The Economics of Ecosystems and Biodiversity: the Quantitative Assessment

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Salman Hussain
United Nations Environment Programme
28 PUBLICATIONS 1,238 CITATIONS

Alistair McVittie
Scotland's Rural College
44 PUBLICATIONS 1,537 CITATIONS

Luke Brander
Brander Environmental Economics
95 PUBLICATIONS 3,717 CITATIONS

Alfred Wagtendonk
VanderSat - remote sensing scientists
47 PUBLICATIONS 771 CITATIONS

The Economics of Ecosystems and Biodiversity

The Quantitative Assessment

Final Report to the United Nations Environment Programme

Overall Study Coordinator:
Salman Hussain, SAC

Coordinating Lead Authors:
Alistair McVittie, SAC
Luke Brander, IVM

Lead Authors:
Olivier Vardakoulias, SAC
Alfred Wagtendonk, IVM
Peter Verburg, IVM
Rudolf de Groot, WUR
Rob Tinch, eftec

Contributing Authors:
Abdulai Fofana, SAC
Corinne Baulcomb, SAC
Sander van der Ploeg, WUR
Laurence Mathieu, eftec
Ece Ozdemiroglu, eftec
Zara Phang, eftec

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The biophysical modelling was coordinated by Ben ten Brink (PBL, 2010), executed by Michel Jeuken (PBL), with inputs from LEI (Wageningen University) and on the marine assessment from the University of British Columbia. Sander van der Ploeg and Rudolf de Groot coordinated the development of the TEEB database of primary valuation studies. Rose Hanley-Nickolls assisted the Quantitative Assessment team in reviewing case studies in this database.

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List of Acronyms

AKST	Agricultural Knowledge, Science and Technology
BAU	Business-As-Usual
CBA	Cost Benefit Analysis
CBD	Convention on Biological Diversity
CGE	Computable General Equilibrium
CL	Conventional Logging
CO_2	Carbon dioxide
DR	Discount Rate
ESS	Ecosystem Service
GIS	Geographical Information System
GMTI	Global Mean Temperature Increase
Ha	Hectare
HANPP	Human Appropriation of Net primary Product
MSA	Mean Species Abundance
MSY	Maximum Sustainable Yield
MVP	Minimum Viable Population
N	Nitrogen
NPP	Net Primary Product
NTFP	Non-Timber forest Products
PA	Protected Area
PPP	Purchasing Power Parity
REDD	Reduced Emissions from Deforestation and forest Degradation
RIL	Reduced Impact Logging
SCC	Social Cost of Carbon
SFM	Sustainable Forest Management
STM	Sustainable Timber Management
TEEB	The Economics of Ecosystems and Biodiversity
UNEP	United Nations Environment Programme
USD	United Stated Dollars
WSSD	World Summit on Sustainable Development
WTA	Willingness to Accept
WTP	Willingness To Pay

1 Introduction

1.1 Background to the Quantitative Assessment

This report is a response to evidence of a widespread loss of ecosystem quality at different scales (local, regional and global), a loss which is on-going and does not appear to be slowing down (Butchart *et al*., 2010). 2010 was the UN Year of Biodiversity. The most recent edition of the United Nations' Global Biodiversity Outlook estimates that 17% of known species are 'endangered' or 'critically endangered' and a further 27% are in a 'vulnerable' or 'near-threatened' condition; further, the 2010 goal of 'significantly reducing the rate of loss' has not been achieved (Convention on Biological Diversity, 2010). The first Global Biodiversity Outlook (Convention on Biological Diversity, 2001), finds that the rate of species extinctions is in the order of 100 to 1000 times faster than natural rates.

The Economics of Ecosystems and Biodiversity (TEEB) is a UNEP-funded project aimed at mainstreaming the valuation and evaluation of ecosystems and biodiversity. It is a response to this conservation agenda, focusing on the lack of the valuation of nature resulting in a failure to take account of the value of ecosystems and biodiversity in decision making (TEEB, 2008). The Convention on Biological Diversity (CBD) Conference of the Parties in Nagoya in October 2010 decided on a new goal and strategic plan. This report aims to contribute to this on-going process.

The overall aim of this report is to evaluate whether or not a range of policy interventions that affect ecosystems and biodiversity are *economically efficient*. A necessary condition for economic efficiency is that benefits exceed costs, i.e. net benefits should be positive. Testing for economic efficiency requires that the incremental impact of a project or policy is assessed in terms of the benefits to society and the costs of implementation, accounting for when these costs and benefits are borne. The application of such cost-benefit analysis (CBA) is mandatory in some countries (e.g.US Presidential Executive Order 12291), the rationale being that applying this economic efficiency test allows scarce resources (financial or otherwise) to be allocated to projects or policies that have the greatest benefit (net of costs) for society.

CBA is a *decision-support* framework. While CBA is an important part of any decision-support framework it is not the only part. For most public decisions it provides a key bit of information to policy makers, who also take account of other factors, notably the distribution of the costs and benefits. The same applies to decisions involving biodiversity conservation, where both the distribution of benefits and costs as well as an estimate of the net benefits is important, particularly so if some countries benefit from a particular conservation policy whilst others incur net costs. The current global recession and associated budgetary constraints imply the need to target policies that deliver the greatest *net* social benefit, i.e. outcomes where nature conservation goes hand-in-hand with an increase in societal welfare. This is where CBA fits in.

In this report we first present evidence of the *extent* of net benefits at the micro level, i.e. if we value ecosystem services and biodiversity at the local level for projects and policies, do we find that conservation tends to be better than an alternative option? Under which circumstance is this likely to be the case[1]? Second, what approaches are available to include changes in ecosystem service provision at a larger scale? Third, having reviewed this literature, we carry out primary research to estimate the benefits and costs of a range of global-scale scenarios: do these outcomes have positive net benefits relative to a given baseline or counterfactual scenario, and if so how clear-cut are they given the uncertainties which pervade the analysis?

In essence we try to answer the following question at different spatial scales: Instead of asking whether we can afford conservation, is it in fact more sensible to ask whether we can afford *not* to conserve ecosystems and biodiversity? If the costs of conservation (including opportunity costs, i.e. alternative productive uses for a patch of land) are outweighed by the benefits (often expressed in terms of avoided losses in the provision of ecosystem services) then there is a strong mandate for policy intervention.

1.2 The 'pricing' of nature

CBA relies on costs and benefits being expressed in monetary terms. This has significant ramifications for the analysis of projects or policies that affect nature in that the benefits provided by nature often do not have a direct market price and have therefore to be valued by some indirect methods. Non-market valuation methods are well-developed (see TEEB Ecological and Economic Foundation study Chapter 5 for a discussion of valuation methods). There are at least four issues pertaining to the 'pricing' of nature as an input to CBA that impact on this study:

(1) *Valuation methods are not routinely applied.* One of the aims of TEEB is to mainstream the use of tools that value nature and decision-making frameworks that use such valuation estimates. Although an environmental CBA should include an evaluation of the impacts of any project or policy on nature, such impacts are often omitted because of a failure to appreciate that nature is valuable and/or a lack of technical expertise. This implies that the evidence-base for CBAs in the field of nature conservation is relatively limited (particularly in terms of scope) as compared with the number of projects and policies that would gain from being assessed using CBA.

(2) *Our suite of non-market valuation methodologies often does not capture the full range of benefits that ecosystems provide*. TEEB has developed a typology for the range of ecosystem services that is set out in the TEEB Ecological and Economic Foundation study, Chapter 2. However, in the database of valuation studies used in this report (see the discussion in Section 7) we find that, for certain ecosystem service categories (such as 'gene-pool') there are very few (if any) estimates in the valuation literature.

[1] The comparison depends on what alternatives are considered and this is an important dimension of any CBA.

(3) *Our valuation estimates rely on the description of changes in bio-physical conditions*. This implies the need to understand the change in bio-physical conditions that would arise were we to apply/not apply the project or policy: there is evidently a corresponding level of uncertainty. It also implies the need (in stated preference valuation methods) to be able to communicate this change to a member of the public in order to elicit his or her valuation of the change, which adds a further layer of uncertainty in terms of the reliability of the benefit estimate.

(4) *There is a lack of standardisation in how valuation methods are applied*. Although some progress has been made in this regard, notably following the Report of the NOAA Panel on Contingent Valuation (Arrow *et al.*, 1993), it remains the case that the methodological rigour applied in non-market valuation studies is variable and methods are under continual refinement.

The ‘pricing’ of nature is characterised by these limitations. It is important to recognise that the analysis presented in this study is based on this limited information base. Our analysis of costs and benefits in this report is presented at different spatial scales (local, regional and global); these four issues apply at all scales, but affect the reliability of our analyses in different ways. We discuss these issues of scale in the next section.

1.3 Geographical scale of analysis

The most robust approach to the estimation of benefits is to carry out a primary valuation at the site in question, allowing for the full gamut of ecosystem services affected, and based on a defensible bio-physical analysis that specifies the incremental change being valued. The cost assessment should equally be robust and defensible, and based on incremental (additional) costs. We detail what constitutes an ideal CBA at project (site) level in Section 2 but we note at this stage that many of the aforementioned issues apply: there are relatively few CBA studies pertaining to nature conservation at project level that are methodologically sound, few value all ecosystem services, and there is likely to be systematic selection bias.

These local-level studies form a valuable contribution to the cost-benefit evidence base for conservation policy but this report considers them as only one element in the evidence base. Section 3 provides a synopsis of ecosystem service-analysis tools that have been applied at a regional scale.

Some policy measures have impacts upon ecosystems (terrestrial, coastal and marine) that go beyond the regional and are truly global in nature. We carry out a CBA for a selection of such policies in Part III. These policies include trade liberalisation, the extension of protected areas designation, changes in dietary patterns etc.[2]. For these policies, the systemic links between the economy and ecosystems requires careful examination. In political terms, the application of such global policies is often more problematic than local or regional interventions (*c.f*. the Copenhagen negotiations on climate change) as there is a need for international

[2] These scenarios and policies are not the only ones of interest and there could be others that yield greater net benefits. They have been selected as they have been mentioned in previous assessments, most recently Global Biodiversity Outlook 3 (http://gbo3.cbd.int/).

cooperation and coordination. But the potential gains from achieving an agreement are massive; just how significant these benefits might be is difficult to estimate. A part of this research, set out in Part III, is the development and application of a methodology that links bio-physical modelling to ecosystem valuation so as to derive an indicative estimate of the benefits of such measures. This is the 'new' element of this study in that sections 2 and 3 are critical systematic reviews of extant literature. We juxtapose this new benefit analysis for these global policies with cost estimates (where available) from the literature.

