms in the text and published between 2005-2008 (included) have been selected. A total of 132 documents have been gathered, showing an increasing trend: 2005: 29 articles, 2006: 36 articles, 2007: 46 articles, 2008: 13 articles published and 8 articles in press.

The selected articles come from different sources the main ones (more than 5 articles) being: Ecological economics (21), Science (14), Biological Conservation (10), and Conservation Biology (6).

5.2 Review of organisations, programmes and networks

The inventory of organizations, programmes and projects was carried out by using the main well-known websites as a starting point (see box 1). Most of these websites have a section with ‘links’ and by following these in a systematic way it is possible to obtain a good overview of the main organisations and programmes.

Similar to the literature search, keywords used were the link between “biodiversity” and “economics / economic loss / economic cost / economic valuation”. When going “deeper” into the various organisations and programmes, many links are found to more specific subjects (e.g. programmes, projects and organisations focusing on specific ecosystems (e.g. watershed-services, coral reefs, forests, etc), services or regions). Within the timeframe of this project it was not possible to explore these links in any detail but that could be possibly recommended as one of the future follow-up exercises.

BOX A. Main networks and websites dealing with economics of biodiversity (loss)

Biodiversity Economics (www.biodiversityeconomics.org)

Ecosystem Service Database (ESD) (//esd.uvm/edu)

Ecosystem Services (www.esa.org/ecoservices)

Ecosystem Services Database (esd-worldbank.org/eei/)

Ecosystem Services project (www.ecosystemservicesproject.org)

Ecosystem Valuation (www.ecosystemvaluation.org)

EnValue: www.epa.nsw.gov.au/envalue/

EVRI (www.evri.ca)

MILLENNIUM ECOSYSTEM ASSESSMENT (www.maweb.org)

Natural Capital Project (www.naturalcapitalproject.org)

Nature Valuation & Financing Network (NV&F) **(**www.naturevaluation.org)

In Annex 6 an overview is given of the main organisations and programmes, organised by type of organisation including the following categories: main networks/websites; multi-lateral institutions, programmes, treaties etc.; government supported initiatives; NGO's; universities, research organizations & programmes; and business supported initiatives. Per category a distinction is made between international and national organisations and programmes. Within these categories, organizations are listed in alphabetical order for easy reference.

This overview does not aim to be exhaustive, however in combination with the links given in Box A on the main networks & websites it should enable quick and easy access to the main organisations and programmes on this topic[11]. In the future, some resources could be directed to identifying relevant networks at the national level. Ideally, a formal "network-analysis" should be performed to obtain more quantitative information on the main "nodes" in the national and international networks.

[11] In case you miss an (important) organization or program, please send an email to *dolf.degroot@wur.nl*

One aim of this inventory has been to establish active links and working relations between the Scoping the Science project and the main organisations. Facilitating communication among the main Networks is an important objective of the Nature valuation and Financing Network (NV&F) (www.naturevaluation.org) which already has a quite extensive list of links and active working relations and could possibly be used to facilitate this process in the future.

5.2.1 Participants

Authors

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REFERENCES

Abdallah, S., Thompson, S., & Marks, N. 2008. Estimating worldwide life satisfaction. *Ecological Economics* **65**: 35-47.

Achard, F., Eva, H. D., Stibig, H., Mayaux, P., Gallego, J., Richards, T., & Malingreau, J. 2002. Determination of deforestation rates of the world's humid tropical forests. *Science* **297**: 999-1002.

Agardy, T., Alder, J., Dayton, P., Curran, S., Kitchingman, A., Wilson, M., Catenazzi, A., Restrepo, J., Birkeland, C., Blaber, S., Saifullah, S., Branch, G., Boersma, D., Nixon, S., Dugan, P., et al. 2005. Coastal Systems. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: World Resources Institute (pp. 515-543).

Aguilar-Støen, M., & Moe, S. 2007. Medicinal plant conservation and management: distribution of wild and cultivated species in eight countries. *Biodiversity and Conservation* **16**: 1973-1981.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., & Siebert, S. 2003. Development and testing of the WaterGAP 2 global model of water use and availability. *Hydrological Sciences Journal* **48**: 317-337.

Alcamo, J., Flörke, M., & Märker, M. 2007. Future long-term changes in global water resources driven by socio-economic and climatic changes / Changements futurs a long terme dans les ressources en eau globales forces par les changements climatiques et socio-economiques. *Hydrological Sciences Journal* **52**: 247-275.

Alcamo, J., Vuuren, D. V., Ringler, C., Cramer, W., Masui, T., Alder, J., & Schulze, K. 2005. Changes in nature’s balance sheet: model-based estimates of future worldwide ecosystem services. *Ecology and Society* **10**: 19.

Alder, J., Guénette, S., Beblo, J., Cheung, W., & Christensen, V. 2007. Ecosystem-based global fishing policy scenarios. *Fisheries Centre Research Reports* **15**: 1-83.

Alfsen, K. H., Bye, T., Glomsrød, S., & Wiig, H. 2001. Soil degradation and economic development in Ghana. *Environment and Development Economics* **2**: 119-143.

Alin, S. R., Cohen, A. S., Bills, R., Gashagaza, M. M., Michel, E., Tiercelin, J., Martens, K., Coveliers, P., Mboko, S. K., West, K., Soreghan, M., Kimbadi, S., & Ntakimazi, G. 296. Effects of Landscape Disturbance on Animal Communities in Lake Tanganyika, East Africa. *Conservation Biology* **13**: 1017-1033.

Alkemade, R., Bakkenes, M., Bobbink, R., Miles, L., Nellemann, C., Simons, H., Tekelenburg, T., 2006. GLOBIO 3: Framework for the assessment of global terrestrial biodiversity. In: Bouwman, A. F., Kram, T., Klein (Eds.), 2006. *Integrated modelling of global environmental change. An overview of IMAGE 2.4. Netherlands Environmental Assessment Agency*, Bilthoven, the Netherlands, pp. 171-186.

Allan, J. D., Abell, R., Hogan, Z., Revenga, C., Taylor, B.W., Welcomme, R. L. & Winemiller, K. 2005 Overfishing of Inland Waters. *BioScience* **55**: 1041-1051 .

Alongi, D. M. 2008. Mangrove forests: Resilience, protection from tsunamis, and responses to global climate change. *Estuarine, Coastal and Shelf Science* **76**: 1-13.

Alves, R. R., & Rosa, I. L. 2005. Why study the use of animal products in traditional medicines? *Journal of Ethnobiology and Ethnomedicine* **1**.

Amornsakchai, S., Annez, P., Vongvisessomjai, S., Choowaew, S., Thailand Development Research Institute (TDRI), Kunurat, P., Nippanon, J., Schouten, R., Sripapatrprasite, P., Vaddhanaphuti, C., Vidthayanon, C., Wirojanagud, W., Watana, E. 2000. *Pak Mun Dam, Mekong River Basin, Thailand. A WCD Case Study prepared as an input to the World Commission on Dams*, Cape Town, South Africa: World Commission on Dams.

Araujo-Lima, C., & Goulding, M. 1997. *So Fruitful a Fish*. Columbia University Press.

Arnell, N. W. 1999. A simple water balance model for the simulation of streamflow over a large geographic domain. *Journal of Hydrology* **217**: 314-335.

Arnell, N. W. 2004. Climate change and global water resources: SRES emissions and socio-economic scenarios. *Global Environmental Change* **14**: 31-52.

Arnell, N. W., 2003. Effects of IPCC SRES emissions scenarios on river runoff: a global perspective. *Hydrology and Earth System Sciences* **7**: 619-641.

Arnold, J. G., & Fohrer, N. 2005. SWAT2000: current capabilities and research opportunities in applied watershed modelling. *Hydrological Processes* **19**: 563-572.

Arthurton, R., Barker, S., Rast, W., Huber, M., Alder, J., Chilton, J., Gaddis, E., Pietersen, K., Zockler, C., Al-Droubi, A., Dyhr-Nielsen, M., Finlayson, M., Fortnam, M., Kirk, E., Heileman, S., Rieu-Clark, A., et al. 2007. Chapter 4 - Water. In *Global Environment Outlook 4*. Nairobi, Kenya: GEO4 United Nations Environment Programme.

Attenborough, D. 2006. *Planet Earth*. Bristol, UK: BBC.

Attenborough, D. 2007. *Life in Cold Blood*. Bristol, UK: BBC.

Aylward, B. 2005. Land use, hydrological function and economic valuation. In M. Bonell & Leendert Adriaan Bruijnzeel (Eds.), *Forests, Water, and People in the Humid Tropics: Past, Present, and Future*. Cambridge, UK: Cambridge University Press (pp. 99-120).

Badola, R., & Hussain, S. A. 2005. Valuing ecosystem functions: an empirical study on the storm protection function of Bhitarkanika mangrove ecosystem. *Environmental Conservation* **32**: 85-92.

Baillie, J.E.M, Hilton-Taylor, C, Stuart, S.N (Eds.) 2004. *2004 IUCN Red List of Threatened Species. A Global Species Assessment*. Gland, Switzerland and Cambridge UK: IUCN – The World Conservation Union.

Baird, A. H., Campbell, S. J., Anggoro, A. W., Ardiwijaya, R. L., Fadli, N., Herdiana, Y., Kartawijaya, T., Mahyiddin, D., Mukminin, A., Pardede, S. T., Pratchett, M. S., Rudi, E., & Siregar, A. M. 2005. Acehnese Reefs in the Wake of the Asian Tsunami. *Current Biology* **15**: 1926-1930.

Baird, A. H., & Kerr, A. M. 2007. The dangerous myth of tsunami greenbelts: response to Stone 2006. *Science: E-Letter* **314** Online: http://www.sciencemag.org/cgi/eletters/314/5798/404#5422.

Baird, A., & Kerr, A. 2008. Landscape analysis and tsunami damage in Aceh: comment on Iverson and Prasad (2007). *Landscape Ecology* **23**: 3-5.

Bale, J., van Lenteren, J., & Bigler, F. 2008. Biological control and sustainable food production. *Philosophical Transactions of the Royal Society B: Biological Sciences* **363**: 761-776.

Balirwa, J. S. 2007. Ecological, environmental and socioeconomic aspects of the Lake Victoria's introduced Nile perch fishery in relation to the native fisheries and the species culture potential: lessons to learn. *African Journal of Ecology* **45**: 120-129.

Balmford, A., Gaston, K. J., Blyth, S., James, A., & Kapos, V. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences USA* **100**: 1046–1050.

Balmford, A., Gravestock, P., Hockley, N., McClean, C. J., & Roberts, C. M. 2004. The worldwide costs of marine protected areas. *Proceedings of the National Academy of Sciences USA* **101**: 9694-9697.

Baran, E., Starr, P., & Kura, Y. (Eds.). 2007. *Influence of Built Structures on Tonle Sap Fisheries*. Phnom Penh, Cambodia: Cambodia National Mekong Committee and the WorldFish Centre.

Barber, R. G., & Díaz, O. 1994. Maintenance of yields and soil fertility in nonmechanized cropping systems, bolivia. *Soil Sci Soc Am J* **58**: 858-866.

Barbier, E. B. 2007. Valuing Ecosystem Services as Productive Inputs. *Economic Policy* **22**: 177-229.

Barbier, E., Koch, E., Silliman, B., Hacker, S., Wolanski, E., Primavera, J., Granek, E., Polasky, S., Aswani, S., Cramer, L., Stoms, D., Kennedy, C., Bael, D., Kappel, C., Perillo, G., et al. 2008. Coastal Ecosystem-Based Management with Nonlinear Ecological Functions and Values. *Science* **319**: 321-323.

Bardgett, R. D., & McAlister, E. 1999. The measurement of soil fungal: bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands. *Biology and Fertility of Soils* **29**: 282-290.

Barlow, J., Mestre, L. A., Gardner, T. A., & Peres, C. A. 2007. The value of primary, secondary and plantation forests for Amazonian birds. *Biological Conservation* **136**: 212-231.

Barnes, R. F. 2002. The bushmeat boom and bust in West and Central Africa. *Oryx* **36**: 236-242.

Barrett, C. B., & Lybbert, T. J. 2000. Is bioprospecting a viable strategy for conserving tropical ecosystems? *Ecological Economics* **34**: 293-300.

Bartholome, E., & Belward, A. 2005. GLC2000: a new approach to global land cover mapping from Earth observation data. *International Journal of Remote Sensing* **26**: 1959-1977.

BBC. 2007. *Wildlife Photographer of the Year: Portfolio 17*. London, UK: BBC Books.

Beattie, A. J., Barthlott, W., Elisabetsky, E., Farrel, R., Kheng, C. T., Prance, I., Rosenthal, J., Simpson, D., Leakey, R., Wolfson, M., & ten Kate, K. 2005. New Products and Industries from Biodiversity. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 271-295).

Beck M. W, Heck Jr K. L, Able K. W, Childers D. L, Eggleston D. B, Gillanders B. M, Halpern B, Hays C. G, Hoshino K, Minello T. J, Orth R. J, Sheridan P. F, & Weinstein M. P. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* **51**: 633-641.

Bell, D., Roberton, S., & Hunter, P. R. 2004. Animal origins of SARS coronavirus: possible links with the international trade in small carnivores. *Philosophical Transactions of the Royal Society B*: 1107–1114.

Bellwood, D. R., Hughes, T. P., Folke, C., & Nystrom, M. 2004. Confronting the coral reef crisis. *Nature* **429**: 827-833.

Bengtsson, J., Ahnstrom, J., & Weibull, A. 2005. The effects of organic agriculture on biodiversity and abundance: a meta-analysis. *Journal of Applied Ecology* **42**: 261-269.

Bennett, E. L. 2002. Is there a link between wild meat and food security? *Conservation Biology* **16**: 590-592.

Bennett, E.L. & Rao, M. 2002. Wild meat consumption in Asian tropical forest countries: is this a glimpse of the future for Africa? In S. Mainka & M. Trivedi (Eds) *Links between Biodiversity, Conservation, Livelihoods and Food Security: The Sustainable Use of Wild Species for Meat.* Gland, Switzerland: IUCN – The World Conservation Union (pp. 39–44).

Benyus, J. M. 1997. *Biomimicry: Innovation Inspired by Nature*. William Morrow.

Bhagwat, S. A., Willis, K. J., Birks, H. J. B., & Whittaker, R. J. 2008. Agroforestry: a refuge for tropical biodiversity? *Trends in Ecology & Evolution* **23**: 261-267.

Bianchi, F., Booij, C., & Tscharntke, T. 2006. Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings of the Royal Society B* **273**.

Biesmeijer, J. C., Roberts, S. P. M., Reemer, M., Ohlemuller, R., Edwards, M., Peeters, T., Schaffers, A. P., Potts, S. G., Kleukers, R., Thomas, C. D., Settele, J., & Kunin, W. E. 2006. Parallel Declines in Pollinators and Insect-Pollinated Plants in Britain and the Netherlands. *Science* **313**: 351-354.

Bioversity International. 2008. Bioversity International - Plants and Animals. Online: http://www.bioversityinternational.org/Plants_and_Animals/index.asp.

Bird, W. 2005. *Natural Fit: Can Green Space and Biodiversity Increase Levels of Physical Activity*? Sandy, UK: RSPB.

Bird, W. 2007. *Natural Thinking*. Sandy, UK: RSPB

BirdLife International. 2004. *State of the World's Birds 2004 - Indicators for Our Changing World*. Cambridge, UK: BirdLife International.

Blaikie, P., Cannon, T., Davis, I., & Wisner, B. 2004. *At Risk: Natural Hazards, People's Vulnerability and Disasters*. London, UK: Routledge.

Bloem J., Hopkins, D.W. & Benedetti, A. (Eds.) 2006. *Microbiological methods for assessing soil quality*. Wallingford, UK: CABI.

Bloem, J., de Ruiter, P.C. & Bouwman, L.A. 1997. Food webs and nutrient cycling in agro-ecosystems. In J.D. van Elsas, J.T. Trevors and E. Wellington (Eds.), *Modern Soil Microbiology*. New York, USA: Marcel Dekker Inc (pp. 245-278).

Bloem, J., Schouten, T., Didden, W., Jagers op Akkerhuis, G., Keidel, H., Rutgers, M., & Breure, T. 2004. Measuring soil biodiversity: experiences, impediments and research needs. In R. Francaviglia (Ed.), *Agricultural impacts on soil erosion and soil biodiversity: developing indicators for policy analysis. Proceedings of the OECD expert meeting on soil erosion and soil biodiversity indicators, 25-28 March 2003, Rome, Italy*. Paris, France: OECD.

Bockstael, N., Freeman, A., Kopp, R., Portney, P., & Smith, V. 2000. On Measuring Economic Values for Nature. *Environmental Science & Technology* **34**: 1384-1389.

Bodeker, G., & Burford, G. 2007. Medicinal Plant Biodiversity and Local Healthcare: sustainable use and livelihood development. In *Traditional, Complementary and Alternative Medicine*. United Kingdom: Imperial College Press (pp. 145-166).

Bodmer, R. E., Eisenberg, J. F., & Redford, K. H. 1997. Hunting and the Likelihood of Extinction of Amazonian Mammals. *Conservation Biology* **11**: 460-466.

Börjeson, L., Höjer, M., Dreborg, K.-H., Ekvall, T., Finnveden, G. 2005. *Towards a user's guide to scenarios - a report on scenario types and scenario techniques*. Royal Institute of Technology, Stockholm, Sweden.

Botsford, L. W., Micheli, F., & Hastings, A. 2003. Principles for the design of marine reserves. *Ecological Applications* **13**: 25.

Boumans, R., Costanza, R., Farley, J., Wilson, M. A., Portela, R., Rotmans, J., Villa, F., Grasso, M., 2002. Modeling the dynamics of the integrated earth system and the value of global ecosystem services using the GUMBO model. *Ecological Economics* **41**: 529-560.

Bouwman, A. F., Kram, T., Klein (Eds.), 2006. *Integrated modelling of global environmental change. An overview of IMAGE 2.4*. Netherlands Environmental Assessment Agency, Bilthoven, the Netherlands.

Boyd, J., & Banzhaf, S. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* **63**: 616-626.

Brack, D. 2007. *Illegal Logging: Briefing Paper*. London, UK: The Royal Institute of International Affairs. Online: http://www.chathamhouse.org.uk/files/9384_bp0707illegallogging.pdf

Bradshaw, C. J. A., Sohdi, N., Peh, K., & Brook, B. 2007. Global evidence that deforestation amplifies flood risk and severity in the developing world. *Global Change Biology* **13**: 2379-2395.

Brainard, J., Bateman, I., & Lovett, A. 2001. Modelling demand for recreation in English woodlands. *Forestry* **74**: 423-438.

Brander, K. M. 2007. Global fish production and climate change. *Proceedings of the National Academy of Sciences USA* **104**:19709-19714.

Brashares, J. S., Arcese, P., & Sam, M. K. 2001. Human demography and reserve size predict wildlife extinction in West Africa. *Proceedings of the Royal Society* B **268**: 2473–2478.

Brashares, J. S., Arcese, P., Sam, M. K., Coppolillo, P. B., Sinclair, A. R. E., & Balmford, A. 2004. Bushmeat hunting, wildlife declines, and fish supply in West Africa. *Science* **306**: 1180-1183.

Brauman, K. A., Daily, G. C., Duarte, T. K., & Mooney, H. A. 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. *Annual Review of Environment and Resources* **32**: 67-98.

Brosi, B. J., Daily, G. C., Shih, T. M., Oviedo, F., & Duran, G. 2007 The effects of forest fragmentation on bee communities in tropical countryside. *Journal of Applied Ecology*. doi:10.1111/j.1365-2664.2007.01412.x.

Brown, C., & Grant, M. 2005. Biodiversity and Human Health: What Role for Nature in Healthy Urban Planning? *Planning Healthy Towns and Cities* **31**: 326-338.

Bruijnzeel, L. A. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment* **104**: 185-228.

Bruijnzeel, L., van Dijk, A. I., van Noordwijk, M., Chappell, N. A., & Schellekens, J. 2007. *Tropical deforestation, people and flooding*. CIFOR. [online] http://www.worldagroforestry.org/water/downloads/bca_bruijnzeel.pdf.

Brussaard, L., de Ruiter, P. C., & Brown, G. G. 2007. Soil biodiversity for agricultural sustainability. *Agriculture, Ecosystems & Environment* **121**: 233-244.

Bubb, P., May, I., Miles, L., & Sayer, J. 2004. *The Cloud Forest Agenda Report*. Cambridge, UK: UNEP-World Conservation Monitoring Centre.

Buchert, G.P., Rajora, O.M.P., & Hood Bruce P. Dancik, J.V. 1997. Effects of harvesting on genetic diversity in old-growth eastern white pine in Ontario, Canada. *Conservation Biology* **11**: 747-758.

Bull, G., Mabee, W., & Scharpenberg, R. 1998. *Global Fibre Supply Model*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO).

Bullock, A., & Acreman, M. 2003. The role of wetlands in the hydrological cycle. *Hydrology and Earth System Sciences* **7**: 358-389.

Capadona, J. R., Shanmuganathan, K., Tyler, D. J., Rowan, S. J., & Weder, C. 2008. Stimuli-Responsive Polymer Nanocomposites Inspired by the Sea Cucumber Dermis. *Science* **319**: 1370-1374.

Carlton, J. T., Geller, J. B., Reaka-Kudla, M. L., Norse, E. A. 1999. Historical Extinctions in the Sea. *Annual Review of Ecology and Systematics* **30**: 515–38.

Carpenter, A. I., Marcus Rowcliffe, J., & Watkinson, A. R. 2004. The dynamics of the global trade in chameleons. *Biological Conservation* **120**: 291-301.

Carpenter, K. E., Abrar, M., Aeby, G., Aronson, R. B., Banks, S., Bruckner, A., Chiriboga, A., Cortes, J., Delbeek, J. C., DeVantier, L., Edgar, G. J., Edwards, A. J., Fenner, D., Guzman, H. M., Hoeksema, B. W., et al. 2008. One-Third of Reef-Building Corals Face Elevated Extinction Risk from Climate Change and Local Impacts. *Science*: 1159196.

Carpenter, S. R., & Kitchell, J. F. 1988. Consumer control of lake productivity. *BioScience* **38**: 764-769.

Carpenter, S. R., Kitchell, J. F., Hodgson, J. R., Cochran, P. A., Elser, J. J., Elser, M. M., Lodge, D. M., Kretchmer, D., He, X., & von Ende, C. N. 1987. Regulation of lake primary productivity by food web structure. *Ecology* **68**: 1863-1876.

Carpenter, S. R., Pingali, P. L., Bennett, E. M., Zurek, M. B. (Eds.), 2005. *Ecosystems and Human Well-Being: Volume 2 Scenarios: Findings of the Scenarios Working Group*. Island Press, Washington DC.

Carr, D., Pan, W., & Bilsborrow, R. 2006. Declining fertility on the frontier: the Ecuadorian Amazon. *Population & Environment* **28**: 17-39.

Carver, S., Evans, A., & Fritz, S. 2002. Wilderness attribute mapping in the United Kingdom. *International Journal of Wilderness* **8**: 25-29.

Cassman, K. G., Wood, S., Choo, P. S., Cooper, H. D., Devendra, C., Dixon, J., Gaskell, J., Khan, S., Lal, R., Lipper, L., Pretty, J., Primavera, J., Ramankutty, N., Viglizzo, E., Wiebe, K., et al. 2005. Chapter 26: Cultivated Systems. In *Millennium Ecosystem Assessment: Current State & Trends Assessment*. Washington DC, USA: Island Press (pp. 745-794).

Cavelier, J., & Goldstein, G. 1989. Mist and fog interception in elfin cloud forests in Colombia and Venezuela. *Journal of Tropical Ecology* **5**: 309-322.

CBD (Convention on Biological Diversity). 2006. 2010 *Biodiversity Target: Indicators.* Online: http://www.cbd.int/2010-target/framework/indicators.shtml.

Ceballos, G., & Ehrlich, P. R. 2002. Mammal population losses and the extinction crisis. *Science* **296**: 904-907.

Ceballos-Lascurain, H. 1996. *Tourism, ecotourism and protected areas*. IUCN, Gland, Switzerland.

Chamberlain, A. 1910. Scenery as a national asset. *Outlook* **95**: 169.

Chatenoux, B. & Peduzzi, P.. 2007. Impacts from the 2004 Indian Ocean Tsunami: analysing the potential protecting role of environmental features. *Natural Hazards* **40**: 289-304.

Chatenoux, B., & Peduzzi, P. 2005. *Analysis on the Role of Bathymetry and other Environmental Parameters in the Impacts from the 2004 Indian Ocean Tsunami*. Geneva, Switzerland: UNEP/GRID-Europe.

Chivenge, P., Murwira, H., Giller, K., Mapfumo, P., & Six, J. 2007. Long-term impact of reduced tillage and residue management on soil carbon stabilization: Implications for conservation agriculture on contrasting soils. *Soil and Tillage Research* **94**: 328-337.

Christensen, V., & Walters, C. J. 2004. Ecopath with Ecosim: methods, capabilities and limitations. *Ecological Modelling* **172**: 109-139.

Christensen, V., & Walters, C. J. 2005. Using ecosystem modeling for fisheries management: Where are we? *ICES Annual Science Conference, Aberdeen, Scotland, 20-24 September 2005*: 1-16.

Christensen, V., Guenette, S., Heymans, J. J., Walters, C. J., Watson, R., Zeller, D., & Pauly, D. 2003. Hundred-year decline of North Atlantic predatory fishes. *Fish and Fisheries* **4**: 1-24.

Christie, M., Han ley, N. & Hynes, S. 2007. Valuing enhancements to forest recreation using choice experiment and contingent behaviour models. *Journal of Forest Economics* **13**: 75-102.

Christner, B. C., Morris, C. E., Foreman, C. M., Cai, R., & Sands, D. C. 2008. Ubiquity of Biological Ice Nucleators in Snowfall. *Science* **319**: 1214.

CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora). 2008. The CITES Species. *Convention on International Trade in Endangered Species of Wild Fauna and Flora*. Retrieved April 26, 2008, from http://www.cites.org/eng/disc/species.shtml

Claudet, J., Osenberg, C. W., Benedetti-Cecchi, L., Domenici, P., Garcia-Charton, J., Perez-Ruzafa, A., Badalamenti, F., Bayle-Sempere, J., Brito, A., Bulleri, F., Culioli, J., Dimech, M., Falcon, J. M., Guala, I., Milazzo, M., et al. 2008. Marine reserves: size and age do matter. *Ecology Letters* **11**: 481-489.

Clayton, L., Keeling, M., & Milner-Gulland, E. J. 1997. Bringing home the bacon: a spatial model of wild pig hunting in Sulawesi, Indonesia. *Ecological Applications* **7**: 642-652.

Coates D., Ouch Poeu, Ubolratana Suntornratana, N Thanh Tung & Sinthavong Viravong. 2003. *Biodiversity and fisheries in the Lower Mekong Basin*. Mekong Development Series No. 2. Phnom Penh, Vietnam: Mekong River Commission.

Cochard, R., Ranamukhaarachchi, S. L., Shivakoti, G. P., Shipin, O. V., Edwards, P. J., & Seeland, K. T. 2008. The 2004 tsunami in Aceh and Southern Thailand: A review on coastal ecosystems, wave hazards and vulnerability. *Perspectives in Plant Ecology, Evolution and Systematics* **10**: 3-40.

Cochrane, M. A., & Laurance, W. F. 2002. Fire as a large-scale edge effect in Amazonian forests. *Journal of Tropical Ecology* **18**: 311-325.

Cole, F. R., Medeiros, A. C., Loope, L. L., & Zuehlke, W. W. 1992. Effects of the Argentine ant on arthropod fauna of Hawaiian high-elevation shrubland. *Ecology* **73**: 1313-1322.

Collen, P. and Gibson, R. J. 2001 The general ecology of beavers (*Castor* spp.), as related to their influence on stream ecosystems and riparian habitats, and the subsequent effects on fish – a review. *Reviews in Fish Biology and Fisheries* **10**: 439-461.

Cook, R. M., Sinclair, A., & Stefansson, G. 1997. Potential collapse of North Sea cod stocks. *Nature* **385**: 521-522.

Cooke, S. J. & Cowx, I. G. 2006 Contrasting recreational and commercial fishing: Searching for common issues to promote unified conservation of fisheries resources and aquatic environments. *Biological Conservation* **128**: 93-108.

Corlett, R. T. 2007. The impact of hunting on the mammalian fauna of tropical Asian forests. *Biotropica* **39**: 292-303.