The error range in the analysis is larger for such a global assessment as compared with a local (project-level) assessment. Global assessments by their nature must apply generalised decision-rules (e.g. if scenario *x* applies then ecosystem impact *y* occurs) that may not be applicable at local level. Further, on both the benefit and cost side of the CBA there is a need to upscale results and apply 'benefits transfer' – the use of estimates of benefits obtained from studies in one location and at a given point in time for another location and another time period. This is discussed in the section below.

The need to have this discussion with regards the global assessment applies as the methodological issues that arise from *any* analysis that uses data from site-specific primary valuation to estimate macro-scale bio-physical changes are controversial and some authors (e.g. Bockstael *et al.*, 2000) argue against such studies, *viz*. no number is better than the wrong number.

We nevertheless believe that it is valid to present just such an analysis which represents, in our opinion, lower-bound benefit estimates (for the reasons set out in the sections that follow). There is a strong appetite amongst policy-makers to extend the evidence-base on such macro policy interventions and we feel that the methodology applied is defensible. But it is incumbent on us to set out the arguments against such a global assessment and to provide responses. We attempt to do so below.

1.4 Benefits Transfer and upscaling value estimates

Benefits transfer estimates the value of a policy site (or the provision of individual ecosystem services at that site) by using valuation estimates derived from one or more study sites, therein avoiding the costs incurred in conducting a primary valuation; benefits transfer is discussed in the TEEB Ecological and Economic Foundation study, Chapter 5. In any global assessment it is necessary to apply some form of benefits transfer as discrete primary valuation estimates cannot possibly exist for every affected patch of land in every affected ecosystem.

The valuation database developed for the benefits transfer in this study is perhaps the largest and most comprehensive of its kind. It has been populated not only with studies from several extant databases[3] but also through a process of expert-review coordinated by TEEB, wherein biome experts reviewed and sorted studies in the

[3] These databases included COPI (ten Brink *et al.* 2009), EVRI (1997), ENVAlue (2004), EcoValue (Wilson *et al.* 2004), Consvalmap (Conservation International 2006), CaseBase (FSD 2007) ValueBaseSwe (Sundberg and Söderqvist 2004), ESD-ARIES (UVM 2008) and FEEM (Ojea *et al.* 2009).

database and contributed further studies (see Section 7). The studies are used to develop biome-specific value functions as described below.

The value functions that we use in this study relate the value of an ecosystem patch (per hectare, per year) to: (i) its bio-physical characteristics (area); (ii) its scarcity (i.e., area of the same ecosystem within vicinity of valued site); (iii) land use intensity (human appropriation of net primary production (HANPP) and fragmentation); and (iv) its socio-economic context (population, income, accessibility etc.).

We then use these value functions to estimate values for all patches of all ecosystems, controlling for their site specific characteristics. The bio-physical modelling carried out for this study reveals a distribution of landscape-types that is affected by the policy scenario being evaluated (e.g. extending protected areas). This distribution is different to the baseline or no new policy scenario (i.e. business-as-usual). It is this change in distribution of landscape-types that we estimate a value for. These site-specific values are used to value changes in the extent of different landscape-types and subsequently aggregated for reporting at a regional and global level. This provides us with one measure of the benefit of the policy being applied.

Although we consider a composite 'package' of policies, following the bio-physical modelling of PBL (2010), our focus is very much on the potential to apply one policy at a time. This is important as, following Hoehn and Randall (1989), multiple projects undertaken together would not realise the same net benefits as a sequence of such projects undertaken independently. This is because undertaking multiple projects changes prices and changes substitute sets, e.g. the availability of other conserved habitats. We stress the need to determine whether the changes in the distribution of landscape-types can be considered 'marginal' for this reason.

It should be noted that we are not attempting to provide a global estimate of the value of a biome (or indeed the global value of all biomes) as per Costanza *et al.* (1997). Some of the policy scenarios do involve non-marginal changes in some regions but none approach the total loss of a biome in global terms. We set out the extent of land-use change for each change scenario in the results section and comment on the extent to which changes might be treated as marginal (Section 12).

We provide this synopsis of the methodology applied for the global assessment at this point so as to contextualise the discussion on benefits transfer and scaling up values; further methodological detail is provided in Part III. There are a series of issues pertaining to scaling up which we feel should be addressed as they question the entire validity of any form of global assessment that uses value estimates from individual site studies, and therein a significant element of this report. Some of these issues are set out in Bockstael *et al.* (2000). We present them below along with how the methodology applied in this study has attempted to address the issue:

(1) *Valuation estimates are diverse and values cannot be compared or standardised to common units: physical units (e.g. per hectare), temporal period (e.g. annual value); or currency (e.g. US$).*

Taking into account this issue, the majority of studies in the 'long list' of studies in the database are not used in the estimation of benefit functions as values cannot be standardised. For instance, the ‘lakes and rivers’ biome uses 388 value estimates in the meta-regression from a list of 1,896 value estimates in the long list. We have been highly selective in the choice of studies used and have chosen those that best permit standardisation. However there is no reason to believe that the studies not included systematically biases the resulting benefit functions.

(2) *Primary valuation estimates are site-specific and so cannot be transferred*.

This is true if transferred values are not adjusted for site characteristics but we use bio-physical, scarcity and socio-economic explanatory variables which ‘explain’ a proportion of the site-specific variability in value estimates. Further, we transfer values at the individual patch/site level – not to the entire stock of an ecosystem within a region – and this approach is more precise. The application of a geographical information system (GIS) model at global scale for the benefit transfer has no precedent to our knowledge.

To give some indication of the level of disaggregation, consider the number of patches that are individually assessed in the benefits transfer in our study: grasslands 1,494,581; tropical forest 292,822; temperate forest 672,942; wetlands 191,539; mangroves 6,850; coral reefs 16,149; and lakes and rivers 375,316. The bio-physical modelling for the change scenarios is restricted to the first three biomes listed, although the other biomes are assessed separately in Section 10. Summing these three biomes alone gives 2,460,345 individual patches that are each *individually valued* using the biome-level value functions. The extent of changes in patch size arising from a scenario option is determined by the GLOBIO bio-physical model (see Section 5) and this allows the estimation of a value change on a patch-by-patch level that is then aggregated.

This level of disaggregation is necessary if the *localised* variables that the environmental economics literature tells us might influence the value of a particular site are to be estimated without primary valuation, i.e. by using benefits transfer. Without this high-resolution GIS analysis we could not for instance link a proxy for intensity of land-use (in the form of ‘human appropriation of net primary product’ (HANPP) – see Section 8) or habitat fragmentation (roads within a 10km, 20km and 50km radius of the site) to the values generated in primary valuation studies. In turn, without this disaggregation and the use of GIS we could not estimate value changes across 2,460,345 sites based in part on the local HANPP and habitat fragmentation. In short, we believe the methodology applied in this study to be as well-developed as comprehensive as is feasible given the period of study. Notwithstanding this, we do accept (following Colombo and Hanley, 2008) that transfer errors (and thus predictions of the benefits of an option scenario) depend on the sub-set of sites and studies used to construct the benefits transfer model.

(3) *Values for individual sites cannot be added together to assess large scale changes in the extent of ecosystems.*

This is the scaling up problem: large losses or large gains in the stock of an ecosystem within a region will impact the value of the remaining stock (i.e. non-constant marginal values). The information that we have from primary valuation studies, however, is largely for marginal changes at individual sites holding the rest of the stock of that ecosystem type constant.

Our response to this concern is two-fold. First, the majority of changes in land-use are marginal and we set out in Section 12 the extent of land-use change for each change scenario. Second, we investigate the effects of variables measuring the abundance of ecosystem stock in the estimated value functions. Where significant effects exist values are adjusted to reflect changes in the scarcity or abundance of substitute sites at localised (patch) level.

(4) *Key ecological functions underlying specific ecosystem services vary spatially and temporally across habitats, implying that an average value of these services expressed on a per hectare basis is misleading*.

Our methodology cannot allow for variations within a patch. Barbier *et al.* (2008) show that different hectares of mangrove have different values within the same patch (seaward hectares are more valuable for wave attenuation). Since we are using average value estimates for the entire patch, this intra-patch variability is not captured. However, our estimate of the benefits from a policy option will be over-stated if (and only if) the selection of study sites in the valuation database is skewed towards those sites that have an atypically high level of ecosystem service provisioning, and vice versa. For instance, if the mangrove sites in our valuation database have a disproportionally high proportion of seaward patches (as compared with the global average proportion of seaward patches across all mangrove sites) then the value for wave attenuation per hectare of mangrove will be over-stated. If this condition does not apply then we would argue that transfer errors are likely to be self-cancelling, i.e. we over-estimate in some cases but under-estimate in others. It is difficult to say whether such a bias applies.

Although the methodology that we apply for the global assessment to some extent mitigates these problems we accept that issues remain. As such, the analysis is intended to be indicative of the benefits of the various policy options. We discuss this further in Part III.

1.5 Links with the TEEB Interim study

The current study is a development of the TEEB Interim Study (Braat and ten Brink, 2008) which was commissioned to provide evidence with respect to the costs of *not* meeting the 2010 biodiversity target, i.e. a significant reduction of the current rate of biodiversity loss at the global, regional and national level as a contribution to poverty alleviation and to the benefit of all life on Earth[4]. The report looks at the Costs of Policy Inaction (COPI) is defined in Braat and ten Brink (2008):

> *'The cost of policy inaction is defined as: the environmental damage occurring in the absence of additional policy or policy revision. Inaction not only refers to the absence of policies, but also to the failure to correct misguided policies in other areas. The costs of policy inaction may be greater than just the environmental*

[4] http://www.cbd.int/2010-target/

damage, if the same inaction also creates societal and economic problems' (Braat and ten Brink, 2008, p.2).