Costanza R., d'Arge R., Groot R.d., Farber S., Grasso M., Hannon B., Limburg K., Naeem S., O'Neill R.V., Paruelo J., Raskin R.G., Sutton P. & Belt M.v.d. 1997. The value of the world's ecosystem services and natural capital. *Nature* **387:** 253 - 260.

Costanza, R., 2000. Visions of alternative (unpredictable) futures and their use in policy analysis. *Ecology and Society* **4**: 5+.

Costanza, R., Mitsch, W. J., & Day, J. 2006. Creating a sustainable and desirable New Orleans. *Ecological Engineering* **26**: 317-320.

Costanza, R., Pérez-Maqueo, O., Martinez, M. L., Sutton, P., Anderson, S. J., & Mulder, K. 2008. The value of coastal wetlands for hurricane protection. *Ambio* **37**: 241–248

Cote, I. M., Mosqueira, I., & Reynolds, J. D. 335. Effects of marine reserve characteristics on the protection of fish populations: a meta-analysis. *Journal of Fish Biology* **59**: 178-189.

Cowlishaw, G., Mendelson, S., & Rowcliffe, J. M. 2005. Evidence for post-depletion sustainability in a mature bushmeat market. *Journal of Applied Ecology* **42**: 460-468.

Cragg, G., & Newman, D. 2001. Medicinals for the Millennia: The Historical Record. *Ann NY Acad Sci* **953**: 3-25.

Cragg, G., & Newman, D. 2003. Plants as a source of anti-cancer and anti-HIV agents. *Annals of Applied Biology* **143**: 127-133.

Crookes, D., Ankudey, N., & Milner-Gulland, E. J. 2005. The value of a long-term bushmeat market dataset as an indicator of system dynamics. *Environmental Conservation* **32**: 333-339.

Crooks, K. R., & Soulé, M. E. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. *Nature* **400**: 563-566.

Daby, D. 2003. Effects of seagrass bed removal for tourism purposes in a Mauritian bay. *Environmental Pollution* **125**: 313-324.

Dahdouh-Guebas, F., Jayatissa, L., Di Nitto, D., Bosire, J., Lo Seen, D., & Koedam, N. 2005. How effective were mangroves as a defence against the recent tsunami? *Current Biology* **15**: R443-R447.

Daily, G. (Ed.). 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press.

Daily, G., Ceballos, G., Pacheco, J., Suzan, G., & Sanchez-Azofeita, A. 2003. Countryside biogeography of Neotropical mammals: conservation opportunities in agricultural landscapes of Costa Rica. *Conservation Biology* **17**: 1814-1826.

Danielsen, F., Sorensen, M. K., Olwig, M. F., Selvam, V., Parish, F., Burgess, N. D., Hiraishi, T., Karunagaran, V. M., Rasmussen, M. S., Hansen, L. B., Quarto, A., & Suryadiputra, N. 2005. The Asian Tsunami: A Protective Role for Coastal Vegetation. *Science* **310**: 643.

Darby, S. E. 1999. Effect of riparian vegetation on flow resistance and flood potential. *Journal of Hydraulic Engineering* **125**: 443-454.

Darwall, W., Smith, K., Lowe, T. and Vié, J.-C. 2005. *The Status and Distribution of Freshwater Biodiversity in Eastern Africa*. Gland, Switzerland and Cambridge, UK: IUCN SSC Freshwater Biodiversity Assessment Programme. IUCN.

Daskalov, G. M., Grishin, A. N., Rodionov, S., & Mihneva, V. 2007. Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. *Proceedings of the National Academy of Sciences* USA **104**: 10518-10523.

DeFoliart, G. R. 1999. Insects as food: why the western attitude is important. *Annual Review of Entomology* **44**: 21-50.

Defra. 2007. *An introductory guide to valuing ecosystem services*. London, UK: Defra - Department for Environment Food and Rural Affairs. Available: http://www.defra.gov.uk/wildlife-countryside/natres/pdf/eco_valuing.pdf

Devine, J. A., Baker, K. D., & Haedrich, R. L. 2006. Fisheries: Deep-sea fishes qualify as endangered. *Nature* **439**: 29.

Di Falco, S., & Perrings, C. 2005. Crop biodiversity, risk management and the implications of agricultural assistance. *Ecological Economics* **55**: 459-466.

Díaz, S., Tilman, D., Fargione, J., Chapin, F.S., Dirzo, R., Kitzberger, T., Gemmill, B., Zobel, M., Vilà, M., Mitchell, C., Wilby, A., Daily, G.C., Galetti, M., Laurance, W.F., Pretty, J., et al. 2005. Chapter 11: Biodiversity Regulation of Ecosystem Services. In *Millennium Ecosystem Assessment: Current State & Trends Assessment*. Washington DC, USA: Island Press (pp. 297-329).

Dickie, I., Hughes, J., & Esteban, A. 2006. *Watched like never before; the local economic benefits of spectacular bird species.* Sandy, UK: RSPB.

do Prado, A., de Barros Cavalcanti, F., & Prudente de Fontes, P. 2005. *Forest Management Working Paper: Computerized Data Gathering and Networking as a Control and Monitoring System for the Improvement of and Reporting on Forest Management in the Amazon: the Case of Brazil*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Dolfman, M. L., Wasser, S. F., & Bergman, B. 2007. The effects of Hurricane Katrina on the New Orleans economy. *Monthly Labor Review* **June 2007**: 3-18.

Döll, P., & Siebert, S. 2002. Global modeling of irrigation water requirements. *Water Resources Research* **38**: 1037.

Döll, P., Kaspar, F., & Lehner, B. 2003. A global hydrological model for deriving water availability indicators: model tuning and validation. *Journal of Hydrology* **270**: 105-134.

Donlan, J., Berger, J., Bock, C. E., Bock, J., Burney, D. A., Estes, J. A., Foreman, D., Martin, P., Roemer, G., Smith, F., Soule, M., & Greene, H. 2006. Pleistocene Rewilding: An optimistic agenda for Twenty First Century conservation. *The American Naturalist* **168**: 660-681.

Douglas, E., Wood, S., Sebastian, K., Vorosmarty, C., Chomitz, K., & Tomich, T. 2007. Policy implications of a pan-tropic assessment of the simultaneous hydrological and biodiversity impacts of deforestation. *Water Resources Management* **21**: 211-232.

Dransfield, J., Tesoro, F., & Manokaran, N. (Eds.). 2002. *Rattan: Current Research Issues and Prospects for Conservation and Sustainable Development*. Rome, Italy: Food and Agriculture Organization of the United Nations and International Network for Bamboo and Rattan.

Duarte, C. M. 2002. Marine biodiversity and ecosystem services: an elusive link. *Journal of Experimental Marine Biology and Ecology* **250**: 117-131.

Dudley, N., & Stolton, S. 2003. *Running pure: the importance of forest protected areas to drinking water*. World Bank/WWF Alliance for Forest Conservation and Sustainable Use.

Dugan, P., Dey, M. M. and Sugunan, V. V. in press. Fisheries and water productivity in tropical river basins: Enhancing food security and livelihoods by managing water for fish. *Agricultural Water Management*

Dugan, P., Sugunan, V. V., Welcomme, R. L., Béné, C., Brummett, R. E., & Beveridge, M. C. 2007. Inland fisheries and aquaculture. In D. Molden (Ed.), *Water for Food, Water for Life: A Comprehensive Assessment of Water Management in Agriculture*. Washington D.C., USA: Earthscan and International Water Management Institute (pp. 459-483).

Duraiappah, A. K., Naeem, S., Agardy, T., Ash, N. J., Cooper, H. D., Díaz, S., Faith, D. P., Mace, G., McNeely, J. A., Mooney, H. A., Oteng-Yeboah, A. A., Pereira, H. M., Polasky, S., Prip, C., Reid, W. V., et al. 2005. Biodiversity Synthesis Report. In *Millennium Ecosystem Assessment: Synthesis Reports*. Washington DC, USA: Island Press.

Eagles, P. 2002. Trends in park tourism: economics, finance and management. *Journal of Sustainable Tourism* **10**: 132-153.

Eagles, P.F.J., McLean, D. & Stabler, M.J. 2000. Estimating the tourism volume and value in parks and protected areas in Canada and the USA. George Wright Forum **17**: 62-82.

East, T., Kümpel, N. F., Milner-Gulland, E., & Rowcliffe, J. M. 2005. Determinants of urban bushmeat consumption in Rio Muni, Equatorial Guinea. *Biological Conservation* **126**: 206-215.

EJF (Environmental Justice Foundation). 2006. *Mangroves: Nature's defence against Tsunamis— A report on the impact of mangrove loss and shrimp farm development on coastal defences.* London, UK: Environmental Justice Foundation.

Elías, E. 2004. The Tarapoto Process: establishing criteria and indicators for the sustainable management of Amazon forests. *Unasylva* **55**.

Elliott, J., Grahn, R., Sriskanthan, G., & Arnold, C. 2002. *Wildlife and Poverty Study*. Livestock and Wildlife Advisory Group, Department for International Development.

Enright, W. D. 2000. The effect of terrestrial invasive alien plants on water scarcity in South Africa. *Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere* **25**: 237-242.

Entwistle, A. & Dunstone, N. 2000. *Priorities for the conservation of mammalian biodiversity*. Cambridge University Press, Cambridge, UK.

Ernst, C. 2004. *Protecting the Source: Land Conservation and the Future of America's Drinking Water*. Washington, D.C.: Trust for Public Land.

Esquinas-Alcázar, J. 2005. Protecting crop genetic diversity for food security: political, ethical and technical challenges. *Nat Rev Genet* **6**: 946-953.

EuroCoML 2008. Census of Marine Life on Seamounts. Available online: http://www.eurocoml.org/index.php?option=com_content&task=view&id=83&Itemid=119.

Fa, J. E., & Purvis, A. 1997. Body size, diet and population density in afrotropical forest mammals: a comparison with neotropical species. *The Journal of Animal Ecology* **66**: 98-112.

Fa, J. E. & Peres, C. 2001. Game vertebrate extraction in African and Neotropical forests: an intercontinental comparison. In J. Reynolds, G. Mace, J.G. Robinson and K. Redford (Eds). *Conservation of Exploited Species.* Cambridge, UK: Cambridge University Press. (pp. 203-241).

Fa, J. E., Currie, D., & Meeuwig, J. 2003. Bushmeat and food security in the Congo Basin: linkages between wildlife and people's future. *Environmental Conservation* **30**: 71-78.

Fa, J. E., Peres, C. A., & Meeuwig, J. 2002. Bushmeat exploitation in tropical forests: an intercontinental comparison. *Conservation Biology* **16**: 232-237.

Fa, J. E., Ryan, S. F., & Bell, D. J. 2005. Hunting vulnerability, ecological characteristics and harvest rates of bushmeat species in afrotropical forests. *Biological Conservation* **121**: 167-176.

Fa, J. E., Seymour, S., Dupain, J., Amin, R., Albrechtsen, L., & Macdonald, D. 2006. Getting to grips with the magnitude of exploitation: Bushmeat in the Cross-Sanaga rivers region, Nigeria and Cameroon. *Biological Conservation* **129**: 497-510.

Fabricant, D. S., & Farnsworth, N. R. 2001. The value of plants used in traditional medicine for drug discovery. *Environmental Health Perspectives* **109**: 69–75.

FACE (Federation of Associations for Hunting and Conservation of the EU). 2007. *Census of the number of hunters in Europe*. Brussels, Belgium: FACE: Federation of Associations for Hunting and Conservation of the EU. Retrieved April 25, 2008, from http://www.face-europe.org/fs-hunting.htm

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics* **34**: 487-515.

FAO (Food and Agriculture Organization) & CIFOR (Center for International Forestry Research). 2005. *Forests and floods. Drowning in fiction or thriving on facts?* Bogor Barat, Indonesia: Center for International Forestry Research and Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 1999. *Sustaining agricultural biodiversity and agro-ecosystem functions: Opportunities, incentives and approaches for the conservation and sustainable use of agricultural biodiversity in agro-ecosystems and production systems*. Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 1997. *The State of the World's Plant Genetic Resources for Food and Agriculture*. Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 2001. *Global Forest Resources Assessment 2000*. Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 2005. *Trends in Wood Products 1961-2003*. Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 2006a. *Global Forest Resources Assessment 2005: Progress Towards Sustainable Forest Management*. Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 2006b. *The State of the World's Fisheries and Aquaculture 2006*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO).

FAO (Food and Agriculture Organization). 2007a. *The State of the World's Animal Genetic Resources for Food and Agriculture* (B. Rischkowsky & D. Pilling, Eds.). Rome, Italy: Food and Agriculture Organization of the United Nations.

FAO (Food and Agriculture Organization). 2007b. *The World's Mangroves 1980-2005*. Rome, Italy: Food and Agriculture Organization of the United Nations (FAO).

Ferreras, P., Gaona, P., Palomares, F., Delibes, M., 2001. Restore habitat or reduce mortality? implications from a population viability analysis of the Iberian lynx. *Animal Conservation* **4**: 265-274.

Finlayson, C. M., D'Cruz, R., Aladin, N., Barker, D. R., Beltram, G., Brouwer, J., Davidson, N., Duker, L., Junk, W., Kaplowitz, M. D., Ketelaars, H., Kreuzberg-Mukhina, E., Espino, G. D. L. L., Leveque, C., Lopez, A., et al. 2005. Inland Water Systems. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 551-583).

Fischer G, Shah M, van Velthuizen H,Nachtergaele FO. 2002. *Global Agroecological Assessment for Agriculture in the 21st Century: Methodology and Results*. Laxenburg, Austria: International Institute for Applied Systems Analysis.

Fischer, G., van Velthuizen, H., Nachtergaele, F., & Medow, S. 2000. *Global Agro-Ecological Zoning (GAEZ) 2000 CD-ROM*. FAO/IIASA.

Fisher B., Turner R.K., Balmford A., Burgess N.D., Green R., Kajembe G., Kulindwa K., Lewis S., Marchant R., Morse-Jones S., Naidoo R., Paavola J., Ricketts T. & Rouget M. (in review) Valuing the Arc: an ecosystem services approach for integrating natural systems and human welfare in the Eastern Arc Mountains of Tanzania.

Fisher, B., Costanza, R., Turner, R. K., & Morling, P. *in press*-a Defining and classifying ecosystem services for decision making. *Ecological Economics*.

Fisher, B., Turner, K., Zylstra, M., Brouwer, R., de Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., et al. *in press*-b Integrating Ecosystem Services and Economic Theory: what can we do, what should we do, and what has been done? *Ecological Applications*.

Fjørtoft, I. 2001. The Natural Environment as a Playground for Children: The Impact of Outdoor Play Activities in Pre-Primary School Children. *Early Childhood Education Journal* **29**: 111-117.

Foahom, B. 2002. *Insect pest incidence on timber tree species in natural forest in south Cameroon*. Kiribi, Cameroon: The Tropenbos-Cameroon Programme. Online: http://www.tropenbos.nl/files/camdoc12.pdf

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., et al. 2005. Global consequences of land use. *Science* **309**: 570-574.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., & Holling, C. S. 2004. Regime shifts, Resilience and Biodiversity in Ecosystem Management. *Annual Review of Ecology, Evolution and Systematics* **35**: 557-581.

Forest, F., Grenyer, R., Rouget, M., Davies, T. J., Cowling, R. M., Faith, D. P., Balmford, A., Manning, J. C., Proches, S., van der Bank, M., Reeves, G., Hedderson, T. A. J., & Savolainen, V. 2007. Preserving the evolutionary potential of floras in biodiversity hotspots. *Nature* **445**: 757-760.

Fosså, J. H., Mortensen, P. B., & Furevik, D. M. 2002. The deep-water coral Lophelia pertusa in Norwegian waters: distribution and fishery impacts. *Hydrobiologia* **471**: 1-12.

Fragoso, J. M. V. 1997. Tapir-generated seed shadows: scale-dependent patchiness in the Amazon rain forest. *The Journal of Ecology* **85**: 519-529.

Fragoso, J. M. V., Silvius, K. M., & Correa, J. A. 2003. Long-distance seed dispersal by tapirs increases seed survival and aggregates tropical trees. *Ecology* **84**: 1998-2006.

Frank, K. T., Petrie, B., Choi, J. S., & Leggett, W. C. 2005. Trophic cascades in a formerly cod-dominated ecosystem. *Science* **308**: 1621-1623.

Fredericksen, T., Justiniano, M., Mostacedo, B., Kennard, D., & McDonald, L. 2000. Comparative regeneration ecology of three leguminous timber species in a Bolivian tropical dry forest. *New Forests* **20**: 45-64.

Freeman, A. M. I. 2003. *The Measurements of Environmental and Resource Values: Theory and Methods*. RFF Press.

Freiwald, A., Fosså, J. H., Grehan, A., Koslow, T., Roberts, J. M. 2004. *Cold-water coral reefs: out of sight - no longer out of mind*. Cambridge, UK: UNEP-WCMC.

Fromentin, J., & Powers, J. E. 2005. Atlantic bluefin tuna: population dynamics, ecology, fisheries and management. *Fish and Fisheries* **6**: 281-306.

Gardner, T. A., Cote, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. 2003. Long-term region-wide declines in Caribbean corals. *Science* **301**: 958-960.

Gascon, C., Williamson, G. B., & da Fonseca, G. A. B. 2000. Receding forest edges and vanishing reserves. *Science* **288**: 1356-1358.

Gerten, D., Schaphoff, S., Haberlandt, U., Lucht, W., & Sitch, S. 2004. Terrestrial vegetation and water balance--hydrological evaluation of a dynamic global vegetation model. *Journal of Hydrology* **286**: 249-270.

Ghazoul, J. 2005. Buzziness as usual? Questioning the global pollination crisis. *Trends in Ecology & Evolution* **20**: 367-373.

Gilbert, F. F. 2000. Considerations in managing wildlife populations for sport. *Wildlife Society Bulletin* **28**: 459-463.

Giles-Corti, B., Broomhall, M. H., Knuiman, M., Collins, C., Douglas, K., Ng, K., Lange, A., & Donovan, R. J. 2005. Increasing walking: How important is distance to, attractiveness, and size of public open space? *American Journal of Preventive Medicine* **28**: 169-176.

Giller, K. & Palm, C. 2004. Cropping systems: slash-and-burn cropping systems of the tropics. In *Encyclopedia of Plant and Crop Science*. UK: CPL Scientific Publishing Services (pp. 363–366).

Giller, K. E., Beare, M. H., Lavelle, P., Izac, A. -. N., & Swift, M. J. 1997. Agricultural intensification, soil biodiversity and agroecosystem function. *Applied Soil Ecology* **6**: 3-16.

Goodwin, H, Kent, I, Parker, K & Walpole, M. 1998. Tourism, conservation and sustainable development: case studies from Africa and Asia. IIED, London, UK.

Goodwin, H. & Leader-Williams, N. 2000. Tourism and Protected Areas - distorting conservation priorities towards, charismatic megafauna? Chapter 14 in Entwistle, A & Dunstone N *Priorities for the conservation of mammalian biodiversity*. Cambridge University Press, Cambridge, UK.

Goodwin, H.J. 1996. In pursuit of ecotourism. *Biodiversity & Conservation* **5**: 277-291.

Gordon, H., Haygarth, P. M., & Bardgett, R. D. 2008. Drying and rewetting effects on soil microbial community composition and nutrient leaching. *Soil Biology and Biochemistry* **40**: 302-311.

Gordon, L. J., Peterson, G. D., & Bennett, E. M. 2008. Agricultural modifications of hydrological flows create ecological surprises. *Trends in Ecology & Evolution* **23**: 211-219.

Gordon, L. J., Steffen, W., Jonsson, B. F., Folke, C., Falkenmark, M., & Johannessen, A. 2005. Human modification of global water vapor flows from the land surface. *Proceedings of the National Academy of Sciences* **102**: 7612-7617.

Gossling, S. 1999. Ecotourism: a means to safeguard biodiversity and ecosystem functions? *Ecological Economics* **29**: 303-320.

Goudarzi, S. 2006. Ape Meat Sold in U.S., European Black Markets. *National Geographic News*: July 18, 2006. [Online] http://news.nationalgeographic.com/news/2006/07/060718-ape-meat.html.

Goulson, D., Hanley, M., Darvill, B., Ellis, J., & Knight, M. 2005. Causes of rarity in bumblebees. *Biological Conservation* **122**: 1-8.

Goulson, D., Lye, G. C., & Darvill, B. 2008. Decline and conservation of bumble bees. *Annual review of entomology* **53**: 191-208.

Grainger, A. 2008. Difficulties in tracking the long-term global trend in tropical forest area. *Proceedings of the National Academy of Sciences USA* **105**: 818-823.

Gram, S. 2001. Economic valuation of special forest products: an assessment of methodological shortcomings. *Ecological Economics* **36**: 109-117.

Green, E. P., & Short, F. T. (Eds.). 2003. *World Atlas of Seagrasses*. Berkley, CA, USA: University of California Press.

Green, R. E., Cornell, S. J., Scharlemann, J. P., Balmford, A., 2005. Farming and the fate of wild nature. *Science* **307**: 550-555.

Greenleaf, S. S., & Kremen, C. 2006. Wild bees enhance honey bees' pollination of hybrid sunflower. *Proceedings of the National Academy of Sciences USA* **103**: 13890-13895.

Griffiths, B. S., Ritz, K., Bardgett, R. D., Cook, R., Christensen, S., Ekelund, F., Sørensen, S. J., Bååth, E., Bloem, J., Ruiter, P. C. D., Dolfing, J., & Nicolardot, B. 2000. Ecosystem response of pasture soil communities to fumigation-induced microbial diversity reductions: an examination of the biodiversity - ecosystem function relationship. *Oikos* **90**: 279-294.

Groot, R. D., Ramakrishnan, P., Berg, A. V. D., Kulenthran, T., Muller, S., Pitt, D., Wascher, D., Wijesuriya, G., Amelung, B., Eliezer, N., Gopal, A. R., & Rossler, M. 2005. Cultural and Amenity Services. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends. Washington* D.C., USA: World Resources Institute (pp. 457-474).

Grosso, F., Jones, R. L., Demetri, G. D., Judson, I. R., Blay, J., Le Cesne, A., Sanfilippo, R., Casieri, P., Collini, P., Dileo, P., Spreafico, C., Stacchiotti, S., Tamborini, E., Tercero, J. C., Jimeno, J., et al. 2007. Efficacy of trabectedin (ecteinascidin-743) in advanced pretreated myxoid liposarcomas: a retrospective study. *The Lancet Oncology* **8**: 595-602.

Gruber, U., & Bartelt, P. 2007. Snow avalanche hazard modelling of large areas using shallow water numerical methods and GIS. *Environmental Modelling & Software* **22**: 1472-1481.

Guenni, L. B., Cardoso, M., Goldammer, J., Hurtt, G., Mata, L. J., Ebi, K., House, J., Valdes, J., & Norgaa, R. 2005. Regulation of Natural Hazards: Floods and Fires. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 441-454).

Guidetti, P. & Sala, E. 2007. Community-wide effects of marine reserves in the Mediterranean Sea. *Marine Ecology Progress Series* **335**: 43-56.

Gullström M, de la Torre C. M., Bandeira S.O., Björk M., Dahlberg M., Kautsky, N., Rönnbäck, P. and Öhman M. C. 2002. Seagrass Ecosystems in the Western Indian Ocean. *AMBIO* **31**: 588–596.

Halls A. & Welcomme R.L. 2004 Dynamics of river fish populations in response to hydrological conditions: A simulation study. *River Research and Applications* **20**: 985-1000.

Halpern, B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? *Ecological Applications* **13**: 117.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., et al. 2008. A global map of human impact on marine ecosystems. *Science* **319**: 948-952.

Hamilton, A. 2004. Medicinal plants, conservation and livelihoods. *Biodiversity and Conservation* **13**: 1477-1517.

Hammond D. S, Gourlet-Fleury S, van der Hout P, ter Steege H, & Brown V. K. 1996. A compilation of known Guianan timber trees and the significance of their dispersal mode, seed size and taxonomic affinity to tropical rain forest management. *Forest Ecology and Management* **83**: 99-116.

Hawkins, B. 2008. *Plants for life: Medicinal plant conservation and botanic gardens*. Richmond: Botanic Gardens Conservation International.

Haynes, R. 2003. *An analysis of the timber situation in the United States: 1952 to 2050*. Portland, OR, USA: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Heal, G. 2000. Valuing Ecosystem Services. *Ecosystems* **3**: 24-30.

Heard, M. S., Carvell, C., Carreck, N. L., Rothery, P., Osborne, J. L., & Bourke, A. F. G. 2007. Landscape context not patch size determines bumble-bee density on flower mixtures sown for agri-environment schemes. *Biology Letters* **3**: 638-41.

Heberlein, T., Ericsson, G., & Wollscheid, K. 2002. Correlates of hunting participation in Europe and North America. *Zeitschrift für Jagdwissenschaft* **48**: 320-326.

Hedenstrom, A., Johansson, L. C., Wolf, M., von Busse, R., Winter, Y., & Spedding, G. R. 2007. Bat flight generates complex aerodynamic tracks. *Science* **316**: 894-897.

Henwood, K. 2003. *Is there a role for environmental and countryside agencies in promoting benefits to health?* London, UK: NHS National Institute for Health and Clinical Excellence.

Hiddink, J., MacKenzie, B., Rijnsdorp, A., Dulvy, N., Nielsen, E., Bekkevold, D., Heino, M., Lorance, P., & Ojaveer, H. in press. Importance of fish biodiversity for the management of fisheries and ecosystems. *Fisheries Research*.

Hijmans, R., Cameron, S., Parra, J., Jones, P., & Jarvis, A. 2005. Very high resolution interpolated climate surfaces for global land areas. *International Journal of Climatology* **25**: 1965-1978.

Hilborn, R., Quinn, T. P., Schindler, D. E., & Rogers, D. E. 2003a. Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences USA* **100**: 6564-6568.

Hilborn, R., Trevor A. Branch, Billy Ernst, Arni Magnusson, Carolina V. Minte-Vera, Mark D. Scheuerell, A., & Juan L. Valero. 2003b. State of the world's fisheries. *Annual Review of Environment and Resources* **28**: 359-399.

Hill, J. O., & Peters, J. C. 1998. *Environmental Contributions to the Obesity Epidemic. Science* **280**: 1371-1374.

Hitzfeld, B. C., Höger, S. J., & Dietrich, D. R. 2000. Cyanobacterial toxins: removal during drinking water treatment, and human risk assessment. *Environmental Health Perspectives* **108**: 113-122.

Hoegh-Guldberg, O., Mumby, P. J., Hooten, A. J., Steneck, R. S., Greenfield, P., Gomez, E., Harvell, C. D., Sale, P. F., Edwards, A. J., Caldeira, K., Knowlton, N., Eakin, C. M., Iglesias-Prieto, R., Muthiga, N., Bradbury, R. H., et al. 2007. Coral reefs under rapid climate change and ocean acidification. *Science* **318**: 1737-1742.

Hofer, D. 2002. *The Lion's Share of the Hunt. Trophy Hunting and Conservation: a Review of the Legal Eurasian Tourist Hunting Market and Trophy Trade Under CITES*. Brussels, Belgium: TRAFFIC Europe.

Holzschuh, A., Steffan-Dewenter, I., & Tscharntke, T. 2008. Agricultural landscapes with organic crops support higher pollinator diversity. *Oikos* **117**: 354-361.

Holzschuh, A., Steffan-Dewenter, I., Kleijn, D., & Tscharntke, T. 2007. Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. *Journal of Applied Ecology* **44**: 41-49.

Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A. J., Vandermeer, J., & Wardle, D. A. 2005. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs* **75**: 3-35.

House, J. I., Prentice, C. I., Le Quéré, C., 2002. Maximum impacts of future reforestation or deforestation on atmospheric CO_2. *Global Change Biology* **8**: 1047-1052

House, J., Brovkin, V., Betts, R., Constanza, B., Dias, M. A. S., Holland, B., Quere, C. L., Phat, N. K., Riebesell, U., Scholes, M., Arneth, A., Barratt, D., Cassman, K., Christensen, T., Cornell, S., et al. 2005. Climate and Air Quality. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 355-390).

Hughes, B. B., Irfan, M. T., 2007. Assessing strategies for reducing poverty. *The International Studies Review* **9**: 690-710.

Hughes, B. B., 2001. Choices in the face of uncertainty: the international futures (IFs) model. *Futures* 33 (1), 55-62.

Hughes, T. P., Bellwood, D. R., Folke, C., Steneck, R. S., & Wilson, J. 2005. New paradigms for supporting the resilience of marine ecosystems. *Trends in Ecology & Evolution* **20**: 380-386.

Hunt, C. 2006. *An economic assessment of present and alternative arrangements for the management of PNG forests*. Papua New Guinea Eco-Forestry Forum. Online: http://www.ecoforestry.org.pg/publications/Wasting%20Our%20Heritage.pdf.