The authors proceed to set out the purpose of the Interim Study, i.e. 'to highlight the need for action, prior to the development of specific policy instruments' (*ibid*.). This current study (the Quantitative Assessment) is the next stage of the assessment in that it is explicitly concerned with global *change* scenarios, i.e. not just business-as-usual, some of which can be linked to policy instruments.

There are certain similarities but also clear methodological difference between the TEEB Interim Study (Braat and ten Brink, 2008) and the current study. Both studies use the IMAGE-GLOBIO bio-physical model (Alkemade *et al.*, 2009)[5] that produces projections of the extent of landscape-types and a measure of ecosystem intactness, the unit-of-account being Mean Species Abundance; the model design and limitations are discussed further in Part III.

In the Interim Study Braat and ten Brink (2008) focus exclusively on the COPI estimation, i.e. the valuation of predicted changes in the provisioning of ecosystem services in 2050 (under a business-as-usual scenario) as compared to estimates of provisioning of ecosystem services in the base year (2000). Unlike the current study, there is no assessment of policy interventions to mitigate the losses.

The approach adopted by Braat and ten Brink (2008) also differs from the present study in that the Interim Study carried out benefits transfer across individual ecosystem service categories. Owing to the paucity of valuation data points, benefits transfer in the Interim Study was carried out across biomes for some ecosystem services and gap-filling was applied to link the state of habitat degradation (presented by land-use intensity, which is a major pressure for biodiversity as measured by Mean Species Abundance by IMAGE-GLOBIO) with estimates of provisioning across the range of ecosystem services. The approach is valid and defensible but requires expert-judgement vis-à-vis gap-filling.

In the Interim Study Braat and ten Brink (2008) provide a value estimate of €14 trillion for the loss of ecosystem services associated with the loss of biodiversity over the period 2000 to 2050, equivalent to around a 7 per cent of gross domestic product (GDP) in 2050. The loss of forest services accounted for a big share of this, with just over 5 per cent of GDP loss in 2050 associated with ecosystem services related to the forest biomes.

We do not present a COPI-type estimate in this study. It is the case that there are three outcomes from the IMAGE-GLOBIO bio-physical modelling that together are the analytical basis for the benefit estimates in the present study for each change scenario: (i) the 2000 base year; (ii) the 2050 (or 2030)[6] baseline scenario; and (iii) the policy scenario. The focus of the global assessment in this study is analysis of (ii) and (iii), whereas a COPI-type estimate could be produced by comparing (i) and (ii) alone. It is noteworthy that the 2050 baselines from IMAGE-GLOBIO bio-physical

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[5] http://www.globio.info/

[6] Some option scenarios are modelled by PBL (2010) to 2030 whereas others are modelled to 2050; this is discussed further in Part III.

modelling used in this study are not significantly different to those used in Braat and ten Brink (2008), i.e. if their methodology were to be re-applied broadly similar overall COPI results would likely arise.

There is a trade-off between the approach applied in this study versus Braat and ten Brink (2008). The approach in this study differs from that applied in the Interim Study in two ways:

(1) The key indicator of ecosystem ‘intactness’ in the IMAGE-GLOBIO modelling is mean species abundance (MSA). Braat and ten Brink (2008) use MSA and determine (through meta-analysis of published data) a series of generalised relationships between ecosystem services, land-use types and biodiversity[7] and then apply scaling coefficients. But there is limited evidence to support the relationships used across all ecosystem services (see for instance Naidoo *et al.*, 2008), particularly as MSA is itself only a partial proxy for biodiversity as we discuss in Part III (Section 5).[8] Our approach does not use MSA in the manner used in the Interim Report.

(2) The Interim study applied benefits transfer without using highly-disaggregated GIS. The use in this study of meta-regression analysis with benefits functions that account for the bio-physical, spatial and socio-economic characteristics, using GIS, is arguably more robust and defensible than the benefits transfer methodology used in the Interim Study.

Although the values in the global assessment segment of this report should only be taken as indicative, we believe that, for the reasons presented above, the estimates should be considered as lower-bound values.

1.6 Structure of the report

The bias in the discussion in this Introduction towards the global assessment in Part III should *not* be taken to be indicative of the importance that we place on this segment of the report vis-à-vis the local-level and regional-level analyses in Parts I and II (i.e. sections 2 and 3) respectively. It is just that the global assessment is more controversial and the caveats applied to the methodology and the interpretation of results should be stated up-front.

Section 2 considers evidence from micro-level studies on the costs and benefits of conservation options, both policies and individual projects. Section 3 considers analyses at a more regional scale, focusing primarily on the benefit-side. The approach adopted in our global assessment (Part III) is one of a range of options – Section 3 sets out alternative methodologies and the results generated. Part III sets out the assumptions, methodological stages, data sources, and overall results of the global assessment.

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[7] See Braat and ten Brink (2008), Figure 9, page 8.

[8] We do however use MSA in our analysis by testing to see whether the component elements that determine MSA are significant as explanatory variables in the meta-regression analysis, i.e. does MSA explain our valuation estimates for individual sites? This is discussed further in Section 8.

PART I LOCAL ASSESSMENT

2 Micro-level analysis

2.1 Introduction

The content of Section 2 is a synthesis of a project report carried out by eftec (Tinch *et al.*, 2010) as an input to this study. Tinch *et al.* (2010) do not carry out original valuation studies, but rather aim to identify and assess literature estimating both the costs and the benefits of biodiversity conservation. The rationale is as follows: (1) there are relatively few studies that assess both the costs and benefits of the same micro-level project or policy; (2) a systematic review to determine this micro-level evidence base had not been conducted; and (3) providing this evidence base might allow general conclusions to be drawn vis-à-vis the benefit-cost ratios of conservation expenditures.

A screening process was carried out to assess the analytical quality of studies that provide cost-benefit estimates at micro level; a similar process is applied to determine which site-specific valuation studies (assessing benefits and benefits alone) are inputted to the benefits transfer database, as set out in Part III. For the CBA screening, the potential for omitting studies on the basis of methodological integrity is substantially higher than applies for the valuation database: there is the need to appraise not only the assessment criteria for the non-market valuation of benefits but also how costs were estimated and how these benefits and costs were compared, i.e. the integrity of the CBA overall. There is thus a need to ensure that quality criteria for the CBA assessment should not be so demanding as to rule out studies which make acceptable approximations, thus providing policy-relevant results within particular decision contexts.

The analysis in Section 2 provides coverage in the following categories:

1. *Range of threats*: These include climate change, sea level rise, desertification, invasive species, eutrophication, other diffuse and point pollution, habitat loss and site development, overexploitation, extractive industry use and abandonment/change of use. As long as costs and benefits of conservation are estimated, any of these threats are relevant to this project.
2. *Range of ecosystem types:* Terrestrial, freshwater and marine ecosystems in both their natural state and also semi-natural states.
3. *Range of ecosystem services*: All ecosystem services are relevant whether the studies capture (in qualitative, quantitative or monetary terms) all or part of their services and whether costs and/or benefits can be disaggregated to the service level.
4. *Geographical scope*: across different regions at different stages of economic development.
5. *Types of cost categories*: Cost profiles over time, i.e. capital and operating expenditures; types of capital, i.e. land, labour, and environmental costs; and agents incurring the costs (e.g. central or local government, local landowners *etc*.)

The remainder of section 2 sets out an abridged version of the Tinch *et al.*, (2010): methods used for classifying and assessing the literature (Section 2.2), including the evaluation criteria (Section 2.3), the case studies developed (Section 2.4), and the conclusions for cost-benefit analysis of biodiversity and conservation decisions (Section 2.5).

2.2 Methods for classifying and assessing the literature

A wide range of published and 'grey' sources of information were reviewed *via* desk top analysis and consultation with researchers in the field. Sources include:

1. academic and industry journals;
2. previous research carried out for the European Commission;
3. the EVRI database (EVRI, 1997);
4. conference and seminar papers including (but not limited to) those presented at the UKNEE's annual applied environmental economics conference since 2003, EAERE[9] (and World Congresses), BIOECON[10] and others; and
5. 'grey' literature including papers forthcoming in journals, working papers from relevant research institutions; see Nunes *et al.* (2009) [11] for a discussion of the importance of this.

Literature tends to focus on a single service, resource, benefit or cost category encouraged by the policy or research interest for which the studies were commissioned (eftec, 2009). However, though there are rather few studies explicitly setting out to compare conservation with exploitation, many studies in effect do this through the specification of the baseline against which values are assessed. Further, a micro-level CBA can in some cases be 'constructed' by combining sources or benefits/costs transfer. A two-level approach was adopted for the classification of the literature:

- First level: long-list of possible studies, noting the coverage of costs and benefits and allowing assessment of relevance with respect to the scope of the project. This level allows us to determine which studies are likely candidates for further development as case studies.
- Second level assessment: a more detailed analysis of the individual cases covering the context, drivers, methodologies, assumptions, results, sensitivities and key policy and science conclusions.

The long list assesses the extent to which the source (e.g. an article, chapter in a book, study report etc.) covers the costs and the benefits of conservation, the extent to which the source can be useful as a case study in this research, and a brief description of the reasoning behind that decision. The assessment criteria used for this level are outlined in Tinch *et al*. (2010).

The long-list database contains 225 entries: 80 are considered to have some potential for case study development; 36 could support case studies if either benefits

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[9] http://www.eaere.org/
[10] http://www.ucl.ac.uk/bioecon/05respap.html
[11] Nunes P, Idega E O, Loureiro M L (2009) *Mapping of Forest Biodiversity Values: A Plural Perspective*, working paper, FEEM. http://www.bepress.com/feem/paper264/

or costs could be transferred or a proxy found; 6 provide possible cost-effectiveness cases studies; 30 do not contain case studies but are of interest for related scientific or methodological issues; and 12 are collections of references/literature reviews containing leads to other possible case studies.

It is perhaps noteworthy that the overlap between studies in the long list and the studies used in Part III of this report is limited, i.e. around 10%; many of those in the long list use benefits transfer to carry out cost-benefit analysis, whereas the valuation database contains only primary valuation studies that often do not consider any costs. The criteria and their application are discussed in Tinch *et al*. (2010).

Table 1 Simple assessment criteria for long-list database

Cost data; Benefit Data (separate)	Overall Assessment
Yes (monetary) Some monetary Quantitative (non-monetary) Qualitative None	YES: most Costs and Benefits YES: some Costs and Benefits YES: for Cost Effectiveness Analysis SUPPORTS another source COLLECTION of case studies MAYBE: if transfer Benefits MAYBE: if transfer Costs NO but relevant to quality/science NO

For each of the 40 studies in the short-list, a detailed assessment is carried out vis-à-vis content, quality of the methods and data, and the potential contribution to allowing generalised conclusions to be drawn. No attempt is made to 'correct' for any perceived errors in the study. The selection of the case studies for the short-list from the long-list was based on selection criteria, a synopsis of which is as follows:

1. Is the *coverage* of the policy context, the good and the change of sufficient quality?
2. Is the *affected population* defined correctly and, where relevant, sampled sufficiently?
3. Are the *results* valid and robust?
4. Has the *economic valuation methodology* been selected and applied correctly?
5. Has the CBA been carried out following *best practice*?
6. How does the study contribute to the *overall evidence base*, given other studies selected? Is there a need for coverage (and thus inclusion) vis-à-vis ecosystem-types, ecosystem services, types of economic value, geographical scope and decision-making contexts?

It should be recognised that it is usually not feasible for any given study to pass all the criteria and that criteria vary in terms of how critical they are in determining the overall reliability of results. For inclusion in the short list, the criteria must be met to the extent that the policy imperative (of providing an evidence-base) is met whilst retaining a *sufficiently high* level of reliability. If only 'perfect' studies were to be included then the short-list would be almost empty, therein not satisfying the requirement to provide local-level evidence of cost-benefit ratios.

Hence, these criteria are not interpreted as strict exclusion criteria, but rather as a set of desirable features that can guide selection and interpretation of case studies. It should also be recognised that it is not always possible to check exactly how a given case study performs with regards a given criterion: reporting is not complete for all cases and any benefits transfer applications are not re-tested using the pertinent original valuation papers or meta-analyses. The objective is not to critique specific sources, but rather to assess the extent to which it is possible to construct a robust case study on the basis of one or several sources. For example, if a study uses an out-of-date or inappropriate monetary value for carbon sequestration, but reports physical emissions, it is straightforward to update it with most recent figures. Similarly, it may not matter whether one source has weak or missing cost estimates if it is possible to gap-fill appropriately through other source data.

Tinch *et al.* (2010, Annex 2) set out the selection criteria in more detail. We do not present this analysis here. Instead we focus on the case studies chosen in the short-list and a general discussion of outcomes[12].

2.3 Micro-level cost-benefit case studies: Overview

There are different ways of grouping the micro-level CBA evidence-base. The pertinent questions with regard to grouping is how will the analysis be used, and where do the commonalities apply (and therefore where might generalised outcomes be appropriate)? The evidence-base is likely to be used by a variety of stakeholders including policy-makers at different levels, Non-Governmental Organisations and local communities. Commonalities might exist across studies pertaining to the same or complimentary biomes, those evaluating a similar ecological threat faced, those with a similar policy-context (e.g. different forms of biodiversity action plans), or those assessing the same sector (e.g. mining). Tinch *et al.* (2010) decide – following the outcomes of a workshop – on a loose combination of policy type and biome:

1. Protected areas - terrestrial and marine habitats;
2. Land use/conversion;
3. Habitat destruction
4. Restoration;
5. Water supply
6. Flood protection;
7. Species conservation;
8. Pollution control;
9. Agricultural systems; and
10. Urban nature.

Each of these policy/biome combinations is treated in turn below in sections 2.3.1-2.3.10. Note that we focus on the studies that appear in the short-list in Tinch *et al.* (2010). Appendix 1 contains further studies pertaining to several of these policy/biome combinations that are reviewed and discussed in Tinch *et al.* (2010) and

[12] In the sections below, we refer to various environmental valuation methodologies (e.g. choice experiments, travel cost method, contingent valuation method) and sub-categories of economic benefits (e.g. use and non-use values). Definitions and discussions can be found in TEEB Ecological and Economic Foundation study.

are relevant to the policy discussion. Appendix 1 also contains summary boxes for each of the 40 studies in the short-list, providing further details in a structured format.

2.3.1 Protected areas

There is a substantial literature on aspects of the costs and benefits of protected areas, although relatively little looking at both costs and benefits together, or applying a full cost-benefit approach. Work on cost-effectiveness is more common: either the greatest protection available for a given price, or the cheapest method of achieving a given standard of protection. Optimisation of location is a particular focus (Wainger *et al* (2010): 'the question of how to choose among land protection options has received the most attention').

For example in the marine environment there is an extensive literature on marine protected areas. There are cost estimates (e.g. Balmford *et al* 2004, The worldwide costs of marine protected areas; Sumaila *et al* 2007, Potential costs and benefits of marine reserves in the high seas) and studies of marine reserve benefits (e.g. Russ *et al*. 2004, Marine reserve benefits local fisheries; Gell and Roberts 2003, Benefits beyond boundaries: the fishery effects of marine reserves; Halpern 2003, The impact of marine reserves: do reserves work and does reserve size matter?). But there is little that combines monetary estimates of costs and benefits.

Some studies that adopt a cost-benefit framework nevertheless do not qualify as cost-benefit studies due to a lack of data preventing valuation of key parts of the appraisal. Although our evaluation criteria do not require that *all* costs and benefits be expressed in monetary terms, we do require at least *some* costs and benefits to be so expressed. For example Sumaila *et al* (2007) present an interesting study in a cost-benefit framework, 'Potential costs and benefits of marine reserves in the high seas', but cannot be considered a cost-benefit analysis since the only monetary estimate is of the opportunity cost of lost fish production in the short term. Longer term benefits, including fishery gains and reduced risks, are discussed but not quantified. The paper nonetheless presents a strong argument for some increase in protection of the high seas: the estimated opportunity costs are only US$270 million annual profit loss from a 20% closure of all pelagic and deep sea fisheries, and it is noted that about US$152 million per annum is currently paid as subsidies to high seas deep-sea bottom trawlers alone.

Owing to the lack of data on the benefits side, this example is not written up as a case study; but it does stand as an example of the usefulness of the cost-benefit framework – or we might call this 'cost-benefit thinking' – as a rational and methodical approach to structuring and presenting information for constructing arguments and decision support, even if it is not possible to put monetary figures on all or most of the impacts.

Marine protected areas

Case Study 1: UK Marine Conservation Zones

One area where there is enough evidence to support a cost-benefit case study is the Marine Conservation Zone (MCZ) provisions in the UK Marine and Coastal Access

Bill. The case study written up here (Case 1: 'UK Marine Conservation Zones') draws primarily on Defra (2009), the Marine and Coastal Access Bill Impact Assessment, and also on several supporting documents (McVittie and Moran 2008, Moran *et al* 2008; Hussain *et al* 2010; ABPMer 2007).

The analysis is applied at national scale – there is in fact little evidence at the individual site level. The study identifies 11 ecosystem service impacts and attempts to value seven of these. Food and raw materials based on market values; recreation on expenditure; nutrient cycling on the Costanza *et al*. (1997) per ha figure; climate regulation on primary productivity and UK carbon value; sea defence on avoided cost; cognitive values on value added (research spending) and expenditure (education) with specific marine focus. Additional carbon savings (not related to the ecosystem services) are also considered. There would appear to be some risk of double counting through the inclusion of nutrient cycling as a separate category, since this underpins food production and other services, however the figure for nutrient cycling derives from an earlier study (Costanza *et al*. 1997) which in turn derives the value from literature review; Costanza *et al*. also include food production values, and state that they have attempted to avoid double counting, but we can not resolve this issue without additional work uncovering exactly how the Costanza *et al*. (1997) figures were derived.

A separate stated preference (SP) survey is carried out for non-use values, but these are not treated as additional in order to avoid possible double counting. This is a frequent issue in the cost-benefit studies identified for this research: there is often a concern that the SP surveys used to assess non-use values may be detecting some part of use values too, and that including both the SP results and the ecosystem service values derived by other means could result in some double counting. Some studies present arguments regarding which other services are thought to be covered and which not – for example, work on Case 29: 'Blackwater Estuary2 considers a 'composite environmental benefit' intended to cover a wide range of impacts (recreation, aesthetics, water quality, biodiversity) but includes separate values for fisheries benefits and climate regulation, which are thought not to be considered by respondents in formulating their SP responses. Other studies, such as the UK MCZ example, avoid adding the numbers together but hold the non-use values 'in reserve' as a further argument for the robustness of positive benefit-cost ratios (BCRs) – i.e. arguing the case for conservation based only on use values, while pointing out that additional values will exist.

The study suggests that active conservation of the UK marine habitat has a positive net present value, estimating that establishment of a network of MCZs throughout UK waters has a positive BCR of between 6.7 and 38.9. Although this is an imprecise conclusion based on far from perfect evidence about benefits, the results are reasonably robust in the sense that sensitivity testing shows that even given the uncertainty in the estimates it is rather unlikely that the BCR could be below 1.

The study is a good illustration of the use of expert judgement to score likely impacts where we have some evidence of the total value of a service, but limited evidence of the impact on that service of a specific policy change. This kind of uncertainty is quite pervasive in studies of conservation decisions, and there are different approaches to it. Some studies in effect push the scientific uncertainty into the valuation study, using stated preference studies of willingness-to-pay for conservation actions or results without actually modelling the ecological relationships. This is the approach in many older studies, for example, Case 3: 'Natura 2000 sites in Scotland' and Case 31: 'Wild goose conservation in Scotland'. More recently, there has been a greater focus on use of one or other ecosystem services framework, explicitly breaking impacts down to individual services and attempting to value them separately, as in the UK MCZ study. This puts greater emphasis on issues of missing data, and the use of expert judgement is one way of trying to deal with this. Intuitively it makes sense that we might expect more accuracy from letting experts make the judgements on scientific and ecological relationships, and limiting valuation tasks to clearly specified outcomes, but where stated preference is used this does depend on people being able to think of different impacts separately. If in fact there are strong linkages between impacts – for example, conservation of a particular species might not be possible without conservation of habitat and good environmental quality – then it may not be reasonable to expect respondents to overlook these linkages, and valuation of the species conservation is indeed likely to involve valuation of the conjoined changes. Where this is the case, even if the assessment framework breaks impacts down into all the component ecosystem services, it may still be preferable to use composite environmental values that are considered to cover several service categories.