Hutchings, J. A. & Reynolds, J. D. 2004 Marine fish population collapses: consequences for recovery and extinction risk. *BioScience*. **54:** 297–309.

Hutchings, J. A. 2000. Collapse and recovery of marine fishes. *Nature* **406**: 882-885.

IPCC. 2001. *IPCC Special Report on Emissions Scenarios*. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

IPCCWGI. 2007. *Climate Change 2007 - The Physical Science Basis. Working Group I Contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK: Cambridge University Press.

IPCCWGII. 2007. *Climate Change 2007 – Impacts, Adaptation and Vulnerability. Working Group II contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, UK: Cambridge University Press.

IPCCWGIII. 2007. *Climate Change 2007 - Mitigation of Climate Change. Working Group III Contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK: Cambridge University Press.

Iriarte, J. A., & Jaksic, F. M. 1986. The fur trade in Chile: An overview of seventy-five years of export data (1910-1984). *Biological Conservation* **38**: 243-253.

Isaac, N. J., Turvey, S., Collen, B., Waterman, C., & Baillie, J. 2007. Mammals on the EDGE: Conservation Priorities Based on Threat and Phylogeny. *PLoS Biology* **2**: e296.

UNISDR (United Nations International Strategy for Disaster Reduction). 2006. The 2004 Indian Ocean Tsunami: One year after. *Disaster Reduction in Asia Pacific - ISDR Informs* **2**: 47-68.

Islam, M., & Haque, M. 2004. The mangrove-based coastal and nearshore fisheries of Bangladesh: ecology, exploitation and management. *Reviews in Fish Biology and Fisheries* **14**: 153-180.

ITTO (International Tropical Timber Organization). 2006a. ITTO Objective 2000. Online: http://www.itto.or.jp/live/PageDisplayHandler?pageId=5.

ITTO (International Tropical Timber Organization). 2006b. *Status of Tropical Forest Management - 2005 Summary Report*. Yokohama, Japan: International Tropical Timber Organization. Online: http://www.itto.or.jp/live/Live_Server/1222/ITTOSFMTropics2005_summary.e.pdf

ITTO (International Tropical Timber Organization). 2007. *2007 Annual Review and Assessment of the World Timber Situation*. Yokohama, Japan: International Tropical Timber Organization. Online: http://www.itto.or.jp/live/Live_Server/377/E-AR07-text.pdf

IUCN (The World Conservation Union). 2007. IUCN Red List of Threatened Species. *IUCN - The World Conservation Union*. Online: http://www.iucnredlist.org/

Iverson, L., & Prasad, A. 2007. Using landscape analysis to assess and model tsunami damage in Aceh province, Sumatra. *Landscape Ecology* **22**: 323-331.

Jackson, E., Rowden, A., Attrill, M., Bossey, S., & Jones, M. 2001a. The importance of seagrass beds as a habitat for fishery species. *Oceanography and Marine Biology* **39**: 269-303.

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., et al. 2001b. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. *Science* **293**: 629-637.

Jacobs. 2004. *An Economic Assessment of the Costs and Benefits of Natura 2000 Sites in Scotland 2004*. Edinburgh, UK: Scottish Executive.

Jalale, T. 1993. *National Parks Tourism in Malawi*. MPhil thesis. Canterbury, Kent: University of Kent.

Jambiya, G., Milledge , S., & Mtango, N. 2007. *'Night Time Spinach' Conservation and Livelihood implications of wild meat meat use in refugee situations in north-western Tanzania*. Daar es Salaam, Tanzania: TRAFFIC.

Jansen, P., & Zuidema, P. 2001. Logging, seed dispersal by vertebrates, and natural regeneration of tropical timber trees. In *The Cutting Edge: Conserving Wildlife in Logged Tropical Forests.* . New York, NY, USA: Columbia University Press (pp. 35-59).

Jennings, S. B., Brown, N. D., Boshier, D. H., Whitmore, T. C., & Lopes, J. D. C. A. 2001. Ecology provides a pragmatic solution to the maintenance of genetic diversity in sustainably managed tropical rain forests. *Forest Ecology and Management* **154**: 1-10.

Jeong, K., Kim, J., & Lee, L. P. 2006. Biologically Inspired Artificial Compound Eyes. *Science* **312**: 557-561.

Jerozolimski, A., & Peres, C. A. 2003. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. *Biological Conservation* **111**: 415-425.

Johansson, D., Azar, C., June 2007. A scenario based analysis of land competition between food and bioenergy production in the US. *Climatic Change* **82:** 267-291.

Johnson, R., 2007. *Recent Honey Bee Colony Declines*. Congressional Research Service Report for Congress. RL 33938. Washington DC, USA.

Kannan, N., White, S., & Whelan, M. 2007. Predicting diffuse-source transfers of surfactants to surface waters using SWAT. *Chemosphere* **66**: 1336-1345.

Kannan, N., White, S., Worrall, F., & Whelan, M. 2007. Hydrological modelling of a small catchment using SWAT-2000 - Ensuring correct flow partitioning for contaminant modelling. *Journal of Hydrology* **334**: 64-72.

Karasov, C. 2001. Who reaps the benefits of biodiversity? *Environmental Health Perspectives* **109**: A582-A587.

Karesh, W. B., Cook, R. A., Bennett, E. L., & Newcomb, J. 2005. Wildlife trade and global disease emergence. *Emerging Infectious Diseases* **11**: 1000-1002.

Kathe, W. 2006. Revision of the 'Guidelines on the conservation of medicinal plants' by WHO, IUCN, WWF and TRAFFIC. In *Medicinal and Aromatic Plants*. The Netherlands: Springer (pp. 109-120).

Kathiresan, K., & Rajendran, N. 2005. Coastal mangrove forests mitigated tsunami. *Estuarine, Coastal and Shelf Science* **65**: 601-606.

Kean, J., Wratten, S., Tylianakis, J., & Barlow, N. 2003. The population consequences of natural enemy enhancement, and implications for conservation biological control. *Ecology Letters* **6**: 604-612.

Kearns, C. A., Inouye, D. W., & Waser, N. M. 1998. Endangered Mutualisms: The Conservation of Plant-Pollinator Interactions. *Annual Review of Ecology and Systematics* **29**: 83-112.

Kerr, A. M., & Baird, A. H. 2007. Natural Barriers to Natural Disasters. *BioScience* **57**: 102-103.

Kerr, A. M., Baird, A. H., & Campbell, S. J. 2006. Comments on "Coastal mangrove forests mitigated tsunami" by K. Kathiresan and N. Rajendran [Estuar. Coast. Shelf Sci. 65 (2005) 601-606]. *Estuarine, Coastal and Shelf Science* **67**: 539-541.

Kier, G., Mutke, J., Dinerstein, E., Ricketts, T. H., Kuper, W., Kreft, H., & Barthlott, W. 2005. Global patterns of plant diversity and floristic knowledge. *Journal of Biogeography* **32**: 1107-1116.

Kindermann, G. E., Obersteiner, M., Rametsteiner, E., McCallum, I., 2006. Predicting the deforestation-trend under different carbon-prices. *Carbon Balance and Management* **1**: 15.

Klein, A., Cunningham, S. A., Bos, M. & Steffan-Dewenter, I. 2008. Advances in pollination ecology from tropical plantation crops. *Ecology* **89**: 935-943

Klein, A., Steffan-Dewenter, I., & Tscharntke, T. 2003a. Pollination of *Coffea canephora* in relation to local and regional agroforestry management. *Journal of Applied Ecology* **40**: 837-845.

Klein, A., Steffan-Dewenter, I., & Tscharntke, T. 2003b. Fruit set of highland coffee increases with the diversity of pollinating bees. *Proceedings of the Royal Society B* **270**: 955–961.

Klein, A., Steffan-Dewenter, I., & Tscharntke, T. 2003c. Bee pollination and fruit set of *Coffea arabica* and *C. canephora* (Rubiaceae). *Am. J. Bot.* **90**: 153-157.

Klein, A., Steffan-Dewenter, I., Buchori, D., & Tscharntke, T. 2002. Effects of Land-Use Intensity in Tropical Agroforestry Systems on Coffee Flower-Visiting and Trap-Nesting Bees and Wasps. *Conservation Biology* **16**: 1003-1014.

Klein, A., Vaissière, B. E., Cane, J. H., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., & Tscharntke, T. 2007. Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society of London* B **274**: 303-313.

Kleinman, P. J. A., Pimentel, D., & Bryant, R. B. 1995. The ecological sustainability of slash-and-burn agriculture. *Agriculture, Ecosystems & Environment* **52**: 235-249.

Kowalewski, M., Serrano, G. E. A., Flessa, K. W., & Goodfriend, G. A. 2000. Dead delta's former productivity: Two trillion shells at the mouth of the Colorado River. *Geology* **28**: 1059-1062.

Kremen, C. 2005. Managing ecosystem services: what do we need to know about their ecology? *Ecology Letters* **8**: 468-479.

Kremen, C., & Chaplin-Kramer, R. 2007. Insects as providers of ecosystem services: crop pollination and pest control. In Stewart, A.J.A., New, T.R., & Lewis, O.T. (Eds.), *Insect Conservation Biology: proceedings of the Royal Entomological Society's 23rd Symposium*. Wallingford, UK: CABI Publishing (pp. 349-382).

Kremen, C., Daily, G. C., Klein, A., & Scofield, D. 2008. Inadequate assessment of the ecosystem service rationale for conservation: a reply to Ghazoul. *Conservation Biology* **22**: 795-798

Kremen, C., Williams, N. M., & Thorp, R. W. 2002. Crop pollination from native bees at risk from agricultural intensification. *Proceedings of the National Academy of Sciences USA* **99**: 16812-16816.

Kremen, C., Williams, N. M., Aizen, M. A., Gemmill-Herren, B., LeBuhn, G., Minckley, R., Packer, L., Potts, S. G., Roulston, T., Steffan-Dewenter, I., Vazquez, D. P., Winfree, R., Adams, L., Crone, E. E., Greenleaf, S. S., et al. 2007. Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. *Ecology Letters* **10**: 299-314.

Kremen, C., Williams, N. M., Bugg, R. L., Fay, J. P., & Thorp, R. W. 2004. The area requirements of an ecosystem service: crop pollination by native bee communities in California. *Ecology Letters* **7**: 1109-1119.

Kruess, A., & Tscharntke, T. 1994. Habitat fragmentation, species loss, and biological control. *Science* **264**: 1581-1584.

Kunkel, C. M., Hallberg, R. W., & Oppenheimer, M. 2006. Coral reefs reduce tsunami impact in model simulations. *Geophysical Research Letters* **33**: L23612.

Lamoreux, J. F., Morrison, J. C., Ricketts, T. H., Olson, D. M., Dinerstein, E., McKnight, M. W., & Shugart, H. H. 2006. Global tests of biodiversity concordance and the importance of endemism. *Nature* **440**: 212-214.

Landis, D. A., Wratten, S. D., & Gurr, G. M. 2000. Habitat management to conserve natural enemies of arthropod pests in agriculture. *Annual Review of Entomology* **45**: 175-201.

Larsen, T. H., Williams, N. M., & Kremen, C. 2005. Extinction order and altered community structure rapidly disrupt ecosystem functioning. *Ecology Letters* **8**: 538-547.

Laurance, W. F. 2007. Environmental science: Forests and floods. *Nature* **449**: 409-410.

Laurance, W., Nascimento, H., Laurance, S., Andrade, A., Ribeiro, J., Giraldo, J., Lovejoy, T., Condit, R., Chave, J., Harms, K., & D'Angelo, S. 2006. Rapid decay of tree-community composition in Amazonian forest fragments. *Proceedings of the National Academy of Sciences USA* **103**: 19010–19014.

Laurance, W., Salicrup, P., Delamônica, P., Fearnside, P., D'Angelo, S., Jerozolinksi, A., Pohl, L., & Lovejoy, T. 2001. Rain forest fragmentation and the structure of Amazonian liana communities. *Ecology* **82**: 105-116.

Lawton, R. O., Nair, U. S., Pielke, R. A., & Welch, R. M. 2001. Climatic impact of tropical lowland deforestation on nearby montane cloud forests. *Science* **294**: 584-587.

Le Quere, C., Rodenbeck, C., Buitenhuis, E. T., Conway, T. J., Langenfelds, R., Gomez, A., Labuschagne, C., Ramonet, M., Nakazawa, T., Metzl, N., Gillett, N., & Heimann, M. 2007. Saturation of the Southern Ocean CO_2 sink due to recent climate change. *Science* **316**: 1735-1738.

Lee, H., Lee, B. P., & Messersmith, P. B. 2007a. A reversible wet/dry adhesive inspired by mussels and geckos. *Nature* **448**: 338-341.

Lee, S., Ryu, J., & Kim, I. 2007b. Landslide susceptibility analysis and its verification using likelihood ratio, logistic regression, and artificial neural network models: case study of Youngin, Korea. *Landslides* **4**: 327-338.

Lehner, B., & Döll, P. 2004. Development and validation of a global database of lakes, reservoirs and wetlands. *Journal of Hydrology* **296**: 1-22.

Liang, X., Lettenmaier, D. P., Wood, E. F., & Burges, S. J. 1994. A simple hydrologically based model of land surface water and energy fluxes for general circulation models. *Journal of Geophysical Research* **99**: 14415–14428.

Lindsey, P. A., Alexander, R., Frank, L. G., Mathieson, A., & Romanach, S. S. 2006. Potential of trophy hunting to create incentives for wildlife conservation in Africa where alternative wildlife-based land uses may not be viable. *Animal Conservation* **9**: 283-291.

Lindsey, P., Roulet, P., & Romanach, S. 2007. Economic and conservation significance of the trophy hunting industry in sub-Saharan Africa. *Biological Conservation* **134**: 455-469.

Liu, P. L., Lynett, P., Fernando, H., Jaffe, B. E., Fritz, H., Higman, B., Morton, R., Goff, J., & Synolakis, C. 2005. Observations by the international tsunami survey team in Sri Lanka. *Science* 308: 1595.

Lobovikov, M., Paudel, S., Piazza, M., Ren, H., & Wu, J. 2007. *Global Bamboo Resources: a Thematic Study Prepared in the Framework of the Global Forest Resources Assessment 2005*. Rome, Italy: Food and Agriculture Organization of the United Nations and International Network for Bamboo and Rattan.

Losey, J. E., & Vaughan, M. 2006. The economic value of ecological services provided by insects. *BioScience* **56**: 311-323.

Lynett, P. J. 2007. Effect of a shallow water obstruction on long wave runup and overland flow velocity. *Journal of Waterway, Port, Coastal, and Ocean Engineering* **133**: 455-462.

MacArthur, R. H., & Wilson, E. O. 1967. *The Theory of Island Biogeography*. New Jersey: Princeton.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., & Niggli, U. 2002. Soil fertility and biodiversity in organic farming. *Science* **296**: 1694-1697.

Mahdavi, A., Ferreira, L., Sundback, C., Nichol, J. W., Chan, E. P., Carter, D. J. D., Bettinger, C. J., Patanavanich, S., Chignozha, L., Ben-Joseph, E., Galakatos, A., Pryor, H., Pomerantseva, I., Masiakos, P. T., Faquin, W., et al. 2008. A biodegradable and biocompatible gecko-inspired tissue adhesive. *Proceedings of the National Academy of Sciences USA* **105**: 2307-2312.

Malakoff, D. 1998. Coastal Ecology: Death by suffocation in the Gulf of Mexico. *Science* **281**: 190-192.

Maxted, N. 2003. Conserving the genetic resources of crop wild relatives in European Protected Areas. *Biological Conservation* **113**: 411-417.

Maxted, N., Scholten, M., Codd, R., & Ford-Lloyd, B. 2007. Creation and use of a national inventory of crop wild relatives. *Biological Conservation* **140**: 142-159.

Mayaux P., Holmgren P., Achard F., Eva H., Stibig H.-J. & Branthomme A. 2005. Tropical forest cover change in the 1990s and options for future monitoring. *Philosophical Transactions of the Royal Society of London* B **360**: 373-384.

Mazda, Y., Magi, M., Ikeda, Y., Kurokawa, T., & Asano, T. 2006. Wave reduction in a mangrove forest dominated by Sonneratia sp. *Wetlands Ecology and Management* **14**: 365-378.

McArthur L. C, Boland J. W, Brown B, & Jones G. K. 2003. Investigation of the influence of seagrass on fisheries catch level using isotonic regression. *Environmental Modeling and Assessment* **8**: 285-290.

McCorquodale, S. M. 1997. Cultural contexts of recreational hunting and native subsistence and ceremonial hunting: their significance for wildlife management. *Wildlife Society Bulletin* **25**: 568-573.

McCullough, D. R. 1996. Spatially structured populations and harvest theory. *The Journal of Wildlife Management* **60**: 1-9.

McGuire, A. D., Sitch, S., Clein, J. S., Dargaville, R., Esser, G., et al. 2001. Carbon balance of the terrestrial biopshere in the twentieth century: Analyses of CO2, climate and land use effects with four process-based ecosystem models. *Global Biogeochem Cycles* **15:** 183-206.

McIntyre, P. B., Jones, L. E., Flecker, A. S., & Vanni, M. J. 2007. Fish extinctions alter nutrient recycling in tropical freshwaters. *Proceedings of the National Academy of Sciences* **104**: 4461-4466.

McIvor, A. 2004. Freshwater mussels as biofilters. PhD Thesis. Online: http://www.ourearth.co.uk/annamcivor/thesis.html.

McIvor, A., & Pungetti, G. 2008. Can sacradeness help protect species? *World Conservation* **January**: 18.

MEA (Millennium Ecosystems Assessment). 2005. *Ecosystems and Human Well-being: Synthesis Report*. Island Press.

Medicinal Plant Specialist Group. 2007. *International Standard for Sustainable Wild Collection of Medicinal and Aromatic Plants (ISSC-MAP)*. Bonn, Germany: Bundesamt für Naturschutz (BfN), MPSG/SSC/IUCN, WWF Germany, and TRAFFIC, Bonn, Gland, Frankfurt, and Cambridge.

Memmott, J., Craze, P. G., Waser, N. M., & Price, M. V. 2007. Global warming and the disruption of plant-pollinator interactions. *Ecology Letters* **10**: 710-717.

Merode, E., Homewood, K., & Cowlishaw, G. 2004. The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo. *Biological Conservation* **118**: 573-581.

Michalski, F., & Peres, C.A. 2007. Disturbance-mediated mammal persistence and abundance-area relationships in Amazonian forest fragments. *Conservation Biology* **21**: 1626-1640.

Micheli, F., & Halpern, B. S. 2005. Low functional redundancy in coastal marine assemblages. *Ecology Letters* **8**: 391-400.

Milner-Gulland, E. J., & Akçakaya, H. R. 2001. Sustainability indices for exploited populations. *Trends in Ecology & Evolution* **16**: 686-692.

Milner-Gulland, E. J., Bennett, E. L., Abernethy, K., Bakarr, M., Bodmer, R. E., Brashares, J. S., Cowlishaw, G., Elkan, P., Eves, H. E., Fa, J. E., Milner-Gulland, E. J., Peres, C., Roberts, C. M., Robinson, J. G., Rowcliffe, J. M., et al. 2003. Wild meat: the bigger picture. *Trends in Ecology & Evolution* **18**: 351-357.

Mitsch, W. J., Day, J. W., Gilliam, J. W., Groffman, P. M., Hey, D. L., Randall, G. W., & Wang, N. 2001. Reducing Nitrogen Loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to Counter a Persistent Ecological Problem. *BioScience* **51**: 373-388.

Mittermeier, R. A., Gil, P. R., Hoffman, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J., & Fonseca, G. A. B. D. 2004. *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions*. Washington, DC, USA: Conservation International.

Moberg, F., & Folke, C. 1999. Ecological goods and services of coral reef ecosystems. *Ecological Economics* **29**: 215-233.

Montag, J. M., Patterson, M. E. and Freimund, W. A. 2005. The wolf viewing experience in the Lamar Valley of Yellowstone National Park. *Human Dimensions of Wildlife* **10**: 273–84.

Montréal Proces. 2003. *First Forest Overvirw Report 2003*. Montréal Proces. Online: http://www.rinya.maff.go.jp/mpci/rep-pub/2003/contents_e.html

Moore, G. and Jennings, S. (eds.) 2000. *Commercial Fishing: The Wider Ecological Impacts*. London, UK: Blackwell Science.

Moran, D. 1994. Contingent valuation and biodiversity: measuring the user surplus of Kenyan protected areas. *Biodiversity & Conservation* **3**: 663-684.

Morandin, L. A., & Winston, M. L. 2006. Pollinators provide economic incentive to preserve natural land in agroecosystems. *Agriculture, Ecosystems & Environment* **116**: 289-292.

Morandin, L. A., Winston, M. L., Abbott, V. A., & Franklin, M. T. 2007. Can pastureland increase wild bee abundance in agriculturally intense areas? *Basic and Applied Ecology* **8**: 117-124.

Morato, T, Cheung, W. W. L., Pitcher, T. J. 2004. Vulnerability of Seamount Fish to Fishing: Fuzzy analysis of life history attributes. In T. Morato & D. Pauly (Eds) *Seamounts: Biodiversity and fisheries*. Vancouver, Canada: Fisheries Centre, University of British Columbia (pp. 52-60).

Morato, T. & Pauly, D. (eds). 2004. Seamounts: Biodiversity and fisheries. *Fisheries Centre Research Reports* **12**: 1-78.

Morato, T., Watson, R., Pitcher, T. and Pauly, D. 2006. Fishing down the deep. *Fish and Fisheries* 7: 24-34.

Moreira, F., Siqueira, J., & Brussaard, L. (Eds.). 2006. *Soil Biodiversity in Amazonian and Other Brazilian Ecosystems*. Wallingford, UK: CABI.

Morrissey, M. M., Wieczorek, G. F., & Morgan, B. A. 2008. A comparative analysis of simulated and observed landslide locations triggered by Hurricane Camille in Nelson County, Virginia. *Hydrological Processes* **22**: 524-531.

Msangi, S., Sulser, T., Rosegrant, M., Valmonte-Santos, R., 2007. Global scenarios for biofuels: Impacts and implications for food security and water use. *10th Annual Conference on Global Economic Analysis*, Purdue University, USA.

Muchaal, P. K. & Ngandjui, G. 1999. Impact of village hunting on wildlife populations in the Western Dja Reserve, Cameroon. *Conservation Biology* **13**: 385–396.

Muir, J. 1898. The wild parks and forest reservations of the west. *Atlantic Monthly* **81**: 15.

Mumby, P. J., Edwards, A. J., Ernesto Arias-Gonzalez, J., Lindeman, K. C., Blackwell, P. G., Gall, A., Gorczynska, M. I., Harborne, A. R., Pescod, C. L., Renken, H., C. C. Wabnitz, C., & Llewellyn, G. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature* **427**: 533-536.

Mumby, P. J., Hastings, A., & Edwards, H. J. 2007. Thresholds and the resilience of Caribbean coral reefs. *Nature* **450**: 98-101.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature* **403**: 853-858.

Myers, R. A., & Worm, B. 2003. Rapid worldwide depletion of predatory fish communities. *Nature* **423**: 280-283.

Myers, R. A., Baum, J. K., Shepherd, T. D., Powers, S. P., & Peterson, C. H. 2007. Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science* **315**: 1846-1850.

Nabhan, G. P., and S. L. Buchmann. 1997. Services provided by pollinators. Pages 133-150 in G. C. Daily, ed. *Nature's services*. Washington, D.C: Island Press.

Nabuurs, G. J., Päivinen, R., & Schanz, H. 2001. Sustainable management regimes for Europe's forests -- a projection with EFISCEN until 2050. *Forest Policy and Economics* **3**: 155-173.

Naidoo, R., & Adamowicz, W. L. 2005. Economic benefits of biodiversity exceed costs of conservation at an African rainforest reserve. *Proceedings of the National Academy of Sciences USA* **102**: 16712-16716.

Naidoo, R., & Ricketts, T. H. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biology* **4**: e360.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R. E., Lehner, B., Malcolm, T. R., & Ricketts, T. H. 2008. Global Mapping of Ecosystem Services and Conservation Priorities. *Proceedings of the National Academy of Sciences* **105**: 9495-9500.

Naiman, R. J., & Decamps, H. 1997. The ecology of interfaces: riparian zones. *Annual Review of Ecology and Systematics* **28**: 621-658.

Naiman, R. J., Melillo, J. M., & Hobbie, J. E. 1986. Ecosystem alteration of boreal forest streams by Beaver (Castor canadensis). *Ecology* **67**: 1254-1269.

Nakicenovic, N., Swart, R. (eds) 2000. *IPCC Special Report on Emissions Scenarios*. Cambridge University Press, Cambridge.

Naughton-Treves, L., Mena, J. L., Treves, A., Alvarez, N., & Radeloff, V. C. 2003. Wildlife Survival Beyond Park Boundaries: the Impact of Slash-and-Burn Agriculture and Hunting on Mammals in Tambopata, Peru. *Conservation Biology* **17**: 1106-1117.

Navrud, S., & Mungatana, E. D. 1994. Environmental valuation in developing countries: The recreational value of wildlife viewing. *Ecological Economics* **11**: 135-151.

Naylor, R. & Ehrlich, P. 1997. Natural pest control services and agriculture. In: Daily, G. (ed.) *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press, Washington (pp. 151-174).

Neigel, J. E. 2003. Species–Area Relationships and Marine Conservation. *Ecological Applications* **13**: 138.

Neiland, A., & Béné, C. (Eds.). 2008. *Tropical river fisheries valuation: background papers to a global synthesis*. Penang, Malaysia: WorldFish Center.

NERC (Natural Environmental Research Council), Coventry University, & University College London. 2000. The Tsunami Risks Project: Physics. Online: http://www.nerc-bas.ac.uk/tsunami-risks/html/Physics1.htm.

Newman, D., & Cragg, G. 2004. Marine natural products and related compounds in clinical and advanced preclinical trials. *Journal of Natural Products* **67**: 1216-1238.

Newman, D., & Cragg, G. 2007. Natural products as sources of new drugs over the last 25 years. *Journal of Natural Products* **70**: 461-477.

Newton, K., Côté, I.M., Pilling, G.M., Jennings, S., Dulvy, N.K. 2007 Current and future sustainability of island coral reef fisheries. *Current Biology* **17**:655-658.

NIES (National Institute for Environmental Studies). 2008. *The 13th AIM (Asia-Pacific Integrated Model) International Workshop*. 16-18 February 2008, Ohyama Memorial Hall, National Institute for Environmental Studies, Tsukuba, Japan.

Nijssen, B., Schnur, R., & Lettenmaier, D. P. 2001. Global Retrospective Estimation of Soil Moisture Using the Variable Infiltration Capacity Land Surface Model, 1980–93. *Journal of Climate* **14**: 1790-1808.

Njiru, M., Kazungu, J., Ngugi, C. C., Gichuki, J., & Muhoozi, L. 2008. An overview of the current status of Lake Victoria fishery: Opportunities, challenges and management strategies. *Lakes & Reservoirs: Research & Management* **13**: 1-12.

NOAA (National Oceanic and Atmospheric Administration). 2008. NOAA Pacific Tsunami Warning Center. Online: http://www.prh.noaa.gov/pr/ptwc/

Norton, B. 1992. Sustainability, Human Welfare and Ecosystem Health. *Environmental Values* **1**: 97-111.

Novaro, A. J., Funes, M., & Walker, R. S. 2005. An empirical test of source-sink dynamics induced by hunting. *Journal of Applied Ecology* **42**: 910-920.

Novaro, A. J., Redford, K. H., & Bodmer, R. E. 2000. Effect of hunting in source-sink systems in the Neotropics. *Conservation Biology* **14**: 713-721.

NRC (National Research Council). 2007. *Status of Pollinators in North America*. Washington D.C., USA: National Academies Press.

Ntiamoa-Baidu, Y. 1997. *Wildlife and food security in Africa*. Rome, Italy: Food and Agriculture Organization of the United Nations [online] http://www.fao.org/docrep/w7540e/w7540e00.htm#Contents.

Ohl-Schacherer, J., Shepard, G. H., Kaplan, H., Peres, C. A., Levi, T., & Yu, D. W. 2007. The sustainability of subsistence hunting by Matsigenka native communities in Manu National Park, Peru. *Conservation Biology* **21**: 1174-85.

Olsen, K. M., & Gross, B. L. 2008. Detecting multiple origins of domesticated crops. *Proceedings of the National Academy of Sciences USA* **105**: 13701-13702.