Case 2: Lyme Bay no dredging zone, England

This is a very different example from the marine environment. The study considers a proposed conservation zone of 60 square nautical miles centred on Lyme Regis, UK. Within this area, scallop dredging would be stopped, but more sustainable forms of fishing would be allowed (e.g. dive catching of scallops, crustacean potting and fixed netting of skates and rays), as would recreational use. This case illustrates a partial cost-benefit approach, focusing only on the market returns from different options. Since the market returns from the protection option exceed those from the business-as-usual case, this provides good evidence that protection would be beneficial, given that the environmental benefits of protection are unknown but certainly positive.

This strategy is used at a number of levels in other case studies: sometimes at this 'extreme' level of focusing only on the market returns, and sometimes in 'milder' forms where certain more easily quantifiable ecosystem service impacts are included. In particular, this is becoming a common approach for projects with strong carbon implications (see e.g. Case 14: 'Hypothetical forest area, Cambodia') where the argument constructed is in essence that the economic impacts plus the potential market value of carbon changes are themselves enough to justify a project, and in addition there are other, non-monetised ecosystem service benefits that are unequivocally positive and therefore can only strengthen the result.

Terrestrial protected areas

Case 3: Natura 2000 sites in Scotland

There are also several examples of cost-benefit analysis applied to terrestrial protected areas. An analogue to the UK MCZ study is Case 3: 'Natura 2000 sites in Scotland' (Spurgeon *et al*. 2005), which seeks to assess the economic costs and benefits of the Scottish Natura 2000 sites. There are 300 separate sites, represented by seven case study areas containing 12 Natura sites. The study presents assessments for the individual case study areas and also an overall assessment for Scotland (the figures presented in the Annex here relate to this aggregate assessment). The study is based on contingent valuation (a form of stated preference) for benefits and expenditure survey/data for costs. Economic impacts (visitor expenditure and jobs supported) are also calculated but it is correctly recognised that these cannot be added to welfare impact estimates.

The contingent valuation approach is good in separating out general public non-use values, site visitor use values, and 'non-Scottish visitor to Scotland' non-use values, and in assessing distance decay of values according to distance from sites assessed.

On the other hand, the stated preference survey for public non-use asks for willingness-to-pay (WTP) in additional taxes each year for next 25 years. This is likely to lead to a 'recontracting' problem, i.e. it is not clear that the respondents really consider having to pay for each of the next 25 years. Generally a more conservative approach is to ask for a one-off payment. This is likely to mean that the WTP values considered are overestimates, but probably not so much as to cast doubt on the overall conclusion.

The study focuses on the marginal costs and benefits of designation, and protection. Therefore it does not seek to evaluate the total values of the sites in terms of their ecosystem service provision, but only the increment in value that is due to thee protection status. In many cases these are remote, little used sites with limited opportunity costs and little direct ecosystem service impact from protection. The use values considered by the study are limited to the incremental recreation value associated with protection, and so it is unsurprising to find that non-use values dominate the benefits. The benefit-cost ratios are strongly positive (about 7:1 for protection overall, and 12:1 for the incremental value of the Natura 2000 designation), and there are additional values not assessed (social, cultural, educational, research, environmental services and health values: all likely to be positive, though possibly partly included in the non-use responses).

Although there are many approximations and assumptions, the broad result that non-use values from local and international populations could justify Natura 2000 costs and opportunity costs seems reasonably robust. However this falls some way short of a detailed assessment of specific costs and ecosystem service impacts for particular sites. This approach therefore seems to be useful and appropriate at a national, political scale (supporting the argument 'Natura 2000 is money well spent') but not for more detailed decisions relating to specific sites, management decisions or compensation payments, where more detailed assessment considering the site-specific features of specific options and impacts on a full range of ecosystem services would be required.

Case 4: Cardamom Mountains Wildlife Sanctuaries Project, Cambodia

This case (Grieg-Gran *et al*. 2008) focuses on the issue of sustainable financing for two protected areas (wildlife sanctuaries) in Cambodia. The analysis is based on market-based estimates of the values achievable from immediate logging compared with ongoing protection with sustainable forestry, agriculture, non-timber forest products and carbon storage values. The results are dominated by two high values: the value of immediate timber extraction, on the one hand, versus the value of carbon storage, on the other. The central estimates show that (over 25 years at a 10% discount rate) the value of the protection scenario exceeds that of the non-protection scenario by a small margin. This conclusion depends on a rather high value assumed for carbon storage: the carbon value from midpoint of IPCC Working Group III: US$73-US$183 per tonne of carbon to achieve 'safe' levels. Actual carbon market values, and prices achievable for REDD+ projects, are not at this level. On the other hand there are important omitted values, notably global biodiversity conservation (non-use) values (which could be high for wildlife sanctuaries in tropical forests), and the costs/risks associated with deforestation's effects on erosion, flooding, and water quality/supply. The conclusion is that the protection of these areas may be globally optimal, but is locally costly: some financing mechanism will be essential to ensuring ongoing conservation. But this case is an example of a CBA with quite a targeted aim: not so much to work out whether or not the areas should be protected, as to work out how much financing/compensation will be required in order for local communities to support the protected status.

One important issue that is not addressed in the cost-benefit analysis is that of irreversibility. The decision to clear-cut the forest would be more-or-less irreversible, certainly within management-relevant time-frames, whereas the decision to protect is reversible in the sense that it would always be possible to log the area later on. This asymmetry is reflected in economic value frameworks by the concept of quasi-option value and this could be significant in many cases where essentially irreversible land-use decisions are at stake.

Case 5: Lowland forest protection, Uganda

Naidoo and Adamowicz (2005) examine the costs and benefits of avian biodiversity at a rainforest reserve through a combination of economic surveys of tourists, spatial land-use analyses, and species–area relationships. The results show that revising entrance fees and redistributing ecotourism revenues would protect 114 of 143 forest bird species (80%) under current market conditions. This total would increase to 131 species (90%) if entrance fees were optimized to capture the tourist's willingness-to-pay for forest visits and the chance of seeing increased numbers of bird species. In contrast, the cost of purchasing agricultural land for ecological rehabilitation of the avian habitat would be economically prohibitive. This is an interesting approach, grounded in explicit modelling of land values and conservation potential. However it relies on a very limited definition of benefits. The context is pressure on the forest from harvesting timber, making charcoal, collecting fuelwood, and encroaching agricultural development, and it is clear that actions that can effectively preserve forest birds will also preserve forests and therefore other ecosystem services that go along with that – notably carbon values, but also watershed and local benefits. In fact this study is not a full CBA of the options, but a partial analysis focusing on the extent to which tapping in to tourist willingness-to-pay to see bird species could provide the funds and local incentives to overcome the opportunity costs of conservation. Adapting this approach by combining it with a fuller analysis of ecosystem service changes could produce additional interesting and policy relevant results: in particular, carbon could be explored as an additional source of finance for the conservation activities.

Cases 6 and 7: Rajiv Gandhi National Park, and Dandeli Wildlife Sanctuary, India

These case studies are applications to protected areas in India, with a focus on the local costs and benefits. Global carbon values and biodiversity benefits are not included. This is not an omission as such, but rather a deliberate limiting of the scope of the CBA to a specific context, the impact on local people. Case 9: 'Old growth forests, Finland II' (see below) presents a similar approach, comparing the CBA results at local and regional levels.

The Rajiv Gandhi case (Ninan *et al*., 2007a) provides an interesting example of how the total Net Present Value (NPV) may look positive or negative depending on the boundaries/parties included, even at very local scale. Tribal communities receive a large positive total benefit from the Park, however this is paid for by neighbouring coffee growers, turning the NPV negative overall. But extending this study to cover the national and global benefits from conservation in this zone, including non-use benefits for iconic biodiversity, would very likely show strong net benefits.

The Dandeli case (Ninan *at al*., 2007b) similarly gives insight into the potential costs and benefits of biodiversity conservation to villagers living in and around a wildlife sanctuary, including their own valuations for biodiversity conservation. The study shows that the agricultural opportunity cost is almost twice as large as the benefits the villagers receive from NTFPs from the sanctuary. The villagers' values for biodiversity conservation balance this out somewhat, however the data available from this study show that overall NPVs are negative from a local perspective. Hence there would be support for the need to compensate villagers for the conservation. The study does not cover regional, national or global tourism and non-use values for conservation in this area; it seems likely that the inclusion of such values would show a positive NPV overall, and that it may be possible to use such a study to construct a case for national or international support to conservation in the area and compensation for local losses.

Cases 8, 9 and 10: Old Growth Forests, Finland

Three case studies have been developed focusing on conservation of old-growth boreal forests in Finland. Case 8: 'Old Growth Forests, Finland I' (Juutinen 2008) presents an interesting use of models and values to show optimal conservation policy. Based on data from 32 stands in northern Finland, the optimal forest rotation period is modelled, taking into account the value of protecting old-growth biodiversity. The results suggest that it is better to produce timber and biodiversity on the same land, but delay harvest significantly, than to specialise (some land for biodiversity, other for timber). The link between the economic value of the biodiversity and the specific biodiversity outcomes is rather weak, because the study uses value transfer from meta-analysis of stated preference studies for biodiversity value of old-growth boreal forests in Finland, and this general conservation value does not link explicitly to the measurement and modelling of the number of species supported based on a model of dead/decaying wood for stands of different ages, and the assessment does not cover spatial interactions across forest stands. The author recognises that the 'big problem is how the valuation of biodiversity benefits can be linked to the chosen biodiversity measure at the landscape level'. Nevertheless the model is successful in showing how plausible ranges would affect outcomes, gives a robust general conclusion (it will often be optimal to delay harvest) and prioritises which areas should be delayed most.

Case 9: 'Old growth forests, Finland II' (Kniivilä *et al* 2002) focuses on 20,000ha of protected forest and peatland including two national parks, one nature park and several mire and old-growth forest protection areas. About two-thirds are protected for mires and one-third for old-growth forest. The study is particularly interesting for the comparison of local and regional level CBA. This involves some difficult reasoning regarding where to draw boundaries, and also means that some items that would be considered transfer payments in national CBA are included in these analyses. Results are interesting in demonstrating the impacts of benefits leaking out of local area – in some scenarios the local benefit-cost ratio is less than one, although the regional and national benefit-cost ratios are strong. It is clear that at a national and regional level, the conservation is beneficial, but it seems not to be at local level. As in the Indian cases cited above, policy conclusions could include the desirability of some financing instrument or compensation for local communities. Carbon budgets and values are not considered, and would strengthen arguments for conservation, as well as potentially introducing a compensation vehicle.

One problematic feature of this study is the use of stated preference intended to focus only on non-use values. While it is possible that this can be achieved, it also seems likely that, given the focus on local/regional populations, aspects of their use values may have influenced their responses. The survey allowed for zero or negative WTP bids; it seems unlikely that many respondents would be anti-conservation *per se* (i.e. they would not have negative non-use values) so those giving negative bids are presumably valuing not just the conservation outcomes, but rather their overall view on the net value of conservation – i.e. taking the opportunity costs into account. Since these opportunity costs are separately included, this could result in double counting of these costs.

Case 10: 'Old growth forests, Finland III' (Siikamäki and Layton, 2005) addresses the question of 'optimal conservation', deciding how much land to protect. The application is not to the whole country but rather to a conservation network, a mosaic of small protected areas outside existing large conservation areas (which are mostly in the North). The study is one of a number that focus on stated preference methods for estimating both costs and benefits (see also Case 25: ‘Riparian habitat restoration, France’ and Case 31: ‘Wild goose conservation in Scotland’). The cost estimates are made via an econometric model of individual landowner enrolment in conservation programme: both participation, and amount of land enrolled, at different conservation payment levels, based on a contingent behaviour survey. This is equivalent to estimating the opportunity cost to the landowners. The focus is on non-industrial private forest owners outside protected areas, with the proposed payment instrument being an extension to existing instruments of incentives and easements for conservation activities (and so realistic and familiar to respondents). Benefits are based on pooled data from contingent valuation and choice experiments.

The results from these surveys were used to estimate demand and supply curves for conservation land. Depending on the underlying assumptions, marginal costs and benefits are balanced around approximately 2– 2.5% of private forests protected, at a payment of around $6000 per ha. Total benefits (sum of consumer and producer surplus) at this level are $1.5bn to $3bn depending on assumptions. Since about 1% of forestland is already protected by current environmental regulations, a doubling of currently protected areas would be justified based on these results.

This study illustrates how simultaneous analysis of marginal costs and benefits of conservation can be useful to highlight trade-offs, and for determining the optimal extent of specific policy options (in contrast with analyses focusing on binary ‘all or nothing’ policy comparisons). The focus on marginal costs and benefits helps determine what magnitude of conservation may be economically sensible and whether any, all, or none of the feasible conservation alternatives generate net benefits. On the other hand, the focus only on stated preference methods means that valuations are limited to those features that individual consumers/landowners are aware of and consider in their choice-making. This can mean that some factors may be overlooked – for example carbon emissions/sequestration – and a fuller analysis might complement the stated preference results with estimates of such other impacts. The methods used here do not distinguish between different grades of land, beyond the basic focus on old growth and hotspot forest, and do not consider any spatial/mosaic effects – further research might profitably examine these areas. On the other hand, to the extent that the policy instrument under consideration is similarly spatially and contextually blunt (flat rate payments for forest owners opting in to the scheme), the assessment can be considered appropriate to the decision context.

2.3.2 Land use/conversion

One of the seminal references in the study of conversion and restoration is Balmford *et al* (2002) who highlight the valuation issues at the heart of the fundamental choice between conserving land in pristine state and converting it to other uses. They sought examples for which it was possible to match estimates of the marginal values of goods and services delivered by a relatively ‘pristine’ biome, and a similar biome converted to typical forms of human use. Studies were only included if they covered the most important marketed goods, as well as one or more non-marketed services delivering local social or global benefits. Figures were cross-validated with other estimates from similar places, and the comparisons across different states of a biome had to use the same valuation techniques for particular goods and services. Following a review of 300 studies, just five examples met all these criteria; all five demonstrated significant losses associated with the more intensive use. They present estimates suggesting that the total cost of an effective, global reserve program for biodiversity conservation on land and at sea would be around $45 billion/year, including compensation for the opportunity costs. Based on the mean proportional loss of value upon conversion recorded in the five case studies, and the per hectare value estimates from Costanza *et al* (1997), they argue that such a reserve system could secure an annual value in the order of $4400 to $5200 billion. Although this is clearly very approximate, the implied benefit-cost ratio of 100:1 leaves substantial room for error without threatening the general conclusion that increases from current levels of global conservation are likely to be beneficial.

Case 11: Tropical forest conversion opportunity costs

Grieg-Gran (2008) report the opportunity costs of avoided tropical deforestation based on returns per hectare for different land-uses in several countries[13]. This does not in itself constitute a cost-benefit analysis but is included here as it represents a very ‘low hanging fruit’ in the sense that these costs could be combined with some basic calculations of forest conversion carbon budgets to check threshold carbon prices at which conservation becomes optimal (even without considering all the other benefits to biodiversity, watershed protection and so on).

Case 12: Forest protection, Guyana

Office of the President, Republic of Guyana (2008) is a good example of a carefully constructed partial CBA focusing on direct economic costs to local populations contrasted with global conservation benefits. Although the global benefits are not valued in detail, in this case it is sufficient to focus on carbon values to make the argument that investing in Guyanan forest conservation would be a cost-effective method of abatement from a global perspective; and the local analysis makes it very clear that without side-payments the economically optimal approach for Guyana is to exploit its forests. So although the CBA is partial, it leads to robust and important policy conclusions.

The clear objective of the study is to make the case for international investment in forest protection, and the headline conclusion is that the carbon abatement cost from such investments is roughly $2 to $11/t$CO_2$e, which compares favourably with most other abatement options available; ‘yet today, the world provides virtually no support to protect rainforests despite enjoying significant value from the ecosystem services they provide.’

Another feature of this study is that it takes account of uncertainty about future costs and benefits. Cost estimates are presented as a skewed probability distribution, with range $4.3 billion to $20.4 billion, most likely $5.8bn.

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[13] Bolivia, Brazil, Cameroon, China, Costa Rica, Democratic Republic of the Congo, Ghana, Indonesia, Malaysia and Papua New Guinea

Case 13: Tapean forest, Cambodia

This case (Bann, 1997) presents four cost-benefit calculations looking at the Net Present Values of different land use options for a high biodiversity tropical forest. Valuation methods used are market prices and replacement costs for NTFPs (non-timber forest products) and value transfer for biodiversity, carbon storage, watershed benefits and timber extraction.

This case illustrates the use of value transfer techniques to value non-timber forest products, which are to a large extent non-marketed. The study concludes that traditional use of the forest by local communities has a greater net present value than timber extraction, even under conservative assumptions which make it likely that the benefits of the traditional use of the forest are undervalued while timber extraction is overvalued, and with certain value categories omitted (notably recreation and tourism, cultural, option and existence values).

Case 14: Hypothetical forest area, Cambodia

Sasaki and Yoshimoto (2010) present a Partial CBA for managing a hypothetical 1 ha of forestland against six land-use options: business-as-usual timber harvesting, forest management under the REDD-plus mechanism, forest-to-teak plantation, forest-to-acacia plantation, forest-to-rubber plantation, and forest-to-oil palm plantation. REDD-plus is essentially a funding mechanism for global climate change mitigation that extends 'reducing emissions from deforestation and forest degradation' (REDD) with the addition of 'promoting sustainable forest management and enhancing carbon sinks'.

This study illustrates how partial conservation can be supported by CBA arguments. The case considered is not forest exploitation versus forest protection; but rather different forms of exploitation with more or less damaging impacts. REDD+ is shown to be potentially effective in encouraging legal logging with lower carbon and other impacts, though these reduced impacts on ecosystem services are not valued or discussed in detail. Their inclusion could only strengthen the case for REDD+, but could also encourage greater use of full protection in certain areas – the need for such protection in areas of high ecosystem service or biodiversity values is highlighted by the report, without any attempt at quantifying the benefits or determining which areas should be protected. It is also noted that if the REDD+ mechanism ignores wider co-benefits (NTFPs, non-carbon ecosystem services) then high-biodiversity, lower tree density forests 'would not be attractive to REDD-plus developers, which would result in high-biodiversity forests being converted to other land uses'. In other words, even though the partial CBA arguments presented here are sufficient to establish a case for REDD+ in general, relying on them without attempting to value the co-benefits could lead to REDD+ focusing in the wrong areas.

Case 15: European cork forest conservation

European Cork Industry Federation (2007) presents a partial CBA focusing on the costs of maintaining existing cork forests (2.7m ha worldwide, requiring expenditure around €340m per year for reforestation and densification), and the benefits of cork forests to livestock rearing, hunting, provisioning, and to the cork sector. This does not consider the non-market benefits of cork forest conservation actions, which could be the most significant values. In particular, there could be very substantial tourism values, and values for water quality, soil conservation, carbon sequestration (tree species in cork forests have higher than average carbon sequestration rates), biodiversity, and landscape values. Costs of managing various activities should also be considered. It appears likely that expenditures to protect cork forest ecosystems would be justified, but further analysis of these ecosystem service values would be useful.

Case 16: The Willamette River watershed, Oregon, USA

The Willamette River watershed is bordered on the east by the crest of the Cascade Range and on the west by the crest of the Coast Range. In this study, a spatially explicit landscape level model, InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) is applied to three stakeholder-defined scenarios of Land Use/Land Cover change: 'plan trend' (business as usual), 'development' and 'conservation' scenarios. Comparison of returns under different scenarios in order to analyse the economic and biological consequences of alternative land-use patterns: an economic model is used to identify efficient land-use patterns that maximise economic return for a given level of biodiversity (and vice versa for the biological model). Market returns and carbon sequestration are valued in monetary terms, but no estimates are made for amenity values, biodiversity conservation, or ecosystem services other than carbon sequestration and the marketed provisioning services, due to lack of data. The modelling demonstrates that, with an appropriate spatial management of land use, it is possible to sustain a high level of biodiversity and at the same time generate high economic returns in the Willamette Basin. It also shows that the divergence between private and social optima can be removed by a side-payments system compensating landowners for opportunity costs of conservation, via a payment for carbon sequestration benefits.