Olson, D., Dinerstein, E., Wikramanayake, E., Burgess, N., Powell, G., Underwood, E., D'amico, J., Itoua, I., Strand, H., Morrison, J., Loucks, C., Allnutt, T., Ricketts, T., Kura, Y., Lamoreux, J., et al. 2001. Terrestrial ecoregions of the world: a new map of life on earth. *BioScience* **51**: 933-938.

O'Reilly, C. M., Alin, S. R., Plisnier, P., Cohen, A. S., & McKee, B. A. 2003. Climate change decreases aquatic ecosystem productivity of Lake Tanganyika, Africa. *Nature* **424**: 766-768.

Orr, J. C., Fabry, V. J., Aumont, O., Bopp, L., Doney, S. C., Feely, R. A., Gnanadesikan, A., Gruber, N., Ishida, A., Joos, F., Key, R. M., Lindsay, K., Maier-Reimer, E., Matear, R., Monfray, P., et al. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature* **437**: 681-686.

Östman, Ö., Ekbom, B., & Bengtsson, J. 2003. Yield increase attributable to aphid predation by ground-living polyphagous natural enemies in spring barley in Sweden. *Ecological Economics* **45**: 149-158.

Palomares, F., Gaona, P., Ferreras, P., & Delibes, M. 1995. Positive effects on game species of top predators by controlling smaller predator populations: an example with lynx, mongooses, and rabbits. *Conservation Biology* **9**: 295-305.

Palumbi, S. R. 2001. Humans as the world's greatest evolutionary force. *Science* **293**: 1786-1790.

Pandolfi, J. M., Bradbury, R. H., Sala, E., Hughes, T. P., Bjorndal, K., Cooke, R. G., McArdle, D., McClenachan, L., Newman, M. J. H., Paredes, G., Warner, R. R., & Jackson, J. B. C. 2003. Global trajectories of the long-term decline of coral reef ecosystems. *Science* **301**: 955-958.

Pandolfi, J. M., Jackson, J. B. C., Baron, N., Bradbury, R. H., Guzman, H. M., Hughes, T. P., Kappel, C. V., Micheli, F., Ogden, J. C., Possingham, H. P., & Sala, E. 2005. Are U.S. Coral reefs on the slippery slope to slime? *Science* **307**: 1725-1726.

Parmesan, C., & Yohe, G. 2003. A globally coherent fingerprint of climate change impacts across natural systems. *Nature* **421**: 37-42.

Parry, M. L., Canziani, O. F., Palutikof, J. P., van der Linden, P. J., Hanson, C. E. (Eds.) 2007. *Climate Change 2007 - Impacts, Adaptation and Vulnerability: Working Group II contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge.

Paterson, I., & Anderson, E. A. 2005. The renaissance of natural products as drug candidates. *Science* **310**: 451-453.

Pauly, D. and Watson, R. 2005 Background and Interpretation of the 'Marine Trophic Index' as a measure of biodiversity. *Phil. Trans. Soc. B*, **360**, 415-423.

Pauly, D., Alder, J. Bakun, A., Heileman, S., Kock, K., Mace, P., Perrin, W., Stergiou, K., Sumaila, U. R., Vierros, M., Freire, K., Sadovy, Y., Christensen, V., Kaschner, K., Palomares, M., et al. 2005 Marine Fisheries Systems. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 745-794).

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P., & Watson, R. 2003. The future for fisheries. *Science* **302**: 1359-1361.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., & Torres, F. 1998. Fishing down marine food webs. *Science* **279**: 860-863.

Pawson, M. G., Tingley, D., & Glenn, H. 2004. *"Sport Fisheries" (or Marine Recreational Fisheries) in the EU*. Brussels, Belgium: The European Commission Directorate-General for Fisheries. Online: http://www.eu.nl/fisheries/publications/studies/sportfishing_2006.pdf

Pearce, D. W., & Turner, R. K. 1990. *Economics of Natural Resources and the Environment*. Hemel Hempstead, UK: Harvester/Wheatsheaf.

Pearce, D., & Moran, D. 1994. *The Economic Value of Biodiversity*. London, UK: Earthscan Publications Ltd.

Peres, C. A. & Dolman, P. M. 2000. Density compensation in neotropical primate communities: evidence from hunted and unhunted Amazonian forests of varying productivity. *Oecologia* **122**: 175–189.

Peres, C. A. 1999. Effects of subsistence hunting and forest types on Amazonian primate communities. In J. G. Fleagle, C. Janson & K. E. Reed (Eds) *Primate Communities*. Cambridge, UK: Cambridge University Press (pp. 268–283).

Peres, C. A. 2000. Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conservation Biology* **14**: 240-253.

Peres, C. A. 2001. Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. *Conservation Biology* **15**: 1490-1505.

Peres, C. A., & Palacios, E. 2007. Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: implications for animal-mediated seed dispersal. *Biotropica* **39**: 304-315.

Pergams, O. R. W., & Zaradic, P. A. 2008. Evidence for a fundamental and pervasive shift away from nature-based recreation. *Proceedings of the National Academy of Sciences USA* **105**: 2295-2300.

Perillo, G. M. E. (Ed.). 1997. *Evaluación de la vulnerabilidad de la costa argentina al ascenso del nivel del mar*. PNUD/SECYT ARG/95/G/31.

Petley, D., Hearn, G., Hart, A., Rosser, N., Dunning, S., Oven, K., & Mitchell, W. 2007. Trends in landslide occurrence in Nepal. *Natural Hazards* **43**: 23-44.

Petley, D.N., Dunning, S.A. & Rosser, N.J. 2005. The analysis of global landslide risk through the creation of a database of worldwide landslide fatalities. In Hungr, O., Fell, R., Counture, R. & Ebergardt, E. (Eds.) *Landslide Risk Management*. Amsterdam: Balkema (pp. 367-374).

Petrosillo, I., Zurlini, G., Corliano, M., Zaccarelli, N., & Dadamo, M. 2007. Tourist perception of recreational environment and management in a marine protected area. *Landscape and Urban Planning* **79**: 29-37.

Pimentel, D. 2008. Conservation biological control. *Biological Control* **45**: 171.

Pimentel, D., Berger, B., Filiberto, D., Newton, M., Wolfe, B., Karabinakis, E., Clark, S., Poon, E., Abbett, E., & Nandagopal, S. 2004. Water resources: agricultural and environmental issues. *BioScience* **54**: 909-918.

Pinet, J. 1995. *The hunter in Europe*. FACE: Federation of Associations for Hunting and Conservation of the EU.

Pitcher, T. J., Watson, R., Forrest, R., Valtysson, H. P., & Guenette, S. 2002. Estimating illegal and unreported catches from marine ecosystems: a basis for change. *Fish and Fisheries* **3**: 317-339.

Pitcher, T., & Cochrane, K. 2002. The use of ecosystem models to investigate multispecies management strategies for capture fisheries. *Fisheries Centre Research Reports* **10**: 1-115.

Plaganyi, E. E. 2007. *Models for an ecosystem approach to fisheries*. Rome: Food and Agriculture Organization of the United Nations.

Postel, S. 1999. *Pillar of Sand: Can the Irrigation Miracle Last*? New York, USA: W.W Norton & Co.

Postel, S. L., & Thompson, B. H. 2005. Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum* **29**: 98-108.

Potting, J., Bakkes, J. (eds) 2004. *The GEO-3 Scenarios 2002-2032 Quantification and Analysis of Environmental Impacts*. RIVM, Bilthoven, Netherlands.

Potts, S. G., Vulliamy, B., Roberts, S., O'Toole, C., Dafni, A., Ne'eman, G., & Willmer, P. 2005. Role of nesting resources in organising diverse bee communities in a Mediterranean landscape. *Ecological Entomology* **30**: 78-85.

Potvin, C., & Gotelli, N. J. 2008. Biodiversity enhances individual performance but does not affect survivorship in tropical trees. *Ecology Letters* **11**: 217-223.

Pregitzer, K. S., & Euskirchen, E. S. 2004. Carbon cycling and storage in world forests: biome patterns related to forest age. *Global Change Biology* **10**: 2052-2077.

Priess, J. A., Mimler, M., Klein, A. M., Schwarze, S., Tscharntke, T., & Steffan-Dewenter, I. 2007. Linking deforestation scenarios to pollination services and economic returns in coffee agroforestry systems. *Ecological Applications* **17**: 407-17.

Pugno, N. M. 2007 Towards a Spiderman suit: large invisible cables and self-cleaning releasable superadhesive materials. *Journal of Physics: Condensed Matter* **19**: 395001.

Pulkki, R. 1997. *Global Fibre Supply Study Working Paper Series: Literature Synthesis on Logging Impacts in Moist Tropical Forests.* Rome, Italy: Food and Agriculture Organization of the United Nations.

Purvis, A., Agapow, P., Gittleman, J. L., & Mace, G. M. 2000. Nonrandom extinction and the loss of evolutionary history. *Science* **288**: 328-330.

Pusey, B. J., & Arthington, A. H. 2003. Importance of the riparian zone to the conservation and management of freshwater fish: a review. *Marine and Freshwater Research* **54**: 1-16.

Putz, F. E., Zuidema, P., Peña-Claros, M., Pinard, M. A., Sheil, D., Vanclay, J. K., & Synnott, T. *in prep.* Tropical timber yields are not being sustained, but maintaining forests is a higher priority.

Rabalais, N. N., Turner, E. R., & Wiseman, W. R. 2002. Gulf of Mexico hypoxia, a.k.a. “The Dead Zone”. *Annual Review of Ecology and Systematics* **33**: 235-263.

Ramankutty, N., & Foley, J. A. 1999. Estimating historical changes in global land cover: Croplands from 1700 to 1992. *Global Biogeochemical Cycles* **13**: 997-1027.

Ramankutty, N., Evan, A. T., Monfreda, C., & Foley, J. A. 2008. Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochemical Cycles* **22**: GB1003.

Rao, M., & McGowan, P. J. K. 2002. Wild-meat use, food security, livelihoods, and conservation. *Conservation Biology* **16**: 580-583.

Redford, K. H. 1992. The empty forest. *BioScience* **42**: 412-422.

Richards, A. J. 2001. Does Low Biodiversity Resulting from Modern Agricultural Practice Affect Crop Pollination and Yield? *Annals of Botany* **88**: 165-172.

Ricketts, T. H. 2004. Tropical Forest Fragments Enhance Pollinator Activity in Nearby Coffee Crops. *Conservation Biology* **18**: 1262-1271.

Ricketts, T. H., Daily, G. C., Ehrlich, P. R., & Michener, C. D. 2004. Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences USA* **101**: 12579-12582.

Ricketts, T. H., Regetz, J., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., Bogdanski, A., Gemmill-Herren, B., Greenleaf, S. S., Klein, A. M., Mayfield, M. M., Morandin, L. A., Ochieng', A., & Viana, B. F. 2008. Landscape effects on crop pollination services: are there general patterns? *Ecology Letters* **11**: 499-515.

Ricketts, T. H., Williams, N. M., & Mayfield, M. M. 2006. Connectivity and ecosystem services: crop pollination in agricultural landscapes. In K. R. Crooks & M. Sanjayan (Eds.), *Connectivity Conservation*. Cambridge, UK: Cambridge University Press (pp. 255-289).

Rist, J. 2007. *Bushmeat Catch per Unit Uffort in Space and Time: A Monitoring Tool for Bushmeat Hunting*, London, UK: Imperial College London / ZSL.

Rizkalla, C., Blanco-Silva, F., & Gruver, S. 2007. Modeling the impact of ebola and bushmeat hunting on western lowland gorillas. *EcoHealth* **4**: 151-155.

Roberts, C. M., Andelman, S., Branch, G., Bustamante, R. H., Carlos Castilla, J., Dugan, J., Halpern, B. S., Lafferty, K. D., Leslie, H., Lubchenco, J., McArdle, D., Possingham, H. P., Ruckelshaus, M., & Warner, R. R. 2003. Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications* **13**: 199.

Roberts, C. M., Bohnsack, J. A., Gell, F., Hawkins, J. P., & Goodridge, R. 2001. Effects of Marine Reserves on Adjacent Fisheries. *Science* **294**: 1920-1923.

Robinson, J. G., & Bennett, E. L. 2004. Having your wildlife and eating it too: an analysis of hunting sustainability across tropical ecosystems. *Animal Conservation* **7**: 397-408.

Robinson, J. G., & Redford, K. H. 1986b. Intrinsic rate of natural increase in Neotropical forest mammals: relationship to phylogeny and diet. *Oecologia* **68**: 516-520.

Robinson, J. G., & Redford. K. H. 1986a. Body Size, Diet, and Population-Density of Neotropical Forest Mammals. *American Naturalist* **128**:665-680.

Robinson, J. G., Redford, K. H., & Bennett, E. L. 1999. Wildlife harvest in logged tropical forests. *Science* **284**: 595-596.

Rockel, M. L., & Kealy, M. J. 1991. The Value of Nonconsumptive Wildlife Recreation in the United States. *Land Economics* **67**: 422-434.

Rogers, C. S. 1990. Response of coral reefs and reef organisms to sedimentation. *Mar. Ecol. Prog. Ser.* **62**, 185-202.

Roldan, A. I., & Simonetti, J. A. 2001. Plant-mammal interactions in tropical Bolivian forests with different hunting efforts. *Conservation Biology* **15**: 617-623.

Ronnback, P. 1999. The ecological basis for economic value of seafood production supported by mangrove ecosystems. *Ecological Economics* **29**: 235-252.

Rosegrant, M. W., Agcaoili-Sombilla, M., Perez, N. D., 1995 *Global Food Projections to 2020: Implications for Investment*. Diane Publishing.

Rosell, F., Bozser, O., Collen, P. and Parker, H. 2005 Ecological impact of beavers *Castor fiber* and *Castor Canadensis* and their ability to modify ecosystems. *Mammal Rev.* **35**: 248-276.

Roubik, D.W. 2002. Feral African bees augment neotropical coffee yield. In *Pollinating Bees: The Conservation Link Between Agriculture and Nature* (eds P. Kevan & V.L. Imperatriz-Fonseca). Brasília, Brazil: Ministry of Environment (pp. 255–266).

Roughgarden, J. 1995. Can economics protect biodiversity? In *The Economics and Ecology of Biodiversity Decline: the Forces Driving Global Change*. Cambridge, UK: Cambridge University Press (pp. 149-155).

Roughgarden, J., & Smith, F. 1996. Why fisheries collapse and what to do about it. *Proceedings of the National Academy of Sciences USA* **93**: 5078-5083.

Rounsevell, M. D. A., Reginster, I., Araújo, M. B., Carter, T. R., Dendoncker, N., Ewert, F., House, J. I., Kankaanpaa, S., Leemans, R., Metzger, M. J., May 2006. A coherent set of future land use change scenarios for Europe. *Agriculture, Ecosystems & Environment* **114:** 57-68.

Rounsevell, M., Reginster, I., Araujo, M., Carter, T., Dendoncker, N., Ewert, F., House, J., Kankaanpaa, S., Leemans, R., Metzger, M., Schmit, C., Smith, P., & Tuck, G. 2006. A coherent set of future land use change scenarios for Europe. *Agriculture, Ecosystems & Environment* **114**: 57-68.

Rowcliffe, J. M., Cowlishaw, G., & Long, J. 2003. A model of human hunting impacts in multi-prey communities. *Journal of Applied Ecology* **40**: 872-889.

Rowcliffe, J. M., Milner-Gulland, E., & Cowlishaw, G. 2005. Do bushmeat consumers have other fish to fry? *Trends in Ecology & Evolution* **20**: 274-276.

Rowell, K., Flessa, K. W., Dettman, D. L., & Roman, M. 2005. The importance of Colorado River flow to nursery habitats of the Gulf corvina (*Cynoscion othonopterus*). *Canadian Journal of Fisheries and Aquatic Sciences* **62**: 2874-2885.

RSPB (Royal Society for the Protection of Birds). 2006. The RSPB: Help us halt illegal hunting: France. Online: http://www.rspb.org.uk/supporting/campaigns/illegalhunting/france.asp (April 26, 2008)

Runte, A. 1987. *National Parks: the American experience*. Lincoln & London: University of Nebraska Press.

Rusuwa, B., Maruyama, A. and Yuma, M. 2006 Deterioration of cichlid habitat by increased sedimentation in the rocky littoral zone of Lake Malawi. *Ichthyol Res* **53**: 431–434.

Ryder, R. A. 1965 A Method for Estimating the Potential Fish Production of North-temperate Lakes. *Transactions of the American Fisheries Society* **94**: 214–218.

Sampson, R. N., Bystriakova, N., Brown, S., Gonzalez, P., Irland, L. C., Kauppi, P., Sedjo, R., Thompson, I. D., Barber, C. V., & Offrell, R. 2005. Timber, Fuel, and Fiber. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 243-269). OnlineL http://www.millenniumassessment.org/documents/document.278.aspx.pdf

Sandker, M., Suwarno, A., Campbell, B. M., 2007. Will forests remain in the face of oil palm expansion? simulating change in Malinau, Indonesia. *Ecology and Society* **12**: 37+.

Sathaye, J., Chan, P., Dale, L., Makundi, W., Andrasko, K., 2003. A summary note estimating global forestry GHG mitigation potential and costs: A dynamic partial equilibrium approach. *Economic and Environmental Modeling Workshop, 19-20 January, 2004*. Habitat Center, New Delhi.

Saunders, M. A., & Lea, A. S. 2008. Large contribution of sea surface warming to recent increase in Atlantic hurricane activity. *Nature* **451**: 557-560.

SAUP (Sea Around Us Project). 2008. Sea Around Us Project. Online: http://www.seaaroundus.org/.

SCBD (Secretariat of the Convention on Biological Diversity). 2003. *Interlinkages between biological diversity and climate change: Advice on the integration of biodiversity considerations into the implementation of the United Nations Framework Convention on Climate Change and its Kyoto protocol.* Montreal, Canada: Secretariat of the Convention on Biological Diversity.

Scheffer, M., Carpenter, S., & Young, B. D. 2005. Cascading effects of overfishing marine systems. *Trends in Ecology & Evolution* **20**: 579-581.

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., & Walker, B. 2001. Catastrophic shifts in ecosystems. *Nature* **413**: 591-596.

Scherr, S. J., & McNeely, J. A. 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. *Philosophical Transactions of the Royal Society of London B* **363**: 477-94.

Schipmann, U., Leaman, D., & Cunningham, A. 2006. A comparison of cultivation and wild collection of medicinal and aromatic plants under sustainability aspects. In *Medicinal and Aromatic Plants*. The Netherlands: Springer (pp. 75-95).

Schjønning, P., Elmholt, S., & Christensen, B. T. (Eds.). 2004. *Managing Soil Quality: Challenges in Modern Agriculture*. CABI.

Schneider, R. R., Arima, E., Verissimo, A., Souza, C., & Barreto, P. 2002. *Sustainable Amazon: Limitations and Opportunities for Rural Development*. Washington D.C., USA: The World Bank.

Schröter, D., Acosta-Michlik, L., Arnell, A., Araújo, M., Badeck, F., Bakker, M., Bondeau, A., Bugmann, H., Carter, T., Vega-Leinert, A. D. L., Erhard, M., Espiñeira, G., Ewert, F., Fritsch, U., Friedlingstein, P., et al. 2004. *Advanced Terrestrial Ecosystem Analysis and Modelling - Final Report 2004* (Section 5 and 6 and Annex 1 to 6). Potsdam, Germany: Potsdam Institute for Climate Impact Research. Online: http://www.pik-potsdam.de/ateam/ateam_final_report_sections_5_to_6.pdf.

Schuttenberg, H. Z. 2001. Coral Bleaching; Causes, Consequences and Response: Selected Papers presented at the 9th International Coral Reef Symposium on "*Coral Bleaching: Assessing and Linking Ecological and Socioeconomic Impacts, Future Trends and Mitigation Planning*". Rhode Island, USA: Coastal Resources Centre, University of Rhode Island.

Schwab, O., Pulkki, R., & Bull, G. 2001. *Reduced Impact Logging in Tropical Forests: Literature synthesis, analysis and protoype statistical framework*. Rome, Italy: Food and Agriculture Organization of the United Nations.

SEI, no date. *Scenarios for sustainability*. Stockholm Environment Institute US Center. Online: http://www.scenariosforsustainability.org/.

Seitzinger, S. P., Harrison, J. A., Dumont, E., Beusen, A. H. W., Bouwman, A. F., 2005. Sources and delivery of carbon, nitrogen, and phosphorus to the coastal zone: An overview of global nutrient export from watersheds (news) models and their application. *Global Biogeochemical Cycles* **19**: GB4S01+

Shennan, C. 2008. Biotic interactions, ecological knowledge and agriculture. *Philosophical transactions of the Royal Society of London B* **363**: 717-39.

Sheppard, C., Dixon, D. J., Gourlay, M., Sheppard, A., & Payet, R. 2005. Coral mortality increases wave energy reaching shores protected by reef flats: Examples from the Seychelles. *Estuarine, Coastal and Shelf Science* **64**: 223-234.

Shrestha RK, Stein TV, Clark J. 2007. Valuing nature-based recreation in public natural areas of the Apalachicola River region, Florida. *Journal of Environmental Management* **85**: 977-985.

Sidle, R. C., Ziegler, A. D., Negishi, J. N., Nik, A. R., Siew, R., & Turkelboom, F. 2006. Erosion processes in steep terrain--Truths, myths, and uncertainties related to forest management in Southeast Asia. *Forest Ecology and Management* **224**: 199-225.

Sierra, R., Rodriguez, F., & Losos, E. 1999. Forest resource use change during early market integration in tropical rain forests: the Huaorani of upper Amazonia. *Ecological Economics* **30**: 107-119.

Silveira, R., & Thorbjarnarson, J. B. 1999. Conservation implications of commercial hunting of black and spectacled caiman in the Mamirauá Sustainable Development Reserve, Brazil. *Biological Conservation* **88**: 103-109.

Simpson, R. D., Sedjo, R. A., & Reid, J. W. 1996. Valuing biodiversity for use in pharmaceutical research. *The Journal of Political Economy* **104**: 163-185.

Simpson, R. L., Good, R. E., Walker, R., & Frasco, B. R. 1983. The role of Delaware River freshwater tidal wetlands in the retention of nutrients and heavy metals. *J. Environ. Qual* **12**: 41-48.

Sinclair, A. R. E., & Krebs, C. J. 2002. Complex numerical responses to top-down and bottom-up processes in vertebrate populations. *Philosophical Transactions of the Royal Society B* **357:** 1221-1231.

Sist, P., Picard, N., & Gourlet-Fleury, S. 2003. Sustainable cutting cycle and yields in a lowland mixed dipterocarp forest of Borneo. *Annals of Forest Science* **60**: 803-814.

Smith, K. G. & Darwall, W. R.T. 2006. *The Status and Distribution of Freshwater Fish Endemic to the Mediterranean Basin.* IUCN, Gland, Switzerland and Cambridge, UK.

Smith, R. S., Shiel, R. S., Bardgett, R. D., Millward, D., Corkhill, P., Evans, P., Quirk, H., Hobbs, P. J., & Kometa, S. T. 2008. Long-term change in vegetation and soil microbial communities during the phased restoration of traditional meadow grassland. *Journal of Applied Ecology* **45**: 670-679.

Soares-Filho, B. S., Nepstad, D. C., Curran, L. M., Cerqueira, G. C., Garcia, R. A., Ramos, C. A., Voll, E., Mcdonald, A., Lefebvre, P., Schlesinger, P. 2006. Modelling conservation in the Amazon basin. *Nature* **440**: 520-523.

Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., Miller, H. L. (Eds.) 2007. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I Contribution to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge.

Spalding, M. D., Ravilious, C., & Green, E. P. (Eds.). 2001. *World Atlas of Coral Reefs*. Berkley, CA, USA: University of California Press. Retrieved April 17, 2008, from http://www.unep-wcmc.org/marine/seagrassatlas/index.htm

Spalding, M., Blasco, F., & Field, C. (Eds.). 1997. *World Mangrove Atlas*. Okinawa, Japan: International Society of Mangrove Ecosystems.

Spotila, J. R., Reina, R. D., Steyermark, A. C., Plotkin, P. T., Paladino, F. V. 2000 Pacific leatherback turtles face extinction. *Nature* **405**:529-530.

Srinivasan, U. T., Carey, S. P., Hallstein, E., Higgins, P. A. T., Kerr, A. C., Koteen, L. E., Smith, A. B., Watson, R., Harte, J., & Norgaard, R. B. 2008. The debt of nations and the distribution of ecological impacts from human activities. *Proceedings of the National Academy of Sciences USA* **105**: 1768-1773.

Stachowicz J., Bruno, J. F. & Duffy, J. E. 2007 Understanding the effects of marine biodiversity on communities and ecosystems. *Annual Review of Ecology, Evolution, and Systematics* **38**: 739-766.

Stachowicz, J. J., Whitlatch, R. B., & Osman, R. W. 1999. Species diversity and invasion resistance in a marine ecosystem. *Science* **286**: 1577-1579.

Steffan-Dewenter, I. 2003. Importance of Habitat Area and Landscape Context for Species Richness of Bees and Wasps in Fragmented Orchard Meadows. *Conservation Biology* **17**: 1036-1044.

Steffan-Dewenter, I., Munzenberg, U., Burger, C., Thies, C., & Tscharntke, T. 2002. Scale-Dependent Effects of Landscape Context on Three Pollinator Guilds. *Ecology* **83**: 1421-1432.

Stein, C. J., & Colditz, G. A. 2004. The Epidemic of Obesity. *J. Clin. Endocrinol. Metab.* **89**: 2522-2525.

Steinfeld, H., Gerber, P., Wassenaar, T., Castel, V., Rosales, M., & de Haan, C. 2006. *Livestock's long shadow: Environmental issues and options*. Livestock, Environment and Development (LEAD) Initiative of the Food and Agriculture Organization (FAO).

Stern, N. 2007. *The Economics of Climate Change: The Stern Review*. Cambridge University Press.

Stevenson, S. 2007. *The European Fishing Industry*. Brussels, Belgium: European Parliament. Online: http://www.scottishtorymeps.org.uk/archive/stevenson/issues/reports/NewEUFish.pdf

Stocks, K. 2004. Seamountsonline: An online resource for data on the biodiversity of seamounts. In T. Morato & D. Pauly (Eds) *Seamounts: Biodiversity and fisheries*. Vancouver, Canada: Fisheries Centre, University of British Columbia (pp. 13-16).

Stoner, K. E., Riba-Hernandez, P., Vulinec, K., & Lambert, J. E. 2007. The role of mammals in creating and modifying seedshadows in tropical forests and some possible consequences of their elimination. *Biotropica* **39**: 316-327.

Straub, C. S., & Snyder, W. E. 2006. Species identity dominates the relationship between predator biodiversity and herbivore suppression. *Ecology* **87**: 277-282.

Stuart, S. N., Chanson, J. S., Cox, N. A., Young, B. E., Rodrigues, A. S. L., Fischman, D. L., & Waller, R. W. 2004. Status and trends of amphibian declines and extinctions Worldwide. *Science* **306**: 1783-1786.

Sundaravadivel, M., & Vigneswaran, S. 2001. Constructed wetlands for wastewater treatment. *Critical Reviews in Environmental Science and Technology* **31**: 351-409.

Sunderland, K., & Samu, F. 2000. Effects of agricultural diversification on the abundance, distribution, and pest control potential of spiders: a review. *Entomologia Experimentalis et Applicata* **95**: 1-13.

Swift, M. J., Izac, A. N., & van Noordwijk, M. 2004. Biodiversity and ecosystem services in agricultural landscapes--are we asking the right questions? *Agriculture, Ecosystems & Environment* **104**: 113-134.

Synolakis, C. E., & Kong, L. 2006. Runup measurements of the December 2004 Indian Ocean tsunami. *Earthquake Spectra* **22**: S67-S91.

Syvitski, J. P. M., Vorosmarty, C. J., Kettner, A. J., & Green, P. 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science* **308**: 376-380.

Szwagrzyk, J., & Gazda, A. 2007. Above-ground standing biomass and tree species diversity in natural stands of Central Europe. *Journal of Vegetation Science* **18**: 555-562.

Takano, T., Nakamura, K., & Watanabe, M. 2002. Urban residential environments and senior citizens' longevity in megacity areas: the importance of walkable green spaces. *J Epidemiol Community Health* **56**: 913-918.

Ten Brink, B., Alkemade, R., Bakkenes, M., Eickhout, B., de Heer, M., Kram, T., Manders, T., van Oorschot, M., Smout, F., Clement, J., van Vuuren, D., Westhoek, H., Miles, L., Lysenko, I., Fish, L., et al. 2006. *Cross-roads of Planet Earth. Exploring means to meet the 2010 biodiversity target*. Montreal, Canada: Secretariat of the Convention on Biological Diversity.