This case study demonstrates possible practical impacts of ecosystem valuation techniques. The Willamette Partnership is currently (September 2009 to September 2011) running a pilot of an ecosystem credit accounting system, which aims to 'quantify the flow of benefits and impacts for multiple ecosystem services stemming from actions taken on a site', verifying, registering and tracking these over time, and providing cost-effective mechanisms for restoring sites via the ability to sell multiple types of credits from the same site[14]. At present, credit systems cover wetlands, salmonid habitat, upland prairie and water temperature. Short-term priorities for new credit 'currencies' include water quality (nitrogen, phosphorous and sediment), water flow, carbon, and 'generalised rare habitat'. Further details are available via the Partnership website[15].

Several studies take a restricted approach to CBA and focus on comparing carbon values to opportunity costs. For example Damania *et al* (2008) present figures for Riau Province in central Sumatra, which has populations of critically endangered

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[14] 'Agreement in Concept on Ecosystem Credit Accounting System': http://willamettepartnership.org/tools-templates/Agreement%20in%20Concept%20signed.pdf
[15] http://willamettepartnership.org/ecosystem-credit-accounting/Willamette%20Partnership%20Presentation%203.3.10.pdf

Sumatran tiger and the endangered Sumatran elephant. Sixty-five percent of its original forest cover is already lost, and a 'business-as-usual' scenario predicts the remaining cover will decline from 27 percent today to just 6 percent by 2015. There are huge carbon emissions associated with this. Where present value returns of arable land can be as low as $100 to $150 per hectare, clearing a hectare of tropical forest could release 500 tons of CO_2, which even at $10 per ton of CO2 equates to $5,000 per hectare. However this ignores the opportunity cost of timber; and arable land is unlikely to be the highest value alternative use. Greig-Gran (2008) (see Case 11: 'Tropical forest conversion opportunity costs') reports total opportunity costs for the highest value land use (large scale oil palm) including one-off timber harvest benefits as $4439/ha (Present Value over 30 years at 10%) – suggesting that the carbon benefits of conservation could indeed be sufficient alone to outweigh the opportunity costs.

Other studies include Yaron (2001) who reports for Cameroon that conversion of forest to agriculture increases private benefits (food and timber) compared to sustainable forestry, but these are outweighed by loss of social benefits (non timber forest products, sedimentation control, flood prevention, carbon storage, option, bequest and existence values). Conversion to oil palm and rubber plantations yields negative private benefits, after allowing for market distortions. Kumari (1995) measures for Malaysia how high intensity, unsustainable logging increases private benefits of timber supply but reduces non forest timber products, flood protection, carbon stocks and endangered species. Convention on Biological Diversity (2001) covers a wide range of estimates and examples, including costs and benefits of forests under different management regimes.

Van Vuuren and Roy (1993) estimate the value of three freshwater marsh types in an agriculturally productive area in Canada threatened with conversion through draining. Their analysis shows that draining would yield private benefits, artificially inflated by drainage subsidies, but these are offset by the lost social benefits of hunting and angling, estimated via travel cost studies. Economic values were higher for conservation than for conversion by a mean of around 60% (around $8800 compared with $3700/ha, after correction for subsidies). Had the full value of other wetland ecosystem services been included, this conclusion would have been strengthened (Turner *et al* 2003), so it is likely that the conclusion is robust.

Habitat destruction

Case 17: Mangrove conservation, Southern Thailand

Sathirathai and Barbier (2001) present a cost-benefit analysis of different land-use options for mangroves in Southern Thailand. Thailand earns more than $1.2 billion annually from exporting frozen shrimps. Mangroves had been disappearing at the rate of over 6,000 ha per year in Thailand, mostly due to conversion to shrimp farming, and Thailand has lost around 15% of mangrove cover over the last 30 years (Mangroves for the Future, 2008). Local communities who have traditionally utilized mangrove resources have not had the legal right to protect mangrove forests from conversion, though there have been proposals for a law to change this via a

'Promotion of Marine and Coastal Resources Management Act'[16]. There are also important restoration projects underway in parts of Thailand.[17] But mangrove loss continues to be a major global problem, with destruction of around 0.7 percent per year due in particular to coastal construction and shrimp farming, and also damage following tsunami events (Spalding *et al* 2010).

The study compares the Net Present Value of three different land-use scenarios (conservation, conversion to shrimp farms, conversion plus forest rehabilitation). The direct use value of mangrove resources was estimated by using market prices for extracted products which are sold, and replacement costs for products which are used for subsistence. Surveys and interviews were carried out with villagers to identify what products local communities extracted from the mangrove forests. The value of mangrove-fishery linkages are calculated using a production function approach (Ellis-Fisher-Freeman model). Coastline protection and stabilization by mangroves is valued through replacement cost of constructing breakwaters. The conclusion was that the value of conserving mangroves in Surat Thani Province in Southern Thailand is higher than that of converting mangroves to shrimp farms. This depends largely on coastal flood protection services. Using a 10% discount rate, the net present values per ha of the different options were estimated as:

- NPV of conservation of mangroves: $8,836 per ha
- NPV of converting to shrimp farms: $209 per ha
- NPV of converting to shrimp farms with forest rehabilitation: -$5448 per ha

This case study is extended by Hanley and Barbier (2009) drawing largely on the 2001 study and also on Barbier (2007), and discussing the possibility of non-linear functional relationships between area and function/service (see Barbier *et al*., 2008). These extensions demonstrate the practical importance of taking into account non-linear relationships between value and area. They show that using an average value for the storm protection value of mangroves in an area of Thailand ($1879 per ha), mangrove conservation clearly dominates conversion for shrimp farms. However, using the marginal values, and therefore taking into account that small reductions in mangrove area have relatively limited impact on flood protection values, this result is nuanced: the highest values overall occur if there is, in this case, 20% mangrove conversion for shrimp farms, and 80% conservation. Of course there is a strong spatial component to the value – the flood defence value of any given hectare depends strongly on where it is and what people and infrastructure it protects, as well as on the extent of mangrove round about. The 20% earmarked for conversion should not be any randomly selected 20%, but the 20% giving the least reduction in coastal protection values.

Taking non-linear values into account is also very important in determining the appropriate level of mangrove restoration where they have already been destroyed.

[16] So far as we are aware, this is still at the draft stage, though it is being promoted by NGOs. See http://www.iucnael.org/index.php?option=com_docman&task=doc_download&gid=388&lang=en

[17] See for example http://www.mangrovesforthefuture.org/Projects/Large-Projects.html

Barbier (2009) reports restoration costs with a present value around $9000 per ha. Considering the average value of flood protection (present value around $11000 per ha) would suggest that restoration is profitable. Looking at marginal values would reveal the more accurate conclusion, that it is profitable *up to a point.* This reasoning can help ensure that scarce resources for restoration and conservation activities are best targeted.

Further examples of mangrove conversion are given by Bann (1997), Gammage (1997) and Janssen and Padilla (1999).

Case 18: Coral conservation, Philippines

Hodgson and Dixon (2000) present an analysis of halting deforestation in a watershed area to prevent sedimentation of downstream coral reefs. In 1985 a logging company began operations in the watershed surrounding Bacuit Bay, causing rapid soil erosion and sedimentation to the bay, affecting fisheries and a rapidly expanding tourist industry based on foreign and local scuba divers. Hodgson and Dixon (1988) had shown that the costs of foregoing income from logging was outweighed by the benefits of preserving the fishing and tourism industries. This study reviews the previous one and qualitatively reviews what has happened in the area since. It shows that the last study severely undervalued the growth of the tourism industry, and confirms the benefits to the tourism and fishery industries far outweigh the costs of banning logging in the area. However it should be noted that the whole area of El Nido is ‘a showcase of the Philippines' geological and biological diversity’ which was designated as a turtle sanctuary in 1984, a marine reserve park in 1991 and a managed resource protected area in 1998, suggesting that this result could not simply be transferred directly to any case of logging affecting coastal and marine habitats.

Case 19: Blast fishing, Indonesia

Pet-Soede *et al.* (2000) is a cost-benefit analysis of the middle-term (20 years) effects of blast fishing on Indonesian society. They assume, based on observations, that blast fishing is taking place in 50% of the reefs in Indonesia, and further assume that this would destroy 75% of coral after 20 years of blast fishing. They model costs and benefits for hypothetical situations on 1km^2 of coral reef in pristine condition. The costs and benefits for fishers are modelled using market prices for revenue from fish and costs of fishing, including all the equipment, fuel, expected police ‘fines’ and the opportunity cost of labour. Against this are set the losses associated with reef destruction, including loss of revenues from non-destructive fishing gear, and value transfer estimates for loss of coastal protection and loss of tourism. Protection and tourism values were transferred from existing studies, based on avoided damages and on tourism income. The spread of values is large: ‘The annualised figures for tourism per km2 of coral reef are taken to be US$ 55,900 representing a ‘high value’ and US$ 333 for a ‘low value’ situation’. Similarly, the annualised values for coastal protection were US$ 61,100 for the ‘high value’ scenario and US$ 2,800 for the ‘low value’ scenario. Overall it is clear that blast fishing has a net negative economic effect, and this result is robust to various values used for sensitivity testing. At a 10% discount rate, with low estimates for tourism and coastal protection, the net present loss of blast fishing is estimated at \$33,900 per km^2; this falls to a loss of \$306,800 per km^2 using the high estimates. White *et al* (2000) present similar results for the Philippines.

Case 20: Coral mining, Indonesia and Sri Lanka

This case (Ohman and Cesar, 2000) is a comparison of cost-benefit analyses of two sites of coral mining. The benefits are estimated using market data, and show the industry is privately profitable in both areas. Costs are estimated for fisheries losses, coastal protection losses and lost tourism potential. For Lombok, the estimates range from -\$33,000 to -\$762,000 per km^2 in net present value while for Sri Lanka the range is -\$76,000 to -\$6,946,000 per km^2. However in part this very high value is due to use of a replacement cost estimate for coastal protection: we might expect that such expenditures for man-made structures would only be incurred in front of highly valuable assets, so the conclusion should be nuanced to suggest that coral mining is especially damaging in such areas. The higher estimates for tourism losses may also be overstated, in the sense that the value of marginal tourism is likely to vary significantly with the amount of space dedicated to it. But across the full range of values presented, the conclusion is that coral mining is not cost-beneficial at the national scale.