Terborgh, J., Lopez, L., Nunez, P., Rao, M., Shahabuddin, G., Orihuela, G., Riveros, M., Ascanio, R., Adler, G. H., Lambert, T. D., & Balbas, L. 2001. Ecological meltdown in predator-free forest fragments. *Science* **294**: 1923-1926.

The World Bank. 2005. *Going, Going, Gone: The Illegal Trade of Wildlife in East and Southeast Asia*. Washington D.C., USA: The World Bank.

Thies, C., Roschewitz, I., & Tscharntke, T. 2005. The landscape context of cereal aphid–parasitoid interactions. *Proceedings of the Royal Society B* **272**: 203-210.

Thomas, C. D., Cameron, A., Green, R. E., Bakkenes, M., Beaumont, L. J., Collingham, Y. C., Erasmus, B. F. N., de Siqueira, M. F., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A. S., Midgley, G. F., Miles, L., et al. 2004. Extinction risk from climate change. *Nature* **427**: 145-148.

Tian, H., Melillo, J. M., Kicklighter, D. W., McGuire, A. D. & Helfrich, J. 1999. The sensitivity of terrestrial carbon storage to historical climate variability and atmospheric CO2 in the United States. *Tellus Ser. B Chem. Phys. Meteorol.* **51**: 414-452.

Tilman, D., Reich, P. B., & Knops, J. M. H. 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* **441**: 629-632.

Tockner, K., & Stanford, J. A. 2002. Riverine flood plains: present state and future trends. *Environmental Conservation* **29**: 308-330.

Toman, M. 1998. Why not to calculate the value of the world's ecosystem services and natural capital. *Ecological Economics* **25**: 57-60.

Tscharntke, T., Bommarco, R., Clough, Y., Crist, T. O., Kleijn, D., Rand, T. A., Tylianakis, J. M., Nouhuys, S. V., & Vidal, S. 2007. Conservation biological control and enemy diversity on a landscape scale. *Biological Control* **43**: 294-309.

Tscharntke, T., Gathmann, A., & Steffan-Dewenter, I. 1998. Bioindication using trap-nesting bees and wasps and their natural enemies: community structure and interactions. *The Journal of Applied Ecology* **35**: 708-719.

Tscharntke, T., Klein, A. M., Kruess, A., Steffan-Dewenter, I., & Thies, C. 2005. Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters* **8**: 857-874.

Tscharntke, T., Rand, T., & Bianchi, F. 2005. The landscape context of trophic interactions: insect spillover across the crop - noncrop interface. *Ann. Zool. Fennici* **42**: 421-432.

Turner, R. K. 1999. The Place of Economic Values in Environmental Valuation. In I. J. Bateman & K. G. Willis (Eds.), *Valuing Environmental Preferences: Theory and Practice of the Contingent Valuation Method in the US, EU and Developing Countries*. Oxford, UK: Oxford University Press (pp. 17 – 41).

Turner, R. K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., & Georgiou, S. 2003. Valuing nature: lessons learned and future research directions. *Ecological Economics* **46**: 493-510.

Turner, R., Adger, W., & Brouwer, R. 1998. Ecosystem services value, research needs, and policy relevance: a commentary. *Ecological Economics* **25**: 61-65.

Tylianakis, J. M., Didham, R. K., & Wratten, S. D. 2004. Improved fitness of aphid parasitoids receiving resource subsidies. *Ecology* **85**: 658-666.

Tylianakis, J. M., Tscharntke, T., & Lewis, O. T. 2007. Habitat modification alters the structure of tropical host-parasitoid food webs. *Nature* **445**: 202-205.

Ulrich, R. S., Simons, R. F., Losito, B. D., Fiorito, E., Miles, M. A., & Zelson, M. 1991. Stress recovery during exposure to natural and urban environments. *Journal of Environmental Psychology* **11**: 201-230.

UNDP (United Nations Development Programme). 2006. *Human Development Report 2006. Beyond scarcity: Power, poverty and the global water crisis*. New York, NY, USA: United Nations Development Programme.

UNEP (United Nations Environment Programme). 2005. *After the Tsunami: Rapid Environmental Assessment*. Nairobi, Kenya: United Nations Environment Programme.

UNESCO-WWAP (United Nations Educational, Scientific and Cultural Organization – World Water Assessment Programme). 2006. *The 2nd UN World Water Development Report: 'Water, a shared responsibility'*. New York, NY, USA: United Nations Educational, Scientific and Cultural Organization – World Water Assessment Programme.

UNFF (United Nations Forum on Forests). 2007. United Nations Forum on Forests. Online: http://www.un.org/esa/forests/about.html.

USFW (U.S. Fish & Wildlife Service). 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation - State Overview (Preliminary Findings)*. Washington D.C.: U.S. Fish & Wildlife Service.

USFWS 2007. 2006 *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation: State Overview*. Washington D.C.: US Fish and Wildlife Service.

Uyarra, MC, Cote, IM, Gill, JA, Tinch, RRT, Viner, D, & Watkinson, AR. 2005. Island-specific preferences of tourists for environmental features: Implications of climate change for tourism-dependent states. *Environmental Conservation* **32**: 11-19.

Valiela, I., Bowen, J. L., & York, J. K. 2001. Mangrove Forests: One of the World's Threatened Major Tropical Environments. *BioScience* **51**: 807-815.

van der Heijden, M. G. A., Bardgett, R. D., & van Straalen, N. M. 2008. The unseen majority: soil microbes as drivers of plant diversity and productivity in terrestrial ecosystems. *Ecology Letters* **11**: 296-310.

van Dijk, P. P., Stuart, B. L., & Rhodin, A. G. 2000. *Asian Turtle Trade: Proceedings of a Workshop on Conservation and Trade of Freshwater Turtles and Tortoises in Asia*. Lunenburg, MA, USA: Chelonian Research Foundation.

Van Voorhees, D., & Pritchard, E. S. 2007. *Fisheries of the United States 2006*. Silver Spring, MD, USA: National Marine Fisheries Service, Fisheries Statistics Division.

van Wilgen, B. W., Cowling, R. M., & Burgers, C. J. 1996. Valuation of ecosystem services: a case study from South African fynbos ecosystems. *BioScience* **46**: 184-189.

Vazquez, D. P., Morris, W. F., & Jordano, P. 2005. Interaction frequency as a surrogate for the total effect of animal mutualists on plants. *Ecology Letters* **8**: 1088-1094.

Veerman, J. L., Barendregt, J. J., van Beeck, E. F., Seidell, J. C., & Mackenbach, J. P. 2007. Stemming the Obesity Epidemic: A Tantalizing Prospect. *Obesity* **15**: 2365-2370.

Verburg, P. H., Soepboer, W., Veldkamp, A., Limpiada, R., Espaldon, V., Mastura, S. S., 2002. Modeling the spatial dynamics of regional land use: The clue-s model. *Environmental Management* **30**: 391-405.

Vilà, M., Vayreda, J., Comas, L., Ibáñez, J. J., Mata, T., & Obón, B. 2007. Species richness and wood production: a positive association in Mediterranean forests. *Ecology Letters* **10**: 241-50.

Vilà, M., Vayreda, J., Gracia, C., & Ibáñez, J. J. 2003. Does tree diversity increase wood production in pine forests? *Oecologia* **135**: 299-303.

Voeks, R. 2004. Disturbance Pharmacopoeias: Medicine and Myth from the Humid Tropics. *Annals of the Association of American Geographers* **94**: 868-888.

Vörösmarty, C. J., Lévêque, C., Revenga, C., Bos, R., Caudill, C., Chilton, J., Douglas, E. M., Meybeck, M., Prager, D., Balvanera, P., Barker, S., Maas, M., Nilsson, C., Oki, T., Reidy, C. A., et al. 2005. Fresh Water. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 165-207).

Vörösmarty, C., Federer, C., & Schloss, A. 1998. Potential evaporation functions compared on US watersheds: Possible implications for global-scale water balance and terrestrial ecosystem modeling. *Journal of Hydrology* **207**: 147-169.

Vukusic, P., Hallam, B., & Noyes, J. 2007. Brilliant whiteness in ultrathin beetle scales. *Science* **315**: 348.

Wade, DJ, Mwasaga, BC & Eagles, PFJ. 2001. A history and market analysis of tourism in Tanzania. *Tourism Management* **22**: 93-101.

Waite, T. 2007. Revisiting evidence for sustainability of bushmeat hunting in West Africa. *Environmental Management* **40**: 476-480.

Wallace, K. J. 2007. Classification of ecosystem services: Problems and solutions. *Biological Conservation* **139**: 235-246.

Walpole, M. J., & Leader-Williams, N. 2002. Tourism and flagship species in conservation. *Biodiversity and Conservation* **11**: 543-547.

Walpole, M.J. & Thouless, C.R. 2005. Increasing the value of wildlife through non-consumptive use? Deconstructing the myths of ecotourism and community-based tourism in the tropics. In *People & Wildlife: Conflict or Coexistence* (Eds R.Woodroffe, s. Thirgood & A.Rabinowitz).

Walpole, M.J. 1997. *Dragon tourism in Komodo National Park, Indonesia: its contribution to conservation and local development*. PhD thesis, University of Kent, Canterbury, UK.

Walsh, P. D., Abernethy, K. A., Bermejo, M., Beyers, R., De Wachter, P., Akou, M. E., Huijbregts, B., Mambounga, D. I., Toham, A. K., Kilbourn, A. M., Lahm, S. A., Latour, S., Maisels, F., Mbina, C., Mihindou, Y., et al. 2003. Catastrophic ape decline in western equatorial Africa. *Nature* **422**: 611-614.

Watson, R., Coles, R. G., & Long, W. J. L. 1993. Simulation estimates of annual yield and landed value for commercial penaid prawns from a tropical seagrass habitat, Northern Queensland, Australia. *Aust. J. Mar. Freshwater Res.* **44**: 211-219.

Watson, R., Morato, T. 2004. Exploitation Patterns in Seamount Fisheries: A preliminary analysis. In T. Morato & D. Pauly (Eds) *Seamounts: Biodiversity and fisheries*. Vancouver, Canada: Fisheries Centre, University of British Columbia (pp. 61-66).

Welcomme, R. L. 1999 A review of a model for qualitative evaluation of exploitation levels in multi-species fisheries. *Fisheries Management and Ecology* **6**:1–19.

Welcomme, R. L., & Halls, A. 2004. Dependence of tropical river fisheries on flow. In R. L. Welcomme & T. Petr (Eds.), *International Symposium on the Management of Large Rivers for Fisheries. Phnom Pehn, Cambodia 11-14 Feb 2005*. Bangkok, Thailand: Food and Agriculture Organization of the United Nations (FAO) and Mekong River Commission (MRC).

Wells, M.P. 1993. Neglect of biological riches: the economics of nature tourism in Nepal. *Biodiversity & Conservation* **2**: 445-464.

Wells, S., Ravilious, C., & Corcoran, E. 2006. *In the front line: Shoreline protection and other ecosystem services from mangroves and coral reefs* . Cambridge, UK: UNEP-World Conservation Monitoring Centre.

Westerkamp, C., & Gottsberger, G. 2000. Diversity Pays in Crop Pollination. *Crop Science* **40**: 1209-1222.

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. 2006. Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science* **313**: 940-943.

Whitman, K., Starfield, A. M., Quadling, H. S., & Packer, C. 2004. Sustainable trophy hunting of African lions. *Nature* **428**: 175-178.

WHO (World Health Organization). 2002. *WHO Traditional Medicine Strategy 2002-2005*. Geneva, Switzerland: World Health Organization.

Wilby, A., & Thomas, M. B. 2002. Natural enemy diversity and pest control: patterns of pest emergence with agricultural intensification. *Ecology Letters* **5**: 353-360.

Wilby, A., Villareal, S. C., Lan, L. P., Heong, K. L., & Thomas, M. B. 2005. Functional benefits of predator species diversity depend on prey identity. *Ecological Entomology* **30**: 497-501.

Wilkie, D. S. & Lee, R. J. 2004. Hunting in agroforestry systems and landscapes: conservation implications in West-Central Africa and Southeast Asia. In G. Schroth, G. A. B. Da Fonseca, C. A. Harvey, H. L. Vasconceles, & A.-M. N. Izac (Eds.). *Agroforestry and Biodiversity Conservation in Tropical Landscapes*. Washington DC: Island Press. (pp 346-370)

Wilkie, D. S. 2005. Hunting in agroforestry systems and landscapes: conservation implications in West-Central Africa and Southeast Asia. In G. Schroth, G. da Fonseca, C. Harvey, C. Gascon, H. L. Vasconcelos, & A. M. N. Izac (Eds.), *Agroforestry and Biodiversity Conservation in Tropical Landscapes*. Washington D.C., USA: Island Press (pp. 346-370).

Wilkie, D. S., & Carpenter, J. 1999a. The potential role of safari hunting as a source of revenue for protected areas in the Congo Basin. *Oryx* **33**: 340-345.

Wilkie, D. S., & Carpenter, J. F. 1999b. Bushmeat hunting in the Congo Basin: an assessment of impacts and options for mitigation. *Biodiversity and Conservation* **8**: 927-955.

Wilkinson, C. 2006. Status of coral reefs of the world: summary of threats and remedial action. In I. M. Cote & J. D. Reynolds (Eds.), *Coral Reef Conservation*. Cambridge, UK: Cambridge University Press (pp. 3-39).

Wilkinson, C., Souter, D., & Goldberg, J. (Eds.). 2006. *Status of Coral Reefs in Tsunami Affected Countries: 2005*. Townsville, AUS: Australian Institute of Marine Science.

Willer, H., & Yussefi, M. (Eds.). 2008. *The World of Organic Agriculture - Statistics and Emerging Trends 2008*. Frick, Switzerland: International Federation of Organic Agriculture Movements (IFOAM), DE-Bonn and Research Institute of Organic Agriculture, FiBL.

Willer, H., Yussefi-Menzler, M., Sorensen, N. **(**Eds**).** 2008**.** *The World of Organic Agriculture - Statistics and Emerging Trends 2008***.** International Federation of Organic Agriculture Movements (IFOAM), DE-Bonn and Research Institute of Organic Agriculture, FiBL, CH-Frick**.**

Williams-Guillen, K., Perfecto, I., & Vandermeer, J. 2008. Bats limit insects in a neotropical agroforestry system. *Science* **320**: 70.

Wilson, E.O. 1984. *Biophilia*. Harvard University Press, Cambridge, MA.

Winfree, R., Griswold, T., & Kremen, C. 2007a. Effect of Human Disturbance on Bee Communities in a Forested Ecosystem. *Conservation Biology* **21**: 213-223.

Winfree, R., Williams, N. M., Dushoff, J., & Kremen, C. 2007b. Native bees provide insurance against ongoing honey bee losses. *Ecology Letters* **10**: 1105-1113.

Wint, W., & Robinson, T. 2006. *Gridded Livestock of the World*. Rome, Italy: Food and Agriculture Organization of the United Nations.

With, K. A., Pavuk, D. M., Worchuck, J. L., Oates, R. K., & Fisher, J. L. 2002. Threshold effects of landscape structure on biological control in agroecosystems. *Ecological Applications* **12**: 52-65 .

Wood, S., Ehui, S., Alder, J., Benin, S., Cassman, K. G., Cooper, D. H., Johns, T., Gaskell, J., Grainger, R., Kadungure, S., Otte, J., Rola, A., Watson, R., Wijkstrom, U., Devendra, C., et al. 2005. Food. In *Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Current States and Trends*. Washington D.C., USA: Island Press (pp. 209-241).

Wood, S., Sebastian, K., & Scherr, S. J. *Pilot Analysis of Global Ecosystems: Agroecosystems*. Washington D.C., USA: International Food Policy Research Institute (IFPRI) and World Resources Institute.

Woolley, H., & Rose, S. 2004. *The Value of Public Space*. London, UK: CABE Space.

World Health Organisation. 2006. *Obesity and overweight factsheet*. Online: from http://www.who.int/mediacentre/factsheets/fs311/en/index.html.

Worm, B., Barbier, E. B., Beaumont, N., Duffy, J. E., Folke, C., Halpern, B., Jackson, J. B. C., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J., & Watson, R. 2006. Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science* **314**: 787-790.

Wösten, J. H. M., van den Berg, J., van Eijk, P., Gevers, G. J. M., Giesen, W. B. J. T., Hooijer, A., Idris, A., Leenman, P. H., Rais, D. S., Siderius, C., Silvius, M. J., Suryadiputra, N., & Wibisono, I. T. 2006. Interrelationships between hydrology and ecology in fire degraded tropical peat swamp forests. *International Journal of Water Resources Development* **22**: 157-174.

Wright, J. P., & Flecker, A. S. 2004. Deforesting the riverscape: the effects of wood on fish diversity in a Venezuelan piedmont stream. *Biological Conservation* **120**: 439-447.

WTTC 2007. *The global travel and tourism summit.* World Travel and Tourism Council, London. UK.

Yamano, Miyajima, & Koike. 2000. Importance of foraminifera for the formation and maintenance of a coral sand cay: Green Island, Australia. *Coral Reefs* **19**: 51-58.

Yeh, H., Liu, P., Briggs, M., & Synolakis, C. 1994. Propagation and amplification of tsunamis at coastal boundaries. *Nature* **372**: 353-355.

Yoshida, S., Shimada, Y., Kondoh, D., Kouzuma, Y., Ghosh, A. K., Jacobs-Lorena, M., & Sinden, R. E. 2007. Hemolytic C-type lectin CEL-III from sea cucumber expressed in transgenic mosquitoes impairs malaria parasite development. *PLoS Pathogens* **3**: e192.

Young, I. R., & Hardy, T. A. 1993. Measurement and modelling of tropical cyclone waves in the Great Barrier Reef. *Coral Reefs* **12**: 85-95.

Zavaleta, E. S. 2000. The economic value of controlling an invasive shrub. *Ambio* **29**: 462-467.

Zedler, J. B., & Kercher, S. 2005. Wetland resources: status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources* **30**: 39-74.

Zhang, W., Ricketts, T. H., Kremen, C., Carney, K., & Swinton, S. M. 2007. Ecosystem services and dis-services to agriculture. *Ecological Economics* **64**: 253-260.

Zhu, Y., Chen, H., Fan, J., Wang, Y., Li, Y., Chen, J., Fan, J., Yang, S., Hu, L., Leung, H., Mew, T. W., Teng, P. S., Wang, Z., & Mundt, C. C. 2000. Genetic diversity and disease control in rice. *Nature* **406**: 718-722.

Zondervan, I., Zeebe, R. E., Rost, B., & Riebesell, U. 2001. Decreasing marine biogenic calcification: a negative feedback on rising atmospheric pCO_2. *Global Biogeochemical Cycles* **15**: 507–516.

ANNEX 1 – COST AND BENEFITS OF AGRICULTURE CONVERSION

It is not straightforward to decide how to deal with systems (such as agricultural fields) that despite being heavily modified by human activities still hold substantial biodiversity and produce ecosystem services. For example, it is not clear if the economic value of wheat production should be considered a benefit of the ecosystem service 'food production' or an opportunity cost of not converting a forest to a wheat field.

This raised considerable discussion in each of the three consultations we had on the conceptual framework. This discussion turned out to be extremely productive, as it became clear that it does not matter how outputs from agricultural fields are classified – what is fundamental is to ensure the adequacy of the 'states' being compared (Section 2). Indeed, as long as these states are equal in everything else but the implementation or not of a set of conservation actions, the end economic consequences are the same irrespective of whether wheat production (for example) is treated as beneficial service enhanced by habitat conversion, or as an opportunity cost of not converting.

We illustrate this with a simplified example of a parcel of land with two possible states: a forest plot and a wheat field. Taken separately, each of these states has a set of running benefits and costs (lists of costs and benefits have been simplified for illustration purposes). The benefits of the forest plot are that it can produce a sustainable timber harvest (FB_1), stores carbon (FB_2) and it provides habitat for wild bees that pollinate nearby crops (FB_3). The costs of running the forest are expenses in forest management such as patrolling (FC_1) and paying compensation to nearby farmers for the damage caused by forest animals (FC_2). The benefits of the wheat field are wheat production (WB_1), and some carbon storage (WB_2), while the costs are those related to crop management, such as ploughing the field (WC_1) and paying to clean the pollution from agricultural runoff to nearby rivers (WC_2).

Note that 'timber production' and 'wheat production' are here simply as economic benefits of particular types of land use, irrespective of whether they are considered ecosystem services or not. In the same way, 'damage to nearby crops' and 'pollution runoff' are treated here as costs rather than 'ecosystem disservices'.

Suppose the question being asked is: **what are the net consequences of converting forest to wheat?**

As described in the general framework (Section 2), a comparison between states can be done by investigating differences in benefits and in costs in both states. When converting from forest to wheat (Figure A1), the differences in benefits are:

- The loss of the benefit of timber production (- FB1);
- The added benefit of wheat production (+ WB1);
- The difference in carbon storage (+ WB2 - FB2); and
- The loss of the benefit of pollination of nearby crops (- FB3).

In addition, there is a one-off benefit of conversion, the windfall of timber harvesting when removing the forest (+ TH).

The net difference in benefits (Δ B) is the sum of each of these parcels.

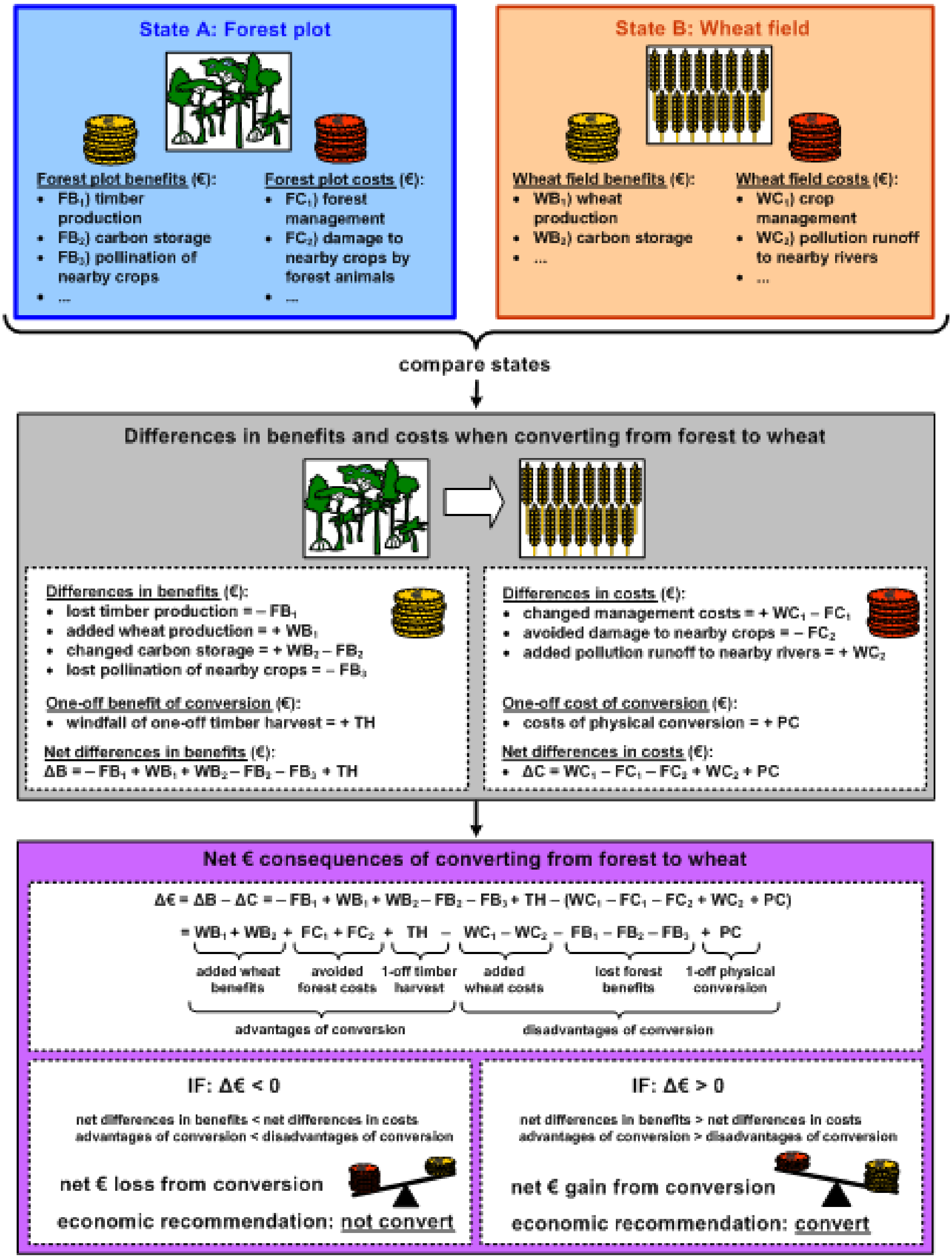

Figure A1. Evaluating the net consequences of converting a forest plot to a wheat field.

When converting from forest to wheat, the differences in costs are:

- The difference in the costs of managing wheat instead of forest (+ WC1 - FC1);

- The avoided costs of compensating for damage to nearby crops (- FC2); and
- The added costs of cleaning the pollution from agricultural runoff (+ WC2).

There is also a one-off cost of physically converting the land to make it suitable for wheat production (e.g., removing tree roots; + PC).

The net difference in costs (Δ C) is the sum of each of these parcels.

As per the general framework (Figure 2, Section 2), the net consequences of conversion (Δ €) can be assessed by comparing the net differences in benefits (Δ B) with the net differences in costs (Δ C). If the former are larger than the latter, there is a net economic gain (Δ € > 0) from conversion, hence it makes economic sense to convert.

The equation for the difference between net benefits and net costs can be rearranged to indicate as positive terms all the advantages of conversion (the added benefits from wheat, the avoided forest costs, and the one-off windfall in timber) and as negative terms all the disadvantages of conversion (the added costs from wheat, the lost benefits of forest, and the one-off costs of land conversion). If the advantages are larger than the disadvantages, it makes economic sense to convert (Figure A1).

Note that 'wheat production' was simply an added economic benefit of converting forest to wheat, irrespective of whether it is classified as an ecosystem service or not.

Suppose the question being asked is: **what are the net consequences of <u>not</u> converting forest to wheat?** Importantly, the question is not "what are the net consequences of retaining the forest" (these would simply be a continuation of state A), but the net consequences of actions put in place to conserve the forest plot from conversion to a specific alternative form of land use. Only by framing the question in this way it is possible to account for the opportunity costs of conserving the forest.

The differences in benefits when not converting to a wheat field are perfectly symmetrical to the differences in the benefits of conversion, and so are the differences in costs (Figure A2). When computing the net consequences of no conversion, all parcels that were previously the advantages of conversion (Figure A1) are now the disadvantages of no conversion (Figure A2), and vice-versa. The end result is exactly the same: the conditions that would result in the recommendation not to convert identical whichever way the question is put (Figure A1 cf Figure A2). In the latter figure, the value of 'wheat production' simply becomes an opportunity cost of maintaining the forest.

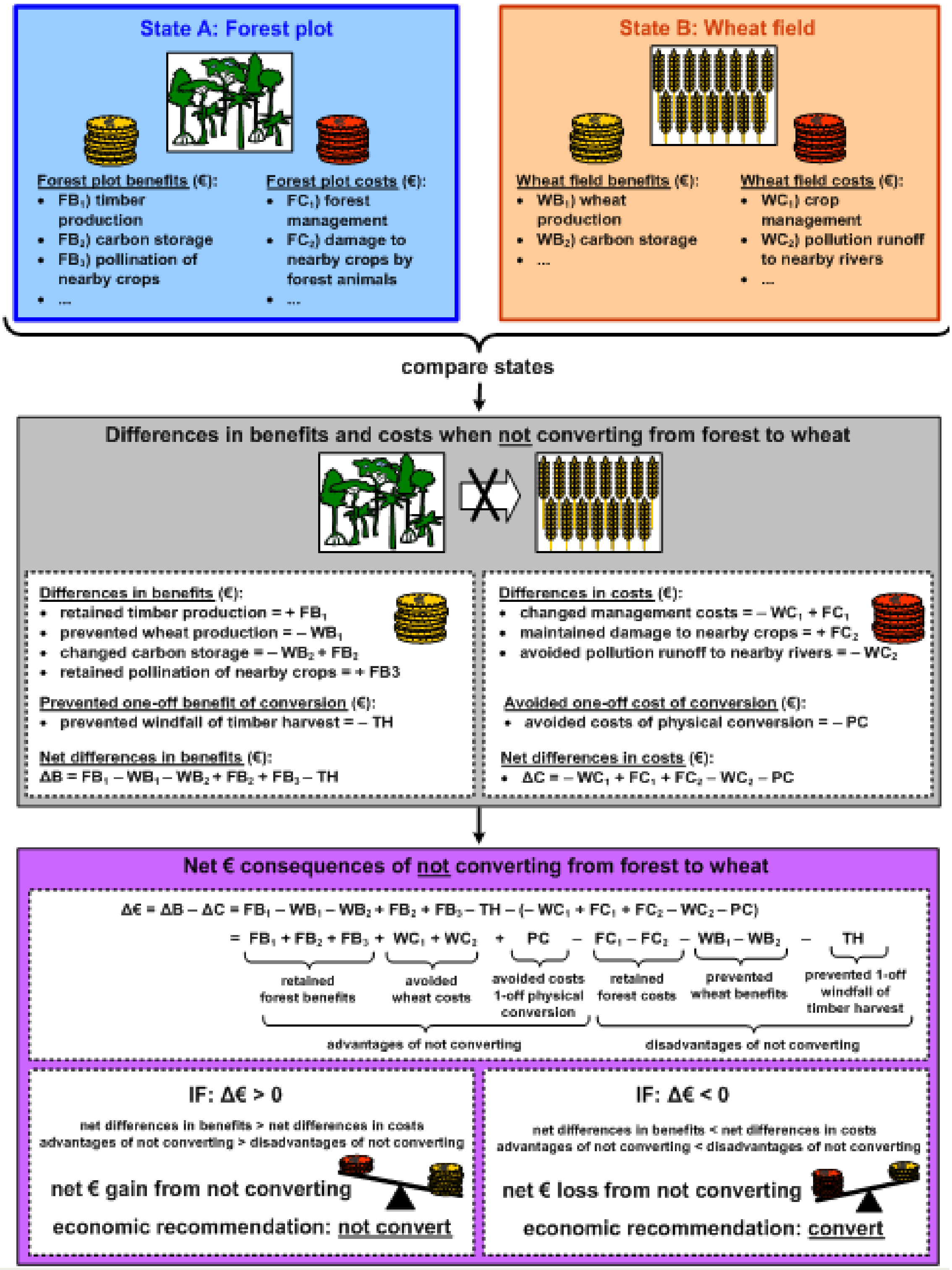

Figure A2. Evaluating the net consequences of not converting a forest plot to a wheat field.

As demonstrated, it is irrelevant which parcels are considered benefits from ecosystem services or not. The relevant information is whether they are benefits or costs of particular states. Accordingly, it does not matter what classification of ecosystem services is being used. Indeed, this same framework could have been used to assess the net economic consequences of converting (or not converting) a forest to a car park, or traditional extensive wheat production to intensive wheat production.

As discussed in Section 2, the fundamental step in ensuring that these comparisons address the remit of the Review is to ensure that the states being compared contrast two specific situations: one where conservation action to prevent wild nature loss has been put in place, and another where it has not. In the previous example of forest plot and wheat field, the comparison is relevant if it is deemed that conserving the forest is the action required for preventing biodiversity loss. The net economic consequences of conversion (Figure A1) can therefore be interpreted as the economic consequences of failing to protect biodiversity, while, conversely, the net economic consequences of no conversion (Figure A2) are the economic consequences of biodiversity loss.

ANNEX 2 – FRAMEWORK DEVELOPMENT AND CONSULTATION

We had three opportunities to present and discuss the framework with different audiences:

- Seminar “Conceptualising ecosystem services and human well-being”, 15 January, Department of Geography, University of Cambridge. We presented a talk to an audience dominated by environmental economists and geographers and solicited additional comments by email.

- Presentation “Framework for a Stern-like review of the economics of biodiversity loss”, 16 January, Department of Zoology, University of Cambridge. We presented a talk in one of our departmental lunchtime seminars. There were about 20 attendees, mainly biologists.

- Zoology workshop, 16 January, Department of Zoology, University of Cambridge: the framework was discussed in a small (8 people) workshop of key experts.

A detailed account of these discussions is presented in the Inception Report to this project.

The systematically raised concern was “the agriculture issue” – to include or not agricultural food production as an ecosystem service?). In these discussions we found what we believe is the solution for that problem (Section 2; Annex 1). Several other interesting issues were raised and discussed. Overall, the framework has proved remarkably robust. The discussions have also been very useful in making us think more carefully about the details of how the framework is applied (e.g., how to consider interactions and trade-offs between services).

ANNEX 3 - PRACTICAL STRATEGY FOLLOWED IN THE THEMATIC REVIEWS

This Scoping the Science project required reviewing very quickly a wide literature, most of which lay outside the immediate areas of expertise of the members of the team. We therefore had two limiting factors: expertise and time.

To make sure we access the relevant expertise, we had to draw on the support of experts for each theme, who know the literature well and have a broad perspective of the state of knowledge in their respective fields. Ideally, we would have wanted a team of experts in each theme producing the respective reports. This would have required a substantially longer and better-funded project than the current one. What was possible to obtain in the current project was more or less detailed guidance from a diversity of experts.

Following such guidance required time, to pursue the relevant references and to compile the overall information into a coherent report. The best way in which we were able to obtain researchers' time in such a short project was by hiring five interns. They were: James Beresford, Kelly Flower, Antares Hernández, Hannah Peck, and James Waters. We were fortunate to hire five bright young biologists, but being very recent graduates they had only limited research experience. It would have been impossible to hire more experienced researchers in such short notice and given the limited funds available. This meant that their outputs needed to be substantially edited before being sent back to the experts.

The ideal strategy for those themes that were fully developed consisted of the following steps:

- **Questionnaire –** A questionnaire (tailored to each task) was sent to a set of experts to try to obtain key pointers (to references, resources, other experts) as well as to capture the experts' experience and opinion across a range of questions.

- **First draft** – An intern reviewed the literature and the information provided by experts, and produced a first draft.

- **Second draft –** Given the limited experience of the interns, the first drafts produced required considerable editing and double-checking of the information. This was done by Ana Rodrigues. A second draft was then produced.

- **Review –** The second draft was sent back to the experts.

- **Final report –** Comments from the experts are incorporated into a final report, that becomes a chapter in the final report of the Scoping the Science project.

In practice, given time limitations, we were only able to follow all of these steps for six of the themes considered (Table A1). For two other themes, we benefited from expert contribution but there was no time for a subsequent review. One theme benefited from expert contribution but only a quick literature review was possible. Six other themes were quick literature reviews without expert contribution or review. And two themes were not developed at all (considered as lower priorities for Phase 2).

The contribution of experts is acknowledged in the following way:

- Authors: experts who have contributed substantial information and reviewed and approved the final report.

- Contributors: experts who provided initial information but were unable to review the report, or experts that considered that this was a more suitable description of their contribution.

- Reviewers: experts who reviewed the report but who were not authors. We gave experts the option to reproduce their comments at the end of each section if substantive comments were provided that we could not incorporate (e.g., given time limitations). Hence, being listed as a reviewer did not have to be considered an endorsement of the text.

- Acknowledgements: experts who contributed with references and/or contacts but not with specific information, and who did not see the final text.

The overall level of expert support was extraordinary, given how busy these experts inevitably are. Their willingness to contribute with their time indicates that they understand the importance of the Review on the Economics of Ecosystems and Biodiversity.

However, the extent to which we feel that we have succeeded in representing the state of knowledge in each field is very variable. In some cases (such as pollination) we feel that we have been able to tap into the knowledge and advice of the key people in the field, that they understood the purpose of this exercise, and that they took the time to provide substantial input. In other fields (e.g. wild harvested fibers) we feel that we were unable to obtain advice from key people and that therefore we may only present a partial view of the state of knowledge.

Table A1 indicates the level of information for each theme and, where applicable, the level of expert support. This is an assessment of the extent to which we feel that we have significantly tapped the overall expertise in the field and produced results that are representative of the overall state of knowledge.

In interpreting this report it is fundamental to keep in mind that, given the brief period for the execution of this project (4 months), all reviews were completed in a very short time. So even those described here are “fully developed” cannot be expected to be complete reviews of the literature, but only as quick assessments of the state of knowledge.

Table A1. Team (intern + head), level of information collected and level of expert support in each thematic review.

Task	**Team**	**Level of information**	**Expert support**
1. Wild crop pollination	JJ Waters + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Very good
2. Biological control in crops	JJ Waters + Ana Rodrigues	Quick literature review (no expert contribution or review)	
3. Genetic diversity of crops and livestock	Hannah Peck + Ana Rodrigues	Quick literature review (no expert contribution or review)	
4. Soil quality for crop production	Antares Hernández + Ana Rodrigues	Quick literature review (expert contribution; no expert review)	Reasonable
5.Livestock	JJ Waters + Ana Rodrigues	Quick literature review (no expert contribution or review)	
6. Marine fisheries	Kelly Flower + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Good
7. Inland fisheries and aquaculture	Kelly Flower + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Good
8. Wild animal products	JJ Waters + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Very good
9. Fresh water provision and regulation	Hannah Peck + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Reasonable
10. Timber from natural forests	Antares Hernández + Ana Rodrigues	Fully developed review (expert contribution; no expert review)	Reasonable
11. Wild medicinal plants	Antares Hernández + Ana Rodrigues	Quick literature review (no expert contribution or review)	
12. Nature-related outdoor activities	James Beresford + Matt Walpole	Fully developed review (led by expert; no further review)	Good
13. Regulation of natural hazards	James Beresford + Ana Rodrigues	Fully developed review (expert contribution and expert review)	Good
14. One-off use benefits	Ana Rodrigues	Quick literature review (no expert contribution or review)	
15. Non-use benefits	Ana Rodrigues	Not developed	
16. Global climate regulation	Hannah Peck + Ana Rodrigues	Quick literature review (no expert contribution or review)	
17. Unknown benefits/processes	Ana Rodrigues	Not developed	

ANNEX 4 - LITERATURE REVIEW "ECONOMICS OF BIODIVERSITY LOSS"

The research has been realized on the following search engines:

- Ecological Economics
- Nature
- Scopus
- Science
- Science direct
- Google scholar (succinctly)

A combination of key words was used during the research for each search engine:

- Biodiversity + Economic Loss
- Biodiversity + Economic Cost
- Biodiversity + Economic Valuation

From these key words, the articles including all the terms in the text and published between 2005-2008 (included) have been selected. A total of 132 documents have been gathered, comprising:

- 2005: 29 articles
- 2006: 36 articles
- 2007: 46 articles
- 2008: 13 articles
- In press: 8 articles

The selected articles come from different sources the main ones being:

- Agriculture, Ecosystems and Environment (5)
- Agricultural and Resource Economics Review (2)
- Biological Conservation (10)
- Coastal Engineering (2)
- Conservation Biology (6)

- Ecological economics, (21)
- Ecological Modeling (3)
- Environmental Resource Economics (4)
- Environmental Modeling and Software (2)
- Environmental Science and Policy (3)
- Forest Ecology and Management (3)
- Journal of Sustainable Forestry (5)
- Land-Use Policy (2)
- Marine Policy (4)
- Marine pollution bulletin (3)
- Nature (2)
- PLOS Biology (2)
- Science (14)

The selected articles have been divided into two groups:

- the articles about topics that concerns general issues (Table A2)
- the articles related to case-studies in specific areas (Table A3)

Authors

Linda Scholten, Emmanuelle Noirtin & Rudolf de Groot (Environmental Systems Analysis group, Wageningen University)

Table A2. Articles related to global concerns

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Antoci, A., Borghesi, S. and Russu, P.	Biodiversity and economic growth: Trade-offs between stabilization of the ecological system and preservation of natural dynamics	*Ecological Modelling*	Vol. 189 Issues 3-4	2005	333-346
Antoci, A., Borghesi, S. and Russu, P. (2)	Interaction between economic and ecological dynamics in an optimal economic growth model	*Nonlinear Analysis*	Vol. 63 Issues 5-7	2005	389-398
Appleton, A.F.	Sustainability: A practitioner's reflection	*Technology in Society*	Vol. 28 Issues 1-2	2006	03-18
Balmford, A., Bennun, L., Brink ten, B., Cooper, D., Côte, I.M., Crane, P., Dobson, A., Dudley, N., Dutton, I., Green, R.E., Gregory, R.D., Harrison, J., Kennedy, E.T., Kremen, C., Leader-Williams, N., Lovejoy, T.E., Mace, G., May, R., Mayaux, P., Morling, P., Phillips, J., Redford, K., Ricketts, T.H., Rodríguez, J.P., Sanjayan, M., Schei, P.J., Jaarsveld van, A.S., and Walther, B.A.	The Convention on Biological Diversity's 2010 Target	*Science*	Vol. 307	2005	212-213
Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S, Cramer, L.A., Stoms, D.M., Kennedy, C.J., Dael,Kappel, C.V., Perillo, G.M.E., and Reed, D.J.	Coastal Ecosystem-Based Management with Nonlinear Ecological Functions and Values	*Science*	Vol. 319	2008	321-323
Baumgartner, S., Becker, C., Faber, M. and Manstetten, R.	Relative and absolute scarcity of nature. Assessing the roles of economics and ecology for biodiversity conservation	*Ecological economics*	Vol. 59 Issue 4	2006	487-498

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Baumgärtner, S., Becker, C., Faber, M. and Manstetten, R.	Relative and absolute scarcity of nature. Assessing the roles of economics and ecology for biodiversity conservation	*Ecological Economics*	Vol. 59	2006	487-498
Beaumont, N.J., Austen M.C., Mangi, S.C. andTownsend, M.	Economic valuation for the conservation of marine biodiversity	*Marine pollution bulletin*	Vol. 56 Issue 3	2008	386-396
Beaumont, N.J., Austen,M.C., Atkins,J.P., Burdon, D., Degraer, S., Dentinho, T.P., Derous,S., Holm, P., Horton, T., Lerland van, E., Marboe, A.H., Starkey, D.J., Townsend, M. and Zarzycki, T.	Identification, definition and quantification of goods and services provided by marine biodiversity: Implications for the ecosystem approach	*Marine pollution bulletin*	Vol. 54	2007	253-265
Bohanec, M., Messéan, A., Scatasta, S., Angevin, F., Griffiths, B., Henning Krogh, P., Znidarsic, M., Dzeroski, S.	A qualitative multi-attribute model for economic and ecological assessment of genetically modified crops	*Ecological Modelling*		in press	
Bohringer, C. and Loschel, A.	Computable general equilibrium models for sustainability impact assessment: Status quo and prospects	*Ecological Economics*	Vol. 60 Issue 1	2006	49-64
Born, W.,Rauschmayer, F. and Brauer, I.	Economic evaluation of biological invasions - survey	*Ecological Economics*	Vol. 55 Issue 3	2005	321-336
Boyd, J. and Banzhaf, S	What are ecosystem services? The need for standardized environmental accounting units	*Ecological Economics*	Vol. 63	2007	616-626
Brauer, I., Mussner, R., Marsden, K., Oosterhuis, F., Rayment, M., Miller, C. and Dodokova , A.	The use of market incentives to preserve biodiversity. Final Report. A Project Under the Framework Contract for Economic Analysis ENV.G.1/FRA/2004/0081.	*Eco-Logic*		2006	
Brosi, B.J., Daily, G.C., Shih, T.M., Oviedo, F. and Duran, G.	The effects of forest fragmentation on bee communities in tropical countryside	*Journal of Applied Ecology*		2007	
Butler, S.J., Vickery, J.A. and Norris, K.	Farmland Biodiversity and the Footprint of Agriculture	*Science*	Vol. 315	2007	381-384

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Cabeza, M. and Moilanen, A.	Replacement cost: A practical measure of site value for cost-effective reserve planning	*Biological Conservation*	Vol. 132 Issue 3	2006	336-342
Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R. and Hyde, T.	Valuing the diversity of biodiversity	*Ecological Economics*	Vol. 58 Issue 2	2006	304-317
Costanza, R.	Enough is enough	*Nature*	Vol. 439	2006	789
Dalle, S.P, Blois de, S., Caballero, J. and Johns, T.	Integrating analysis of local land-use regulations, cultural perceptions and land-use/ land-cover data for assessing the success of community-based conservation	*Forest Ecology and Management*	Vol. 222 Issues 1-3	2006	370-383
Defra	An introductory guide to valuing ecosystem services	*Department for Environment Food and Rural Affairs*		2007	
Díaz, S., Fargione, J., Stuart Chapin, F.and Tilman, D.	Biodiversity loss threatens human well-being	*PLoS Biol*	Vol. 4 Issue 8	2006	1300-1305
Díaz, S., Tilman, D., Fargione, J., Chapin, F.I., Dirzo, R., et al.	Biodiversity regulation of ecosystem services-Ecosystems and human wellbeing: Current state and trends: Findings of the Condition and Trends Working Group.	*In: Hassan R, Scholes R, Ash N, editors.*		2005	297-329
Drechsler, M., Johst, K.,Ohl, C and Watzold, F.	Designing Cost-Effective Payments for Conservation Measures to Generate Spatiotemporal Habitat Heterogeneity	*Conservation Biology*	Vol 21 Issue 6	2007	1475-1486
Drechsler, M., Watzold, F., Johnst, K., Bergmann, H., Settele	A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes	*Biological Conservation*	Vol. 140 Issues 1-2	2007	174-186
Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z-I., Knowler, D.J., Léveque, C., Naiman, R.J., Prieur-Richard, A-H., Soto, D., Stiassny, M.L.J. and Sullivan, C.A.	Freshwater biodiversity: importance, threats, status and conservation challenges	*Biological Reviews*	Vol. 81 Issue 2	2006	163-182

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Dymond, J.R., Ausseil, A-G.E. and McC. Overton, J.	A landscape approach for estimating the conservation value of sites and site-based projects, with examples from New Zealand	*Ecological Economics*		in press	
Emerton, L., Bishop, J. and Thomas, L.	Sustainable financing of protected areas: a global review of challenges and options	*IUCN*		in press	
Eppink, F.V and Bergh van den, J.C.J.M.	Ecological theories and indicators in economic models of biodiversity loss and conservation: A critical review	*Ecological Economics*	Vol. 61 Issues 2-3	2007	284-293
Ferraro, P.J. and Kiss, A.	Direct payments to conserve biodiversity	*Science*	Vol. 298	2002	1718-1719
Fisher, B., Costanza, R., Turner, R.K. and Morling. P.	Defining and classifying ecosystem services for decision-making	*Ecological Economics*		In press	
Fisher, J., Manning, A.D., Steffen, W., Rose, D.B., Daniell, K., Felton, A., Garnett, S., Gilna, B., Heinsohn, R., Lindenmayer, D.B., MacDonald, B., Mills, F., Newell, B., Reid, J., Robin, L., Scherren, K. and Wade, A.	Mind the sustainability gap	*TRENDS in Ecology and Evolution*	Vol. 22 Issue 12	2007	621-624
Foley, J.A., DeFries, R.,Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R.,Chapin, F.S., Coe, M.T.,Daily, G.C.,Gibbs, H.K.,Helkowski, J.H.,Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C.,Ramankutty, N. and Snyder, P.K.	Global Consequences of Land Use	*Science*	Vol. 309	2005	570-574
Gentry, B.S.	Emerging markets for Ecosystem Services: What did we learn?	*Journal of Sustainable Forestry*	Vol. 25 Issues 3-4,	2007	365-374
Gillson, L. and Hoffman, T.	Rangeland Ecology in a Changing World	*Science*	Vol. 315	2007	53-54
Grafton, R.Q., Kompas, T. and Hilborn, R.W.	Economics of Overexploitation Revisited	*Science*	Vol. 318	2007	318-1601

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Grafton, R.Q., Hilborn, R., Ridgeway, L., Squires, D., Williams, M., Garcia, S., Groves, T., Joseph, J., Kelleher, K., Kompas, T., Libecap, G., Lundin, C.G., Makino, M., Matthiasson, T., McLoughlin, R., Parma, A., San Martin, G., Satia, B., Schmidt, C-C., Tait, M., et al.	Positioning fisheries in a changing world	*Marine Policy*	Vol. 32 Issue 4	2008	630-634
Griffiths, G.J.K., Holland, J.M., Bailey, A. and Thomas, M.B.	Efficiency and economics of shelter habitats for conservation biological control	*Biological Control*	Vol. 45 Issue 2	2008	200-209
Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., and Briggs, J.M.	Global Change and the Ecology of Cities	*Science*	Vol. 319	2008	756-760
Heal, G.	Chapter 21 Intertemporal Welfare Economics and the Environment	*Handbook of Environmental Economics*	Vol. 3	2005	1105-1145
Heide van der, C.M., Bergh van den J.C.J.M. and Lerland van, E.C.	Extending Weitzman's economic ranking of biodiversity protection: combining ecological and genetic considerations	*Ecological Economics*	Vol. 55 Issue 2	2005	218-223
Hodgson, J.G., Montserrat-Martı, G., Tallowin, J., Thompson, K., Dıaz, S., Cabido, M., Grime, J.P., Wilson, P.J., Band, S.R., Bogard, A., Cabido, R., Caceres, D., Castro-Dıez, P., Ferrer, C., Maestro-Martınez, M., Perez-Rontome, M.C., Charles, M., Cornelissen, J.H.C., Dabbert, S., Perez-Harguindeguy, N., Krimly, T., Sijtsma, F.J., Strijker, D., Vendramini, F., Guerrero-Campo, J., Hynd, A., Jones, G., Romo-Diez, A., de Torres Espuny, L., Villar-Salvador, P. and Zak, M.R.	How much will it cost to save grassland diversity?	*Biological Conservation*	Vol. 122 Issue 2	2005	263-273
Holdren, J.P.	Science and Technology for Sustainable Well-Being	*Science*	Vol. 319	2008	425-434

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Holzkamper, A. and Seppelt, R.	Evaluating cost-effectiveness of conservation management actions in an agricultural landscape on a regional scale	*Biological Conservation*	Vol. 136 Issue 1	2007	117-127
Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., et al.	Effects of biodiversity on ecosystem functioning: A consensus of current knowledge	*Ecol Monogr*	Vol. 75	2005	3-35
Jackson, R.B., Jobbágy, E.G., Avissar, R., Roy, S.B., Barrett, D.J., Cook, C.W., Farley, K.A., Maitre le, D.C., McCarl, B.A., and Murray, B.C.	Trading Water for Carbon with Biological Carbon Sequestration	*Science*	Vol. 310	2005	1944-1947
Johann, E.	Traditional forest management under the influence of science and industry: the story of the alpine cultural landscapes	*Forest Ecology and Management*	Vol. 249 Issues 1-2	2007	54-62
Kessler, j.j., Rood, T., Tekelenburg T. and Bakkenes M.	Biodiversity and Socioeconomic Impacts of Selected Agro-Commodity Production Systems	*Journal of Environment and Development*	Vol. 16 Issue 2	2007	131-160
Knoke, T., Stimm, B., Ammer, C. and Moog, M.	Mixed forests reconsidered: A forest economics contribution on an ecological concept	*Forest Ecology and Management*	Vol. 213 Issues 1-3	2005	102-116
Leroux, A.D. and Creedy, J.	Optimal land conversion and growth with uncertain biodiversity costs	*Ecological Economics*	Vol. 61 Issues 2-3	2007	542-549
Lichtenfels, M., Kuppalli, R., Burtis, P., Lichtenfeld, M., Hovani, A. and Miyata, Y.	Improving Markets for Ecosystem Services	*Journal of Sustainable Forestry*	Vol. 25 Issues 3-4,	2007	337-364
Liu, J., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H., and Taylor, W.W.	Complexity of Coupled Human and Natural Systems	*Science*	Vol. 317	2007	1513-1516
Losey, J.E. and Vaughan, M.	The economic value of ecological services provided by insects	*Bioscience*	Vol. 56 Issue 4	2006	311-323

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Lovell, S.J., Stone, S.F and Fernandez, L.	The economic impacts of aquatic invasive species: a review of the literature	*Agricultural and Resource Economics Review*	Vol. 35 Issue 1	2006	195-208
Mace G, Masundire H, Baillie J, Ricketts T, Brooks T, et al.	Ecosystems and human well-being: Current state and trends: Findings of the Condition and Trends Working Group	*Biodiversity*		-2005	77–122
MacLeod, N.D. and McIvor, J.G.	Reconciling economic and ecological conflicts for sustained management of grazing lands	*Ecological Economics*	Vol. 56 Issue 3	2006	386-401
Marggraf, R.	Global Conservation of Biodiversity from an Economic Point of view	*Valuation and Conservation of Biodiversity*	part I	2005	3-21
Martín-López, B., Montes, C. and Benayas, J.	The non-economic motives behind the willingness to pay for biodiversity conservation	*Biological Conservation*	Vol. 139 Issues 1-2	2007	67-82
Martín-López, B., Montes, C. and Benayas, J.	Economic Valuation of Biodiversity Conservation: the Meaning of Numbers	*Conservation Biology*		in press	
Mayer, A.L. and Tikka, P.M.	Biodiversity conservation incentive programs for privately owned forests	*Environmental Science and Policy*	Vol. 9 Issues 7-8	2006	614-625
McCauley, D.J.	Selling out on nature	*Nature*	Vol. 443	2006	27-28
Mikkelson, GM., Gonzalez, A. and Peterson, GD.	Economic Inequality Predicts Biodiversity Loss	*PLoS ONE*	Vol. 2 Issue 5	2007	1-5
Moschella, P.S., Abbiati, M., Aberg, P., Airoldi, L., Anderson, J.M., Bacchiocchi, F., Bulleri, F., Dinesen, G.E., Frost, M., Gacia, E., Granhag, L., Jonsson, P.R., Satta, M.P., Sunderlof, A., Thompson, R.C. and Hawkins, S.J.	Low-crested coastal defense structures as artificial habitats for marine life: Using ecological criteria design	*Coastal Engineering*	Vol. 52 Issues 10-11	2005	1053-1071
Naidoo, R. and Adamowicz, W.L.	Modeling opportunity Costs of Conservation in Transitional Landscapes	*Conservation Biology*	Vol 20 Issue 2	2006	490-500

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Naidoo, R. and Iwamura, T.	Global-scale mapping of economic benefits from agricultural lands: Implications for conservation priorities	*Biological Conservation*	Vol. 146 Issues 1-2	2007	40-49
Naidoo, R. and Ricketts, T.H.	Mapping the Economics Costs and Benefits of Conservation	*PLOS Biology*	Vol. 4 Issue 11, e360	2006	2153-2164
Olson, L.J.	The economics of terrestrial invasive species: a review of the literature	*Agricultural and Resource Economics Review*	Vol. 35 Issue 1	2006	178-194
Pascual U. and Perrings C.	Developing incentives and economic mechanisms for in situ biodiversity conservation in agricultural landscapes	*Agriculture, Ecosystems and Environment*	Vol. 121 Issue 3	2007	256-268
Pascual, U. and Perrings, C.	Developing incentives and economics mechanisms for *in situ* biodiversity conservation in agricultural landscapes	*Agriculture, Ecosystems and Environment*	Vol. 121 Issue 3	2007	256-268
Perfecto, I., Vandermeer, J., Mas, A. and Soto Pinto, L.	Biodiversity, yield, and shade coffee certification	*Ecological economics*	Vol. 54 Issue 4	2005	435-446
Pierce, D.	Do we really care about Biodiversity?	*Environmental Resource Economics*	Vol. 37	2007	313-333
Polasky, S., Costello, C. and Solow, A.	Chapter 29 The Economics of Biodiversity	*Handbook of Environmental Economics*	Vol. 3	2005	1517-1560
Polasky, S., Nelson, E., Lonsdorf, E., Fackler, P. and Starfield, A.	Conserving species in a working landscape: land use with biological and economic objectives	*Ecological Applications*	Vol. 15 Issue 4	2005	1387-1401
Polomé, P., Marzetti, S. and Veen van der, A.	Economic and social demands for coastal protection	*Coastal Engineering*	Vol. 52 Issues 10-11	2005	819-840
Potts, M.D. and Vincent, J.R.	Spatial distribution of species populations, relative economic values, and the optimal size and number of reserves	*Environmental and Resource Economics*	Vol. 39	2008	91-112

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Rosales, J.	Economic Growth and Biodiversity Loss in an Age of Tradable Permits	*Conservation Biology*	Vol. 20 Issue 4	2006	1042 - 1050
Schnier, K.E.	Biological "hot spots" and their effect on optimal bioeconomic marine reserve formation	*Ecological Economics*	Vol. 55 Issue 4	2005	453-468
Schou, J.S., Tybirk, K., Lofstrom, P. and Hertel, O.	Economic and environmental analysis of buffer zones as an instrument to reduce ammonia loads to nature areas	*Land Use Policy*	Vol. 23 Issue 4	2006	533-541
Schrader, S. and Boning, M.	Soil formation on green roofs and its contribution to urban biodiversity with emphasis on Collembolans	*Pedobiologia*	Vol. 50 Issue 4	2006	347-356
Shanmuganathan, S., Sallis, P. and Buckeridge, J.	Self-organizing map methods in integrated modeling of environmental and economic systems	*Environmental Modeling and Software*	Vol. 21 Issue 9	2006	1247-1256
Simpson, R.D.	David Pearce and the economic valuation of biodiversity	*Environmental Resource Economics*	Vol. 37	2007	91-109
Spash, C.L., Urama, K., Burton, R., Kenyon, W., Shannon, P.and Hill, G.	Motives behind willingness to pay for improving biodiversity in water ecosystem: Economics, ethics and social psychology	*Ecological Economics*	2654	2006	
Stewart, R.R., and Possingham, H.P.	Efficiency, costs and trade-offs in marine reserve system design	*Environmental Modeling and Assessment*	Vol. 10	2005	203-213
Stokstad, E.	Forest Conservation: Learning to Adapt	*Science*	Vol. 309	2005	688-690
Storkey, J. and Cussans, J.W.	Reconciling the conservation of in-field biodiversity with crop production using a simulation model of weed growth and competition	*Agriculture, Ecosystems and Environment*	Vol. 122 Issue 2	2007	173-182

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Tilman, D., Polasky, S. and Lehman, C.	Diversity, productivity and temporal stability in the economies of humans and nature	*Journal of Environment Economics and Management*	Vol. 49	2005	405-426
Tisdell, C.	Linking policies for biodiversity conservation with advances in behavioral economics	*The Singapore economic review*	Vol. 50 Issue 1	2006	449-462
Treweek, J.R., Brown, C., and Bubb, P.	Assessing biodiversity impacts of trade: a review of challenges in the agriculture sector	*Impact Assessment and Project appraisal*	Vol. 24 Issue 4	2006	299-309
Turner, R.K. and Daily, G.C.	The Ecosystem Services Framework and Natural Capital Conservation	*Environmental and Resource Economics*	Vol. 39	2008	25–35
Ulbrich, K., Drechsler, M., Watzold, F., Johnst, K. and Settele, J.	A software tool for designing cost-effective compensation payments for conservation measures	*Environmental Modeling and Software*	Vol. 23 Issue 1	2008	122-123
Wallace, K.J.	Classification of ecosystem-services: Problems and solutions	*Biological Conservation*	Vol. 139	2007	235-246
Wallander, S.	The Dynamics of Conservation Financing: A Window into the Panamanian Market for Biodiversity Existence Value	*Journal of Sustainable Forestry*	Vol. 25 Issues 3-4,	2007	265-280
Weidema, B.P.	Using the budget constraint to monetarise impact assessment results	*Ecological Economics*		in press	
Wilcox, C. and Donlan, C.J.	Compensatory mitigation as a solution to fisheries bycatch-biodiversity conservation conflicts	*Frontiers in Ecology and the Environment*	Vol. 5 Issue 6	2007	325-331
Wilkie D, Morelli G, Demmer J, Starkey M, Telfer P, et al.	Parks and people: Assessing the human welfare effects of establishing protected areas for biodiversity conservation	*Conservation Biology*	Vol. 20	2006	247-249

AUTHORS	TITLE	SOURCE	VOLUME	DATE	PAGES
Worm, B., Barbier E.B., Beaumont, N., Emmett Duffy, J., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K.,Micheli, Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J. and Watson, R.	Impacts of Biodiversity Loss on Ocean Ecosystem Services	*Science*	Vol. 314	2006	787 - 790
Worm, B., Barbier E.B., Beaumont, N., Emmett Duffy, J., Folke, C., Halpern, B.S., Jackson, J.B.C., Lotze, H.K.,Micheli, Palumbi, S.R., Sala, E., Selkoe, K.A., Stachowicz, J.J. and Watson, R.	Response to Comments on "Impacts of Biodiversity Loss on Ocean Ecosystem Services"	*Science*	Vol. 316	2007	1285d

Table A3. Articles related to case-studies

AUTHORS	TITLE	*SOURCE*	VOLUME	DATE	PAGES
Adams, C., Seroe da Motta, R., Ortiz, R.A., Reid, J., Ebersbach Aznar, C. and Almeida Sinisgalli de, P.A.	The use of contingent valuation for evaluating protected areas in the developing world: Economic valuation of Morro do Diabo State Park, Atlantif Rainforest, Sao Paulo State (Brazil)	*Ecological Economics*		2007	
Asquith, N.M., Vargas, M.T. and Wunder, S.	Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia	*Ecological Economics*		in press	
Baral, N., Gautam, R., Timilsina, N. and Bhat, M.G.	Conservation implications for contingent valuation of critically endangered white-rumped vulture Gyps bengalensis in South Asia	*International Journal of Biodiversity Science and Management*	Vol. 3	2007	145-156
Berger, G., Kaechele, H. and Pfeffer, H.	The greening of the European common agricultural policy by linking the European-wide obligation of set-aside with voluntary agri-envrionmental measures on a regional scale	*Environmental Science and Policy*	Vol. 9 Issue 6	2006	509-524
Burnett, K., Kaiser, B. and Roumasset, J.	Economic lessons from control efforts for invasive species: *Miconia calvescens* in Hawaii	*Journal of Forest Economics*	Vol. 13 Issues 2-3	2007	151-167
Cousins, B., Hoffman, M.T., Allsopp, N. and Rohde, R.F.	A synthesis of sociological and biological perspectives on sustainable land use in Namaqualand	*Journal of Arid Environments*	Vol. 70 Issue 4	2007	834-846
Donald, P.F., Sanderson, F.J, Burfield, I.J. and Bommel van, F.P.J	Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990-2000	*Agriculture, Ecosystems and Environment*	Vol. 116 Issues 3-4	2006	189-196
Echeverria, C., Coomes, D.A, Hall, M., Newton, A.C	Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile	*Ecological Modelling*	Vol. 212 Issues 3-4	2008	439-449

AUTHORS	TITLE	*SOURCE*	VOLUME	DATE	PAGES
Eppink, F.V., Rietveld, P., Bergh Van den, J.C.J.M, Vermaat, J.E., Wassen, M.J. and Hilferink, M.	Internalizing the costs of fragmentation and nutrient deposition in spatial planning: Extending a decision support tool for the Netherlands	*Land Use Policy*		in press	
Etten van, J.	Molding maize: the shaping of a crop diversity landscape in the western highlands of Guatemala	*Journal of Historical Geography*	Vol. 32 Issue 4	2006	689-711
Franklin, K.A., Lyons, K., Nagler, P.L., Lampkin, D., Glenn, E.P., Molina-Freaner, F., Markow, T. and Huete, A.R.	Bufflegrass (*Pennisetum ciliare*) land conservation and productivity in the plains of Sonora, Mexico	*Biological Conservation*	Vol. 127 Issue 1	2006	62-71
Hänninen, R. and Maarit I. Kallio, A.	Economic impacts on the forest sector of increasing forest biodiversity conservation in Finland	*Silva Fennica*	Vol. 41 Issue 3	2007	507-523
Hunt, C.	Economy and ecology of emerging markets and credits for bio-sequestered carbon on private land in tropical Australia	*Ecological Economics*		2007	
Kerkvliet, J. and Langpap, C.	Learning from endangered and threatened species recovery programs: A case-study using U.S. Endangered Species Act recovery scores	*Ecological Economics*	Vol. 63 Issues 2-3	2007	499-510
Kettunen, M. and ten Brink, P.	Value of Biodiversity. Documenting EU examples where biodiversity loss has led to the loss of ecosystem services. Final report for the European Commission ENV.G.1/FRA/2004/0081	*Institute for European Environmental Policy (IEEP)*		2006	131pp
Maarit I. Kallio, A., Moiseyev, A. and Solberg, B.	Economic impacts of increased forest conservation in Europe: a forest sector model analysis	*Environmental Science and Policy*	Vol. 9 Issue 5	2006	457-465
Meyerhoff, J. and Dehnhardt, A.	The European Water Framework Directive and Economic Valuation of Wetlands: the Restoration of Floodplains along the River Elbe	*European Environment*	Vol. 17	2007	18-36
Miyata, Y.	Markets for Biodiversity: Certified Forest Products in Panama	*Journal of Sustainable Forestry*	Vol. 25 Issues 3-4,	2007	281-307

AUTHORS	TITLE	*SOURCE*	VOLUME	DATE	PAGES
Naidoo, R. and Adamowicz, W.L.	Economic benefits of biodiversity exceed costs of conservation at an African rainforest reserve	*PNAS*	Vol. 102 Issue 46	2005	16712-16716
Ovetz, R.	The bottom line: An investigation of the economic, cultural and social costs of high seas industrial longline fishing in the Pacific and the benefits of conservation	*Marine Policy*	Vol. 31 Issue 2	2007	217-228
Pauleit, S., Ennos, R. and Golding Y.	Modeling environmental impacts of urban land use and land cover change- a study in Merseyside, UK	*Landscape and Urban Planning*	Vol. 71 Issues 2-4	2005	295-310
Rangel, T.F.L.V.B.	Human development and biodiversity conservation in Brazilian Cerrado	*Applied Geography*	Vol. 27 Issue 1	2007	14-27
Rogers, S.I. and Greenaway, B.	A UK perspective on the development of marine ecosystem indicators	*Marine Pollution Bulletin*	Vol. 50 Issue 1	2005	9-19
Rondinni, C. and Boitani, L.	Systematic Conservation Planning and the Cost of Tackling Conservation Conflicts with Large Carnivores in Italy	*Conservation Biology*	Vol. 21 Issue 6	2007	1455-1462
Ruggerio, A., Céréghino, R., Figuerola, J., Marty, P. and Angélibert, S.	Farm ponds make a contribution to the biodiversity of aquatic insects in a French agricultural landscape	*Comptes Rendus Biologies*	Vol. 331 Issue 4	2008	298-308
Singh, A. and Mee, L.	Examination of policies and MEAs commitments by SIDS for sustainable management of the Caribbean Sea	*Marine Policy*	Vol. 32 Issue 3	2008	274-282
Soderqvist, T., Eggert, H., Olsson, B. and Soutukorva, A.	Economic valuation for sustainable development in the Swedish Coastal Zone	*Bioone*	Vol. 34 Issue 2	2005	169-175
Strange, N., Jacobsen, J.B., Thorsen, B.J. and Tarp, P.	Value of Money: protecting endangered species on Danish heathland	*Environmental Management*	Vol. 40	2007	761-774
Strange, N., Rahbek, C., Jepsen, J.K. and Lund, M.P.	Using farmland prices to evaluate cost-efficiency of national versus regional reserve selection in Denmark	*Biological Conservation*	Vol. 128 Issue 4	2006	455-466

AUTHORS	TITLE	*SOURCE*	VOLUME	DATE	PAGES
Strijker, D.	Marginal Lands in Europe- causes of decline	*Basic and Applied Ecology*	Vol. 6 Issue 2	2005	99-106
Subade, R.F.	Mechanisms to capture economic values of marine biodiversity: the case of Tubbatahe Reefs UNESCO World Heritage Site, Philippines	*Marine Policy*	Vol. 31	2007	135-142
Sultanian E. and Beukering van, P.J.H.	Economics of migratory birds: Market creation for the protection of migratory birds in the Inner Niger Delta (Mali)	*Human dimensions of wildlife*	Vol. 13 Issue 1	2008	3-14
Verboom, J., Alkemade, R., Klijn, J., Metzger, M.J. and Reijnen, R.	Combining biodiversity modeling with political and economic development scenarios for 25 EU countries	*Ecological Economics*	Vol. 62 Issue 2	2007	267-276
Wallander, S., Lauterbach, S., Anderson, K., Chou, F., Grossman, J.M. and Catherine Schloegel, C.	Existing markets for ecosystem services in the Panama Canal Watershed	*Journal of Sustainable Forestry*	Vol. 25 Issues 3-4,	2007	308-336
Watzold, F. and Schwerdtner, K.	Why be wasteful when preserving a valuable resource? A review article on the cost-effectiveness of European biodiversity conservation policy	*Biological Conservation*	Vol. 123 Issue 3	2005	327-338
Westhoek, H.J, Berg van den, M and Bakkes, J.A.	Scenario development to explore the future of Europe's rural areas	*Agriculture, Ecosystems and Environment*	Vol. 114 Issue 1	2006	7-20

ANNEX 5 - INVENTORY OF RESEARCH ORGANIZATIONS, PROGRAMS AND PROJECTS DEALING WITH "ECONOMICS OF BIODIVERSITY LOSS"

As a contribution to the Scoping the Science project, an inventory was made of the main current research programs and organisations dealing with "economics of biodiversity loss" whereby 5 categories were distinguished:

- 0 - Main networks/websites
- 1 - Multi-lateral institutions, programs, treaties, etc
- 2 - Government supported initiatives
- 3 - NGO's
- 4 - Universities / Research organizations & programs
- 5 - Business supported initiatives

The inventory was carried out by using existing and well-known websites as a starting point (category 0 in below table). Most of these websites have a section with "links" and by following these in a systematic way it is possible to obtain a good overview of the main organizations and programs.

Keywords used for the search were the link between "biodiversity" and "economics / economic loss / economic cost / economic valuation".

When going "deeper" into the various organisations and programs, many links are found to more specific subjects (e.g. programs, projects and organisations focusing on specific ecosystems (e.g. watershed-services, coral reefs, forests, etc), services or regions). Within the timeframe of this project it was not possible to explore these links in any detail but it would be very interesting and useful to carry out a full "network-analysis" as one of the future follow-up activities.

Below, a summary is given of the main organisations and programs, organised by type of organisation. Per category a distinction is made between international and national organisations & programs. Within these categories, organisations are listed in alphabetical order for easy reference.

This overview does not aim to be exhaustive, however in combination with the links given under the "Main Networks & Websites" (category 0) it should enable quick and easy access to the main organisations and programs on this topic. In the future, some resources could be directed to identifying relevant networks at national level.

NOTE: In case you miss an (important) organization or program, please send an email to dolf.degroot@wur.nl]

0 - Main (long-term) networks / websites / databases

(are also listed under respective country or organization for easy cross-reference)

Name	*Website*	*Additional information*
Biodiversity Economics	www.biodiversityeconomics.org	Website from IUCN and WWF
Ecosystem Service Database (ESD)	http://esd.uvm/edu	UVM – Gund inst. Ecol. Economics
Ecosystem Services	www.esa.org/ecoservices	Ecological Society of America
Ecosystem Services Database	esd-worldbank.org/eei/	World Bank Email: EnvEc@worldbank.org
Ecosystem Services project (Australia)	www.ecosystemservicesproject.org	CSIRO www.cse.csiro.au/ecoservices
Ecosystem Valuation	www.ecosystemvaluation.org	Univ. Maryland
EnValue	www.epa.nsw.gov.au/envalue/ (start 1995)	(EPA New South Wales**)**
EVRI	www.evri.ca (since 1997)	Env. Canada & EPA-US (also World Bank & EC are involved)
MILLENNIUM ECOSYSTEM ASSESSMENT(2001-2005)	www.maweb.org	
Natural Capital Project	www.naturalcapitalproject.org	Stanford University
Nature Valuation & Financing Network (NV&F)	www.naturevaluation.org	

1 – Multi-lateral Ints. / Org. / Programs / Treaties / etc.

Name	*Website*	*Additional information*
CIFOR / ICRAF	www.cifor.cgiar.org	• Environmental Services and Sustainable Use of Forests-program • RUPES program (Rewarding Upland Poor for Environmental Services)
Convention on Biological Diversity - CBD	www.biodiversity.org or www.cbd.int	• Ecosystem Approach
European Commission	ec.europa.eu	• Env. Economics [program] ec.europa.eu/environment/enveco • EXTERNE – calculating the costs of externalities ec.europa.eu/research/
European Environment Agency	www.eea.europa.eu	• call for evidence on economics of biodiversity loss • EURECA (European Ecosystem Assessment)

European Investment Bank	www.eib.org	• Topic: Biodiversity (esp. irt health loss); collaborate with IUCN
FAO	www.fao.org	• Financing Strategies for Sust. Forest Management (together with IUCN and CCAD) (www.fao.org/forestry/mecanismosfinancieros)
MILLENNIUM ECOSYSTEM ASSESSMENT (2001-2005)	www.maweb.org	
Ramsar Convention	www.ramsar.org	• Guidelines for Wetland Valuation (in collab. with CBD)
UN CEEA	unstats.un.org/unsd/envaccounting/ceea	• UN Committee of Experts on Environmental Economic Accounting
UNDP - GEF	www.undp.org/gef	• Biodiversity portfolio (240 projects; Gustavo Fonseca)
UNEP	www.unep.ch	• UNEP Finance Initiative (www.unepfi.org) • UN Committee of Experts on Env. Economic Accounting (UNCEEA) • WCMC (www.unep.wcms.org)
UNESCO	www.unesco.org	• WWF UNESCO project about Financing Mechanisms for Protected Areas • (in Argentina) : www.vidasilvestre.org.ar
UN-FCCC	unfccc.int	• Impact CC on ecosystem services
WORLD BANK	www.worldbank.org	• Ecosystem Services Database – World Bank (esd-worldbank.org/eei/) • + environmental economics, environmental valuation, PES

2 – Government supported initiatives

[see also section 4: univ. and research programs]

Country	*Name*	*Website*	*Additional information*
Australia	**CSIRO Ecosystem Services project**	www.cse.csiro.au/ecoservices *[see also: main Networks]*	
Australia	**EnValue**	www.epa.nsw.gov.au/envalue/ (start 1995)	• EPA New South Wales
Canada	**EVRI**	www.evri.ca (since 1997)	• Env. Canada & EPA-US (also World Bank & EC are involved)

Canada	**IDRC** (Int. Dev. Research Centre)	www.idrc.ca	• established EEPSEA (Economy & Env. Program for SE Asia) in 1993
Denmark	**DANIDA**		• supported by Royal Danish Min. of Foreign Affairs
Germany	**GTZ**	www.gtz.de	
Netherlands	**MNP - Netherlands Environmental Assessment Agency**	www.mnp.nl	
UK	**DEFRA**	www.defra.goc.uk/environment	• Ecosystem Services Project
USA	**EPA**	www.epa.gov	• Env. Economics Publ • Atlas of Ecosystem Services
USA	**National Research Council**	www.nationalacademies.org	• Committee on Assessing and Valuing Ecosystem Services

3 - NGO's (organizations and programs)

International

Name	*Website*	*Additional information*
Association of Environmental and Resource Economists	www.aere.org	
Conservation Finance Alliance (CFA)	www.conservationfinance.org	• Started in 2002, last update 2006 - seems not very active anymore • Joint project of 12 org., among others: CI, IUCN, WWF, World Bank etc
Conservation International (CI)	www.conservation.org	• mapping and valuing ecosystem services
Defenders of Wildlife	www.defenders.org	• Economic analysis of benefits and costs of biodiversity concervation
DIVERSITAS	www.diversitas-international.org	
East-West Centre	www.eastwestcenter.org	• Ecoservices program (Charles Perrings)

European Centre for Nature Conservation (ECNC)	www.ecnc.nl	• Biodiversity for Sustainability program / Business & Biodiv.
Forest Trends	www.forest-trends.org)	• see Katoomba Group
Guyana Shield program	www.guianashield.org	• development of payment mechanisms for ecosystem services
IEEP (Inst. European Env. Policy)	www.ieep.eu	
IIED	www.iied.org	• Env. Economics Program (EEP) • Support el. Newsletter on Payments for Watershed Services (www.flowsonline.net)
IISD (Int. Inst. For Sust. Development)	www.iisd.org)	• Funded by Env. Canada in 1990 • Topics among others "Valuing Natural Capital"
ISEE (Int. Society for Ecological Economics)	www.ecoeco.org or www.ecologicaleconomics.org	
IUCN	www.iucn.org	Support many activities irt Ecosystem Services and Economics: • Economics Division (since 1998) • Business and Biodiversity Program (since 2000) (cms.iucn.org/etc ..) • Global Economics and Env. Program (GEEP) > 2009 (cms.iucn.org/etc ..) • IUCN-Regional Env. Economics Program Asia • WANI (PES Watershed services) (www.iucn.org/themes/wani) • CEM-Ecosystem Services • CEM-Ecosystem Approach
Katoomba Group	www.katoombagroup.org [see also Forest Trends]	• Ecosystem Market Place (ecosystemmarketplace.com)
Resources for the Future	www.rff.org	
Restoring Natural Capital (RNC) Alliance:	www.rncalliance.org	
SANDEE (South Asian Network for Dev. and Env. Economics)	www.sandeeonline.org	
Society for Ecological Restoration (SER).	www. ser.org	
WCMC (see also UNEP)	www.unep.wcmc.org	• mapping ecosystem services
Wetlands International	www.wetlands.org	• Biorights program

World Resources Inst (WRI)	(www.wri.org)	• People and Ecosystems Program www.wri.org/ecosystems/ecosystem-services
WWF	www.worldwildlife.org	Many activities eg: • (Centre for) Conservation Finance (www.worldwildlife.org/conservationfinance) • WWF Macro-economics Program Office www.panda.org/etc.. • Meta-analysis of wetland values … • Natural Capital Project (with Stanford Univ) • WWF UNESCO project about Financing Mechanisms for Protected Areas • Argentina: www.vidasilvestre.org.ar

National

> **NB:** *many national organizations were found but the list is rather incomplete and unbalanced and is therefore not included here (can be provided at request); an exception is made for the Ecological Society of America which made an interesting website with Toolkits on Ecosystem Services: www.esa.org/ecoservices*

4 – Universities / Research programs (& assessments)

International

Name	*Website*	*Additional information*
EVE concerted action	www.landecon.cam.ac.uk/eve	
RUBICODE	www.rubicode.net	• EU funded project (2007-2009) • "Rationalising Biodiversity Conservation in Dynamic Ecosystems"

National *[STILL VERY INCOMPLETE]*

Country	*Name*	*Website*	*Additional information*
Australia	**Australian Nat. University Economics and Environment Network**	een.au.edu.au	• Australian Nat. University

Italy	**FEEM**	www.feem.it	• a.o. host of global network of environmental economists
Netherlands	**IVM**-VU	www.ivm.falw.vu.nl	• PREM (Poverty Reduction and Env. Management) www.premonline.nl
Netherlands	**Tilburg Univ**	www.tilburguniversity.nl	
Netherlands	**WUR** (Wageningen Univ. & Research Centre)	www.wur.nl	• SELS-program (www.ecosystemservices.nl): research program on Ecosystem & Landscape Services • Wageningen International (www.wi.wur.nl): many projects
South Africa	**Pretoria Univ** .	www.up.ac.za	• **CEEPA** (Center for Env. Economics and Policy in Africa) (www.ceepa.co.za)
Sweden	**Beijer** Int. Inst. of Environmental Economics	www.beijer.kwa.se	
Sweden	**Gothenborg Univ.**	www.gu.se	• Env. Economics Unit • Env. For Development Initiative (EfD) (www.efdinitiative.org) • supported by SIDA
Sweden	**Stockholm Univ**		• Dept. Systems Ecology
UK	**CSERGE** Centre for Social & Economic Research) UEA	www.uea.ac.uk/cserge	
UK	**Green Economics Inst**	www.greeneconomics-org.uk	• Center for Env. Management
UK	**Univ. Oxford**	www.ox.ac.uk	• Green College
USA	**Gund Inst. Ecological Economics-**, UVM	www.uvm.edu	• Various projects, including MIMES; the ***Ecosystem Service Database (ESD)*** (esd.uvm/edu) • to be linked to ARIES-project (an interactive data base & DSS-tool), and the EcoValue Project (ecovalue.uvm.edu/evp)

USA	**Maryland, Univ**	www.umd.edu	• **Ecosystem Valuation** www.ecosystemvaluation.org
USA	**Stanford Univ.**		• Natural Capital Project (www.naturalcapitalproject.org) (G.Daily et al),together with The Nature Conservancy & WWF • NatCap Network & Natural Capital Database • InVEST – tool to model value of ES and Trade-offs
USA	**Wyoming**, **Univ** Dept. of Economics & Finance	business.uwyo.edu/econfin	

5 - Business Supported Init. (Banks/ Consultants, etc)

Here only a few examples are listed but there are many more private organizations becoming increasingly involved in economics of biodiversity, and biodiversity loss

Name	*Website*	*Additional information*
SHELL	www.shell.com	• "Building Biodiversity Business" Publication together with IUCN – 2008)
TripleE	www.tripleee.nl	• Valuation Studies & Payment mechanisms
Economics for the Environment Consultancy Ltd	www.Eftec.co.uk	• Exist since 1992, set up UKNEE (in 2004)

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ANNEX 6 - KEY INTERNET RESOURCES ON BIODIVERSITY AND ECOSYSTEM SERVICES

General: Biodiversity and Ecosystem Services

Key institutions, groups and experts

- Center for Agri-Environmental Research, (University of Reading) – The biodiversity and ecosystem services research group carries out rigorous research on the linkages between agricultural land-use, biodiversity, ecosystem function and service provision and how respond to global change and their value to man. http://www.reading.ac.uk/caer/theme_services.html
- Defenders of Wildlife – Defenders' Conservation Economics Program focuses on objective and transparent economic analysis of the benefits and costs of biodiversity conservation design of economic incentives for wildlife conservation Ecosystem Services. http://www.defenders.org/programs_and_policy/science_and_economics/conservation_economics/index.php
- Defra's Ecosystem Services Project (UK)- Defra research project on ecosystem services aims to establish the basis for an ecosystems approach and how it may be used to make effective assessments of the benefits that the natural environment provides.

  http://www.ecosystemservices.org.uk/index.htm
- Ecology and Society - An electronic, peer-reviewed, multi-disciplinary journal devoted to the rapid dissemination of current research on intergrative science for resilince and sustainability (formerly Conseration Ecology).

  http://www.ecologyandsociety.org/
- Ecological Society of America and Union of Concerned Scientists. Ecosystem Services Toolkit – Tool for scientists to engage the public. Tool Kits have been completed on the following services: Pollination and Water Purification. Tool kit on Flood Damage Control is in development. http://www.esa.org/ecoservices/
- Ecosystem Services Project - The Ecosystem Services Project was initiated by CSIRO Sustainable Ecosystems and The Myer Foundation. The Project is a collaborative natural resource management project studying the services people obtains from their environments, the economic and social values inherent in these services and the opportunities that can arise from considering these services more fully in land management policies and decisions.http://www.ecosystemservicesproject.org/html/aboutus/index.htm
- Ecosystem Services Management & Restoration - Online Database on Ecosystem Services: Linking Valuation and Financing of Ecosystem Services to Sustainable Management. This database on the website of the Nature Valuation & Financing (NV&F) network (www.naturevaluation.org) provides a list of ongoing and completed case studies, initiatives and projects from around the world that link the valuation and financing of ecosystem services to sustainable management. http://topshare.wur.nl/naturevaluation/73766

- Millennium Ecosystem Assessment. - Designed by a partnership of UN agencies, international scientific organizations, and development agencies, this ongoing study assesses the capacity of ecosystems worldwide to provide goods and services that are important for human development. http://www.millenniumassessment.org/
- Natural Capital Project. 2006. "Toolbox." A joint venture among the Woods Institute for the Environment at Stanford University, the Nature Conservancy and the World Wildlife Fund is developing tools for modeling and mapping the delivery, distribution and economic value of ecosystem services and biodiversity. Online at: http://www.naturalcapitalproject.org/toolbox.html
- Global Biodiversity Outlook (GBO) - The GBO, is a periodic report on biological diversity, which provide a summary of the status of biological diversity and an analysis of the steps being taken by the global community to ensure that biodiversity is conserved and used sustainably, and that benefits arising from the use of genetic resources are shared equitably. http://www.cbd.int/gbo/
- Global Environment Outlook (GEO) - The Global Environment Outlook (GEO) project is the implementation of UNEP's mandate to keep the global environment under review. GEO is both a process and a series of reports, analyzing environmental change, causes, impacts, and policy responses. It provides information for decision-making, supports early warning and builds capacity at the global and sub-global levels. GEO is also a communication process that aims at raising awareness on environmental issues and providing options for action. http://unep.org/GEO/
- Global Environment Outlook Year Book Series - The GEO Year Book series, produced annually by the United Nations Environment Programme in collaboration with many world environment experts keeps abreast of environmental issues as they unfold (i.e. an annual survey of the changing global environment). The Year Book includes global and regional overviews. It also highlights the most significant environmental developments in the year. http://www.unep.org/geo/yearbook/yb2008/
- World Resources Institute – People and Ecosystems Programme. http://www.wri.org/ecosystems/ecosystem-services

Reports, publications and journal articles

- Ecosystem Services: A Primer - Ecological Society of America (ESA) – This online article highlights the importance of natural ecosystems and the services they produce upon which humans are dependent. http://www.actionbioscience.org/environment/esa.html
- FAO. 2003. Biodiversity and the ecosystem approach in agriculture, forestry and fisheries Proceedings of the satellite event on the occasion of the Ninth Regular Session of the Commission on Genetic Resources for Food and Agriculture - Rome 12 - 13 October 2002. http://www.fao.org/docrep/005/y4586e/y4586e00.htm
- FAO.2007. The State of food and agriculture: Paying farmers for environmental services. Rome. Italy.

  http://www.fao.org/docrep/010/a1200e/a1200e00.htm
- *Issues in Ecology*, "Ecosystem Services: Benefits Supplied to Human Societies by Natural Ecosystems, No. 2, Spring, 1997, Ecological Society of America. Available on ESA's website at: http://www.esa.org/science_resources/issues/FileEnglish/issue2.pdf
- Mainka, S., McNeely, J and B, Jackson. 2005. Depend on Nature: Ecosystem Services Supporting Human Livelihoods. *World Conservation Union*. Available at: http://www.undp.org/pei/pdfs/IUCN-DependonNature.pdf
- Millennium Ecosystem Assessment, 2005, Living Beyond Our Means: Natural Assets and Human Well-being. Statement of the Board.

  http://www.millenniumassessment.org/documents/document.429.aspx.pdf
- Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Wetlands and Water

Synthesis. World Resources Institute, Washington, DC.

http://www.millenniumassessment.org/documents/document.358.aspx.pdf

- Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

  http://www.millenniumassessment.org/documents/document.356.aspx.pdf

- Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Health Synthesis*. Island Press, Washington, DC. http://www.millenniumassessment.org/documents/document.357.aspx.pdf

- Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Desertification Synthesis. World Resources Institute, Washington, DC.

  http://www.millenniumassessment.org/documents/document.355.aspx.pdf

- Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Opportunities and Challenges for Business and Industry. http://www.millenniumassessment.org/documents/document.353.aspx.pdf

- Naidoo, R and Ricketts, T.H. 2006. Mapping the Economic Costs and Benefits of Conservation *PLoS Biology* 4, 11. (2006). Available at:

  http://biology.plosjournals.org/perlserv/?request=getdocument&doi=10.1371/journal.pbio.0040360

- RAND. Nature's Services: Ecosystems Are More Than Wildlife Habitat

  http://www.rand.org/scitech/stpi/ourfuture/NaturesServices/section1.html

- UNEP-WCMC. 2007. Biodiversity and Poverty Reduction. *The importance of biodiversity for ecosystem services.* Available at: http://www.unep-wcmc.org/latenews/Biodiversity%20and%20Poverty%20Reduction%20UNEP-WCMC.pdf

- World Bank, 2006. Where is the Wealth of Nations? Measuring Capital for the 21st Century. Washington, D.C.: The World Bank.

  http://siteresources.worldbank.org/INTEEI/214578-1110886258964/20748034/All.pdf

- World Resources Institute, 2005. *World Resources 2005: The wealth of the poor – managing ecosystems to fight poverty.* Washington DC: WRI.

  http://pdf.wri.org/wrr05_lores.pdf

- Giller, P., and G. O'Donovan, 2002. Biodiversity and Ecosystem Function: Do Species Matter? *Proceedings of the Royal Irish Academy.* 102B 3 129-139.

  http://www.ria.ie/cgi-bin/ria/papers/100066.pdf

Crop pollination

Key institutions, groups and experts

- CBD. Agricultural Biodiversity International Initiative for the Conservation and Sustainable Use of Pollinators. Established by the Fifth Conference of Parties to the Convention on Biological Diversity in 2000. Declared an urgent need to address the issue of worldwide decline of pollinator diversity? Available online at: http://www.cbd.int/programmes/areas/agro/pollinators.aspx

- Ecological Society of America (ESA). Pollination Fact Sheet: Pollination: an essential ecosystem service – Revealing secrets about the birds and the bees http://www.esa.org/ecoservices/comm/body.comm.fact.poll.html or http://www.esa.org/ecoservices/PollinationFactSheet.pdf

- European Pollinator Initiative (Hope Page) provides information on a wide range of activities which helps to conserve and manage pollinators to enhance the servives they provide. http://www.europeanpollinatorinitiative.org/

- IUCN. Task Force on Declinining Pollination of the Species Survival Commission of IUCN. Provides information on pollination as an ecosystem service in the conservation and sustainability of natural systems. http://www.uoguelph.ca/~iucn/
- North American Pollinator Protection Campaign (Home Page) http://www.nappc.org/
- The Arizona-Sonora Desert Museum, Forgotten Pollinators Campaign, Provides information on pollinators and links to key websites on this subject. www.desertmuseum.org/pollination/introduction.html
- USDA-ARS Bee Biology and Systematics Lab Research provide information based on various research projects on: the development and improvement of management systems for bee populations, biological studies of bees, plant-pollination systems, and bee biosystematics. www.LoganBeeLab.usu.edu/
- Apimondia Journal. APIMONDIA (International Federation of Beekeepers' Associations) promote scientific, ecological, social and economic apicultural development in all countries and the cooperation of beekeepers` associations, scientific bodies and of individuals involved in apiculture worldwide. Has various standing committees and publications on pollination. http://www.beekeeping.com/apimondia/index_us.htm

Reports, publications and journal articles

- Alexandra-Maria Klein et al., 2007. Importance of pollinators in changing landscapes for world crops. Proc. R. Soc. B (2007) 274, 303–313. Available online at:

  http://www.environnement.ens.fr/perso/claessen/e3/tpe/kleinetal2006.pdf
- C. Kremen, N. Williams, M.A. Aizen, B. Gemmill-Herren, G. LeBuhn, R. Minckley, L. Packer, S. Potts, T. Roulston, I. Steffan-Dewenter, D. Vazquez, R. Winfree, L. Adams, E. Crone, S. Greenleaf, T. Keitt, A.M. Klein, J. Regetz, T. Ricketts. Pollination and Other Ecosystem Services Produced by Mobile Organisms: A Conceptual Framework for the Effects of Land Use Change. *Ecology Letters* 10, 4: 299-314. (2007) Available at: http://www.blackwell-synergy.com/doi/abs/10.1111/j.1461-0248.2007.01018.x
- Daily, G.C, Ehrlich, P.R, and C.H. Sekercioglu. 2004. Ecosystem Consequences of Bird Declines. *Proceedings of the National Academy of Sciences, USA* 101, 52: 18042-18047. http://www.pnas.org/cgi/reprint/101/52/18042
- Eardly, C .Pollinator –A conservation priority. Science on Africa Magazine. Available online at http://www.scienceinafrica.co.za/pollinator.htm
- Goulson, D. 2003. Conserving wild bees for crop pollination. Food, Agriculture & Environment Vol.1 (1): 142-144. Available online at: http://www.sbes.stir.ac.uk/people/goulson/intjourfoodagricenvir2003.pdf
- Kearns, C.A., and D. Inouye. 1997. “Pollinators, Flowering Plants and Conservation Biology,” *BioScience* 47: 297–397. Available at: http://www.jstor.org/pss/1313191 [Accessed 25 April 2008]
- Kremen, C., Williams, N. M , and R. W. Thorp. 2002. Crop pollination from native bees at risk from agricultural intensification. PNAS. December 24, 2002. Vol. 99, no. 26http://www.pnas.org/cgi/reprint/262413599v1
- RAND. Pollination Services: No Food Without Them http://www.rand.org/scitech/stpi/ourfuture/NaturesServices/sec1_pollinators.html
- Ricketts, T.H 2004. Tropical Forest Fragments Enhance Pollinator Activity in Nearby Coffee Crops. *Conservation Biology* 18, 5: 1262-1271. (2004). http://www.blackwell-synergy.com/doi/abs/10.1111/j.1523-1739.2004.00227.x
- APIMONDIA Standing Commission on Pollination and Bee Flora -

http://www.beekeeping.com/apimondia/index_us.htm

- Foraging Behaviour Of Honeybee On Parental Lines Of Hybrid Cauliflower Pusa Selvakumar. http://www.apimondia.org/apiacta/slovenia/en/selvakumar.pdf
- The Variability Of Yield Structure Of Black Currant Cultivars (*Ribes nigrum* L) In Different Pollination Conditions Denisow http://www.apimondia.org/apiacta/slovenia/en/denisow.pdf
- Induction Feeding Of Honey-Bees To Improve *Actinidia deliciosa* Pollination Gardi. http://www.apimondia.org/apiacta/slovenia/en/gardi.pdf
- Honeybee Pollination In Sunflower Hybrid Seed Production Yadav. http://www.apimondia.org/apiacta/slovenia/en/yadav.pdf
- Reforestation With Major Bee Food Trees In El Salvador Sandker http://www.apimondia.org/apiacta/slovenia/en/sandker.pdf

Genetic diversity of crops and livestock

Key institutions, groups and experts

- Biodiversity International (Home Page)- Bioversity International is the world's largest international research organization dedicated solely to the conservation and use of agricultural biodiversity. http://www.bioversityinternational.org/. Carries research projects and publish extensively on genetic diversity of crops and livestock. http://www.bioversityinternational.org/Themes/Agricultural_Ecosystems/index.asp
- UK Agricultural Biodiversity Coalition (UKabc) Home Page:

  http://www.ukabc.org/ukabc3.htm
- The EU Biodiversity in Development Project

  www.wcmc.org.uk/biodev
- CGIAR agricultural biodiversity research centre

  www.cgiar.org/ipgri
- Open Directory Project links on Agricultural Biodiversity

  www.dmoz.org/science/environment/biodiversity/agricultural

Reports, publications and journal articles

- Daily, G.C, Ehrlich, P.R, Michener, C. and T.H. Ricketts. 2004. Economic Value of Tropical Forest to Coffee Production. *Proceedings of the National Academy of Sciences, USA* 101, 34: 12579-12582. Available at: http://www.pnas.org/cgi/reprint/101/34/12579
- Cromwell, E., Cooper, D and Mulvany, P. Agriculture, biodiversity and livelihoods: issues and entry points for development agencies. http://www.ukabc.org/odi_agbiod.pdf
- FAO. 1997. The state of the world’s plant genetic resources for food and agriculture. Rome: Italy. Available at: http://www.fao.org/WAICENT/FAOINFO/AGRICULT/AGP/AGPS/Pgrfa/pdf/swrfull.pdf
- ITDG. Agricultural biodiversity: farmers sustaining the web of life. Farmer’s World Network Briefing. http://www.practicalaction.org/docs/advocacy/fwn_bio-div_briefing.pdf
- Sustaining Agricultural Biodiversity and Agro-ecosystem Functions: Opportunities, incentives and approaches for the conservation and sustainable use of agricultural biodiversity in agro-ecosystems and production systems. Report of the FAO/CBD Agricultural Biodiversity Workshop, 2-4 December 1998, Rome. http://www.fao.org/WAICENT/FAOINFO/SUSTDEV/EPdirect/EPre0063.htm
- *In Situ* Agricultural Biodiversity Conservation Project A research project of the Intermediate

Technology Development Group (ITDG) and the Overseas Development Institute, UK (ODI). http://www.ukabc.org/abc.htm

- Developing Diversity: European NGOs' PGRFA activities Illustrated keynote paper presented to the 1998 European PGRFA Symposium, Braunschweig, 30th June 1998. By Patrick Mulvany, Intermediate Technology, ITDG. http://www.ukabc.org/bschweigNGO.htm#p
- Breeds of Livestock resource presented by the Department of Animal Science at Oklahoma State University - an educational and informational resource on breeds of livestock throughout the world. http://www.ansi.okstate.edu/breeds/
- Heifer Project International Information on livestock projects. http://www.heifer.org/

Marine Fisheries

Key institutions, groups and experts

- Communication Partnership for Science and the Sea: Marine Ecosystem Services - COMPASS is a collaborative effort to advance marine conservation science and communicate scientific knowledge to policymakers, the public, and the media.

  http://www.compassonline.org/marinescience/ecosystem.asp

  http://www.compassonline.org/
- FAO. Fisheries and Aquaculture Department (Hope Page) –Inland Aquatic Ecosystems and Coastalk and Marine Information

  http://www.fao.org/fishery/
- The FAO Fisheries and Aquaculture Department collects, analyzes and disseminates information on the sector operations (catch, production, value, prices, fleets, farming systems, employment). It also develops methodology, assesses and monitors the state of wild resources and elaborates resources management advice.

  http://www.fao.org/fishery/about
- World Resources Institute (WRI) 2006. The Value of coastal ecosystems

  http://earthtrends.wri.org/updates/node/118

Reports, publications and journal articles

- Duffy, J Emmett (Lead Author); Walker Smith (Topic Editor). 2006. "Marine ecosystem services." In: Encyclopedia of Earth. Eds. Cutler J. Cleveland (Washington, D.C.: Environmental Information Coalition, National Council for Science and the Environment). [First published in the Encyclopaedia of Earth October 10, 2006; Last revised October 16, 2006; Retrieved April 24, 2008]. http://www.eoearth.org/article/Marine_ecosystem_services
- Boris Worm, Edward B. Barbier, Nicola Beaumont, J. Emmett Duffy, Carl Folke, Benjamin S. Halpern, Jeremy B. C. Jackson, Heike K. Lotze, Fiorenza Micheli, Stephen R. Palumbi, Enric Sala, Kimberley A. Selkoe, John J. Stachowicz, and Reg Watson. Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science* 314, 5800: 787-790. (2006). http://www.sciencemag.org/cgi/content/abstract/314/5800/787
- *Issues in Ecology*, " The Role of Nearshore Ecosystems as Fish and Shellfish Nurseries, No. 11, Spring, 2003, Ecological Society of America. Available on ESA's website at: http://www.esa.org/science_resources/issues/FileEnglish/issue11.pdf

Timber from natural forests

Key institutions, groups and experts

- The FAO Forestry Web site provides literally thousands of pages of information, access to all of FAOs forest-related databases, detailed country profiles and links to documents on all aspects of forestry including new sites on forest fire, national forest programmes and forest reproductive material, among others. http://www.fao.org/forestry/en/

Information on specific forest products and services can be found at:

1. Wood energy http://www.fao.org/forestry/site/energy/en/
2. Pulp, paper and wood industries http://www.fao.org/forestry/site/harvesting/en/
3. Trade in forest products and services http://www.fao.org/forestry/site/trade/en/
4. Non-wood forest products http://www.fao.org/forestry/site/nwfp/en/

Nature-related outdoor activities

Reports, publications and journal articles

- RSPB. 2007. Wellbeing through Wildlife in the EU.

  http://www.birdlife.org/eu/pdfs/Wellbeing_EU_final_version_2mb.pdf

- Pretty J, Griffin M, Peacock J, Hine R, Sellens M and South N. 2005. *A countryside for health and wellbeing: The physical and mental health benefits of green exercise*. University of Essex, Colchester. Available online at: http://www.countrysiderecreation.org.uk/pdf/CRN%20exec%20summary.pdf http://195.92.230.85/Images/Hine_tcm2-30031.pdf

- Bird W. 2004. Natural Fit: Can Green Space and Biodiversity Increase Levels of

  Physical Activity? RSPB. The RSPB. Available at:

  http://www.rspb.org.uk/Images/natural_fit_full_version_tcm9-133055.pdf

- Collins, S. 2006. The Makuleke model for good governance and fair benefit sharing Steve Collins in IUCN. Policy Matters

  http://www.iucn.org/themes/ceesp/Publications/newsletter/PM14-Section%20IV.pdf

- Health Council of the Netherlands. 2004. The influence of nature on social, physical and psychological wellbeing. Part 1: review of current knowledge. Report to the Minister of Agriculture, Nature and Food. http://www.rmno.nl/files_content/Nature%20and%20Health.pdf

- UNEP / CMS Secretariat (2006). Wildlife watching and tourism. Bonn. Available at:

  http://www.cms.int/publications/pdf/wildlifewatching_text.pdf

Soil Quality

Key institutions, groups and experts

- FAO. Soil Biodiversity Portal

  http://www.fao.org/ag/AGL/agll/soilbiod/index_en.stm

Reports, publications and journal articles

- Coleman, D.C and Whitman, W.B. 2004. Linking species richness, biodiversity and ecosystem function in soil systems. http://www.fao.org/ag/AGL/agll/soilbiod/promotxt.stmInternational symposium on impacts of soil biodiversity on biogeochemical processes in ecosystems, Taipei, Taiwan, 2004

  http://www.umanitoba.ca/faculties/afs/soil_science/MSSS/Ecology/Graduate/Coleman%20and%20Whitman%202005.pdf

Biological control of crop pests and diseases

Key institutions, groups and experts

- HRDA. Natural Pest and Disease Control –Carries out research on biological control of pest, diseases and weeds. http://www.gardenorganic.org.uk/research/ir_pdw_man.php

Natural hazard regulation

Key institutions, groups and experts

- Dartmouth Flood Observatory keeps a Global Active Archive of Large Flood Events (1985-present) using information derived from a wide variety of news, governmental, instrumental, and remote sensing sources.
- International landslide centre is maintaining a worldwide landslide fatality database.
- The Hawai'i Solar Observatory has data on tropical storm paths worldwide stretching back to 1995, and a facility to calculate probability of strike given the co-ordinates of the location.
- Reliefweb records past natural disasters worldwide, including the extent of destruction and human suffering (but good maps of the area affected are not always available).
- UNEP-WCMC has compiled a *World Atlas of Seagrasses* (Green & Short 2003), and a *World Atlas of Coral Reefs* (Spalding et al. 2001). A *World Mangrove Atlas* (Spalding et al. 1997) is currently being updated.
- Global Lakes and Wetlands Database developed through a partnership between WWF and the University of Kassel in Germany (Lehner and Döll 2004).

Reports, publications and journal articles

- Bruijnzeel, L., van Dijk, A. I., van Noordwijk, M., Chappell, N. A., & Schellekens, J. 2007. *Tropical deforestation, people and flooding*. CIFOR. [online] http://www.worldagroforestry.org/water/downloads/bca_bruijnzeel.pdf
- Danielsen, F., M.K. Sørensen, M.F. Olwig, V. Selvam, F. Parish, N.D. Burgess, T. Hiraishi, V.M. Karunagaran, M.S. Rasmussen, L.B. Hansen, A. Quarto, and N. Suryadiputra. 2005. "The Asian Tsunami: A Protective Role for Coastal Vegetation." *Science*. 310(5748): 643. Online at: http://www.sciencemag.org/cgi/content/abstract/310/5748/643
- FAO, & CIFOR. 2005. *Forests and floods. Drowning in fiction or thriving on facts?* Bogor Barat, Indonesia: Center for International Forestry Research and Food and Agriculture Organization of the United Nations. Retrieved April 14, 2008, from http://www.fao.org/docrep/008/ae929e/ae929e00.htm
- MA. 2005. Regulation of Natural Hazards: Floods and Fires. http://www.millenniumassessment.org/documents/document.285.aspx.pdf
- NOAA. 2008. NOAA Pacific Tsunami Warning Center. http://www.prh.noaa.gov/pr/ptwc/

Medicinal Plant Species

Key institutions, groups and experts

- Biodiversity and Human Health: http://www.ecology.org/biod/
- CIFOR Livelihood Briefs: Available at: http://www.cifor.cgiar.org/Publications/Briefs/Livelihoods/
- CIFOR Forest and Health Initiative - http://www.cifor.cgiar.org/Research/Livelihoods/MainActivities/ForestHealth/introduction.htm

Reports, publications and journal articles

- Chivian E. et al., (2004) *Biodiversity: Its importance to human health*. The Center for Health and the Global Environment Harvard Medical School.

  Available online at: http://chge.med.harvard.edu/publications/documents/Biodiversity_v2_screen.pdf
- Colfer, C.J.P., Sheil, D and M. Kishi. Forests and Human Health: Assessing the Evidence *Center for International Forestry Research* (2006)

  Available at: http://www.cifor.cgiar.org/publications/pdf_files/OccPapers/OP-45.pdf
- Husselman, M. 2008. Beekeeping in Zambia. CIFOR Livelihood Brief 7. 4p.

  http://www.cifor.cgiar.org/publications/pdf_files/livebrief/livebrief0801.pdf
- Shackleton, S.; Kaschula, S.; Twine, W.; Hunter, L.; Holding-Anyonge, C.; Petheram, L. 2006. Forests as safety nets for mitigating the impacts of HIV/AIDS in southern Africa. CIFOR Livelihood Brief 4. 4p. http://www.cifor.cgiar.org/publications/pdf_files/livebrief/livebrief0604e.pdf
- Colfer, C.J.P.; Sheil, D.; Kishi, M.2006. Forests and human health: assessing the evidence. CIFOR Occasional Paper No. 45. Center for International Forestry Research (CIFOR), Bogor, Indonesia. 111p. Available:

  http://www.cifor.cgiar.org/publications/pdf_files/OccPapers/OP-45.pdf
- Colfer, C.J.P.; Sheil, D.; Kaimowitz, D.; Kishi, M.2006. Forests and human health in the tropics: some important connections**.** Unasylva 57(224): 3-10 online [URL] ftp://ftp.fao.org/docrep/fao/009/a0789e/a0789e02.pdf and http://www.fao.org/docrep/009/a0789e/a0789e00.htm
- Koppert, G.J.A, Dounias, E, Froment, A and Pasquet, P. 1993. Food consumption in the forest populations of the southern coastal area of Cameroon: YASSA – MVAE – BAKOLA. In: Hladik, C.M., Hladik, A., Linares, O.F. Pagezy, H., Semple, A. and Hadley, M. (1993). Tropical Forests, People and Food. Biocultural Interactions and Applications to Development. Man and the Biosphere Series, 13 (ParisÊ: UNESCO et CarnforthÊ: The Parthenon Publishing Group), pp. 295-310. Available online at: http://www.cifor.cgiar.org/publications/pdf_files/research/forests_health/17.pdf
- Shanley, P and Luiz, L. 2003. The Impacts of Forest Degradation on Medicinal Plant Use and Implications for Health Care in Eastern Amazonia. Vol. 53 No. 6 BioScience 573. June 2003. Available online at: http://www.cifor.cgiar.org/publications/pdf_files/research/forests_health/22.pdf

Non use values

Key institutions, groups and experts

- Amazon Conservation Team - work in partnership with indigenous people in conserving biodiversity, health, and culture in tropical America. http://www.amazonteam.org/publications.html
- Sacred Mountains Program is a program of the Mountain Institute's which mission it is to advance mountain cultures and preserve mountain environments. http://www.mountain.org/work/sacredmtns/index.cfm
- The Theme on Indigenous and Local Communities, Equity, and Protected Areas (TILCEPA), was set up in 2000 by the World Commission on Protected Areas (WCPA) and the Commission on

Environmental, Economic, and Social Policy (CEESP) of the World Conservation Union (IUCN). It advocates, in all countries, the recognition of community conserved and managed areas that are significant from biodiversity point of view, and the development of management partnerships with the communities resident in or surrounding official PAs. http://www.iucn.org/themes/ceesp/Wkg_grp/TILCEPA/TILCEPA.htm

- The IUCN Commission on Environmental, Economic and Social Policy (CEESP), is an inter-disciplinary network of professionals whose mission is to act as a source of advice on the environmental, economic, social and cultural factors that affect natural resources and biological diversity and to provide guidance and support towards effective policies and practices in environmental conservation and sustainable development. http://www.iucn.org/themes/ceesp/index.html
- The Forum on religioin and Ecology is the largest international multireligious project of its kind. With its conferences, publications, and website it is engaged in exploring religious worldviews, texts, and ethics in order to broaden understanding of the complex nature of current environmental concerns. http://environment.harvard.edu/religion/main.html
- IUCN WCPA Task Force on Cultural and Spiritual Values of Protected Areas (CSVPA) http://www.iucn.org/themes/wcpa/theme/values/values.html
- World Water Day. 2006. Facts and figures about water religions and beliefs. http://www.worldwaterday.org/page/442

Freshwater

Key institutions, groups and experts

- Ecological Society of America (ESA). Water purification fact sheet: Water purification an essential ecosystem services - Revealing secrets about natural water purification

  http://www.esa.org/ecoservices/comm/body.comm.fact.wate.html

  http://www.esa.org/ecoservices/WaterPurificationFactSheet.pdf
- Environment Canada: The Great Lakes Fact Sheet

  http://www.on.ec.gc.ca/wildlife/factsheets/fs_wetlands-e.html
- The hydrology module of the Natural Capital Project is being developed by Sue White at Cranfield University and Guillermo Mendoza at Stanford University.

Reports, publications and journal articles

- Alcamo, J., Vuuren, D. V., Ringler, C., Cramer, W., Masui, T., Alder, J., & Schulze, K. 2005. Changes in nature's balance sheet: model-based estimates of future worldwide ecosystem services. *Ecology and Society* 10: 19. [online] URL: http://www.ecologyandsociety.org/vol10/iss2/art19/
- *Issues in Ecology*, " Sustaining Healthy Freshwater Ecosystems. No. 10, Winter, 2003, Ecological Society of America. Available on ESA's website at

  http://www.esa.org/science_resources/issues/FileEnglish/issue10.pdf
- RAND. Nature's Services: Ecosystems Are More Than Wildlife Habitat: Watershed - http://www.rand.org/scitech/stpi/ourfuture/NaturesServices/sec1_watershed.html

Inland fisheries

Key institutions, groups and experts

- Environment Canada: The Great Lakes Fact Sheet

  http://www.on.ec.gc.ca/wildlife/factsheets/fs_wetlands-e.html
- FAO. Fisheries and Aquaculture Department (Hope Page) – nland Aquatic Ecosystems and Coastalk

and Marine Information: http://www.fao.org/fishery/

Reports, publications and journal articles

- *Issues in Ecology,* " Effects of Aquaculture on World Fish Supplies. No. 8, Winter, 2001, Ecological Society of America. Available on ESA's website at: http://www.esa.org/science_resources/issues/FileEnglish/issue8.pdf

Wild Meat

Key institutions, groups and experts

- Bushmeat Trade POSTnote, Parliamentary Office of Science and Technology. http://www.parliament.uk/documents/upload/POSTpn236.pdf
- Conservation Science Group, Imperial College. http://www.iccs.org.uk/
- ODI Wild Meat, Livelihoods Security and Conservation in the Tropics. This research project focus on the human and social dimensions of hunting wild meat for consumptive use in tropical forests. http://www.odi-bushmeat.org/#home_research
- UK Tropical Forest Forum Bushmeat Working Group. http://www.forestforum.org.uk/tradee.htm
- WCS Hunting and Wildlife Trade Program. World Conservation Society Hunting and Wildlife Trade Programme. http://www.wcs.org/international/huntingandwildlifetrade
- ZSL Bushmeat and Forests Conservation Programme. http://www.zsl.org/field-conservation/bushmeat-and-forest/

Reports, publications and journal articles

- DFID, Wildlife and Poverty Study, DFID Livestock and Wildlife Advisory Group London, 2002. http://www.dfid.gov.uk/pubs/files/wildlifepovertystudy.pdf
- Bowen-Jones, E. What are the impacts of the bushmeat trade on biodiversity, and what entry points can the EU most effectively use to reduce these. European Association of Zoos and Aquaria (EAZA). Available online at: http://www.eaza.net/download/summebj.PDF

Livestock

Key institutions, groups and experts

- Agriculture and Agri-food Canada: Biodiversity and Livestock Production

  http://www.agr.gc.ca/pfra/biodiversity/grazing_e.htm
- The UK agricultural biodiversity coalition (UK*abc*) www.ukabc.org
- The EU Biodiversity in Development Project. www.wcmc.org.uk/biodev
- CGIAR agricultural biodiversity research centre www.cgiar.org/ipgri
- Open Directory Project links on Agricultural Biodiversity. www.dmoz.org/science/environment/biodiversity/agricultural

Reports, publications and journal articles

- DFID, Wildlife and Poverty Study, DFID Livestock and Wildlife Advisory Group London, 2002. http://www.dfid.gov.uk/pubs/files/wildlifepovertystudy.pdf

Global Climate regulation

Key institutions, groups and experts

- CIFOR. TroFCCA: Tropical Forests & Climate Change Adaptation Project. http://www.cifor.cgiar.org/trofcca/_ref/home/index.htm
- The World Agroforestry Center (ICRAF). http://www.worldagroforestry.org/sea/Networks/RUPES/index.asp
- IUCN Forests and Climate Change Initiative. http://cms.iucn.org/about/work/programmes/forest/fp_our_work/fp_our_work_thematic/fp_our_work_fcc/index.cfm
- UNEP-WCMC. Biodiversity and Climate Change. http://www.unep-wcmc.org/Climate/

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