One interesting aspect of this study is the strong differences between specific value estimates, and in the bottom line results, across the two cases. There are important differences in the fisheries ecology in the two areas, in the methods used for producing lime from coral, and in the values estimated for tourism and flood protection. This suggests that values may not be transferable for these kinds of assessments/services, although the ‘bottom line’ conclusions do converge..

2.3.3 Restoration

Another key focus for cost-benefit work has been the restoration of converted or degraded habitats. This is essentially the other side of the land conversion decision, and all the examples here are from industrialised countries. Benayas (2009) presents a meta-analysis of 89 restoration assessments in a wide range of ecosystem types across the globe, showing responses for different habitats. Results indicate that ecological restoration increased provision of biodiversity and ecosystem services by 44 and 25%, respectively. However, values of both remained lower in restored versus intact reference ecosystems. The paper does not assess the costs or opportunity costs of the restorations.

Case 21: The National Forest, UK

This case (eftec, 2010) is an example of using a broad-brush methodology designed for quick application at regional or national level. The main assumptions are that it is possible to represent a complex network of specific sites by a general typology of habitat types, proximities to populations, and access levels. Different woodland types within this typology are then associated with different levels of certain ecosystem services and values. Timber is valued at market values; carbon at official UK rates; recreation based on visitor numbers and value transfer; aesthetic and biodiversity values based on approximate value transfer. Non-use/cultural heritage value transfer is discussed but not used, in order to avoid double counting with other categories.

This approach clearly involves rough approximation. A more detailed study involving GIS (Geographical Information System) analysis to link specific sites to specific services and values is planned. But despite the approximations, studies such as this can give a clear indication that conservation (here woodland planting and access creation) spending is broadly justified. An earlier study (eftec, 2010) similarly demonstrated that the Public Forest Estate in England was a large net provider of ecosystem service benefits, notably recreation and carbon sequestration, and that the conclusion that benefits exceed costs was robust across a wide range of sensitivities. Using the method to compare different future scenarios suggested that a strategy of investing in urban and peri-urban woodlands, with recreation access, would be the best way of allocating funds. The approximate approach may not be as accurate as a detailed, spatially explicit modelling exercise; but it is much quicker and cheaper. Bearing in mind that cost-benefit analysis should be proportionate to the values at stake and the requirements of the decision context, it may often be the case that a faster, cheaper, less accurate approach may be the right one to adopt.

Case 22: Agriculture-forest conversion, Wales

This case (Bateman *et al*., 2005) is an example of a much more detailed and spatially explicit approach to a similar decision context, selecting most cost-beneficial sites for establishing multi-purpose woodland. It is interesting in particular for the combination of GIS and value transfer methods to carry out a CBA at a very broad national scale. The overlaying of different value layers in the GIS gives a clear picture of where tree planting is likely to be most beneficial, from a national perspective, and this could guide the use of scarce resources for forestry projects. The valuation is based on modelling timber production value, agricultural opportunity costs, carbon sequestration and its value and recreation potential and value, all conducted within a GIS framework at 1km resolution. Results suggest that the Net Present Value of changing land use to woodland ranges from strongly negative (planting on peat soils leading to both agricultural loss and significant loss of stored carbon) to strongly positive (near to highly populated areas where recreation benefits will be large). The transfer function approach developed is explicitly designed to incorporate local environmental characteristics within production and value estimates, so for example the marginal values of recreation would change as more forests are planted. Thus the actual values in the GIS would need to be updated iteratively as forest cover increases. A remaining weakness is the lack of explicit consideration of substitution effects in the recreation valuation studies, although this is being addressed in more recent work (Jones *et al*., 2010; Bateman *et al*., 2010).

Case 23: River Elbe floodplain restoration, Germany

This is a partial CBA of restoration along River Elbe (Germany) through dike shifting, reducing agriculture impact and constructing fish ladders (Meyerhoff and Dehnhardt, 2007). Restoration options of 10,000ha and 15,000ha are considered. Valuation is based on a contingent valuation (CV) study for use and non-use values, avoidance costs, engineering costs and land opportunity costs. The CV approach is similar to that taken in Case 29: 'Blackwater Estuary' and other studies that use a composite environmental good to cover all the aspects respondents are likely to consider/be aware of when responding to valuation questions, and another method for more complex or indirect values arising from ecosystem processes.

Some potentially important impacts are not valued in this study: notably recreation and flood protection benefits, which might both be significant and almost certainly positive. Carbon benefits and/or methane dis-benefits are not considered and could be significant, but ambiguous in direction. Despite these omissions, the CBA provides good evidence for positive benefit-cost ratio in this case ranging from 2.5 to 4.1 depending on the scenario; and the bulk of omitted values are almost certainly benefits.

Case 24: Skjern river, Denmark

This case is an ex-post CBA of the restoration of the River Skjern (Dubgaard, 2003) from a channelled river to a meandering course, including creation of outflows from the river to the Fjord with the intention of forming a delta of c. 220ha in time, and creation of a lake of c.160ha. Land within the project area will flood periodically; 1,550ha of arable land is switched to extensive grazing. Valuation uses value transfer, market prices and replacement costs to value costs and benefits. The costs are front-loaded, with the result that the Net Present Value appears positive only up to about 7% discount rate. A DKK32 million grant from the European Union is counted as a benefit (cost reduction) in the study, because from the Danish perspective this was (arguably) additional funding that would not otherwise accrue to Denmark. From the European perspective, of course, this represents a transfer payment that should not appear in the CBA. The removal of this would somewhat reduce the break-even discount rate.

Case 25: Riparian habitat restoration, France

This study (Amigues *et al*., 2002) presents an attempt to value the creation of a 10-50 metre strip of land along the Garonne river as a right of way and also to restore riparian habitat via a dual stated preference approach. Riparian habitat mitigates flooding, serves as temporary reservoirs, prevents bank erosion, improves water quality through the filtering of nitrates and other compounds from run-off, and preserves habitat for plant and animal species. However in this study these effects are not separately identified and valued. Instead, town residents without land on the Garonne River were asked how much they were willing to pay to preserve riparian habitat along the river. Landowners were asked how much they were willing to accept compensation to preserve a riparian strip of land from 10-50m in a 'natural' state. On the face of it, the resulting Net Present Value estimates seem positive and the benefit-cost ratio reasonably strong. However the authors flag doubts about the validity of the WTP survey, with strong embedding, little consideration of income constraints, and general signs of treatment as a 'donation' rather than a value. The descriptions of the valuation goods do not match and are ambiguous: a strip of land between 10 and 50 metres in the willingness-to-accept compensation (WTA) question and 70 km of riparian habitat in the WTP case – which further complicates interpretation. So although the approach taken here is very interesting and could be promising for further development, this particular implementation has some shortcomings. Further research into the use of dual stated preference methods for evaluating conservation decisions relying on private landowner compensation would be useful.

Case 26: River Bouvade restoration, France

This study also addresses a restoration proposal, in the context of a river not used for recreation, flowing through an area with a population of only 4000. The study uses contingent valuation to give an estimate of the non-market value related to achieving good status on the Bouvade river, compared to costs, within the context of assessing if the costs could be considered disproportionate in the sense of the EC Water Framework Directive. here is no attempt to assess ecosystem service impacts directly, the only benefits considered are those reflected in the contingent valuation study. The study suggests that benefits from river restoration are substantially lower than costs, with a benefit-cost ratio of around 0.05. This is partly because the river is very little used for recreation, and there may also be an impact of timing and payment vehicle (asking about increased water bills during the financial crisis). There is also a population effect: there are only about 1750 households in the area, and with expected restoration costs in excess of 2 million Euros, rather high WTP values would be required to make the scheme appear beneficial. It is worth considering whether there might be households or businesses outside the immediate area, and in particular downstream along the Moselle, willing to contribute to this project, either for non-use reasons, or associated with the effects of water quality improvements downstream.

Case 27: River Gardon downstream restoration, France

This case has the same context, but differs in being a river used for recreation, with around 165,000 regular users for walking (€19.30), swimming (€12), kayaking (€12.60) or fishing (€12.80) (figures are consumer surplus estimates per visit based on a travel cost survey). The total value is €45m, but as the study stresses this is not a value that is at stake, but rather contextual information. To estimate the possible value improvements associated with achieving good status for the river, contingent valuation was used, with an on-site survey for users, and a telephone survey for non-users. WTP per household per year for improvements was around €35 for walkers and fishers, around €14 for swimmers and kayakers, and around €30 for non-use, making a total estimate of €2.86m per year. Overall estimates were €38m for users, and the same (by chance) for non-users; costs of the scheme are €40m, so the Net Present Value of the improvements is estimated as €36m. Delaying the scheme by 6 years would reduce this to €24m, and delaying by 12 years would reduce to €16m, from today's perspective. Sensitivity analysis supports the conclusion that the restoration actions are very likely cost-beneficial.

A particularly interesting feature of this study is the availability of a similar study, carried out for a similar river. As an experiment, benefit transfer was carried out between these studies, using a point value transfer, and a function transfer. The errors are shown in Table 2. The errors are more substantial for non-use values than for use values, but overall seem acceptably low, suggesting that we can have some confidence about transferring values between very similar resources.

Table 2 Transfer errors between Gardon and Loir cases

Population	From Loir to Gardon		From Gardon to Loir	
	Value transfer	Function transfer	Value transfer	Function transfer
Users	1% to 5%	10%	1% to 5%	**2%**
Non-users	8% to 16%	16%	19% to 24%	**5%**
All	**7% to 11%**	**6% to 12%**	**8% to 12%**	**9% to 16%**

Source : Lettre Evaluation, Novembre 2007 :
http://portaildoc.oieau.fr/entrepotsOAI/OIEAU/44/223221/223221_doc.pdf

Aylward *et al* (1999) is a (rare) example of a study concluding that restoration is not justified on cost-benefit arguments. They studied one of the only micro-watersheds in the upper Arenal watershed of Costa Rica largely converted to pasture, Río Chiquito. They present analyses of hydrological function and resulting externalities suggesting that the net effect of having pasture in place of forest (and of having pasture interspersed with cloud forest) are to cause an increase in hydroelectricity production. The value of negative externalities would need to exceed $600/ha in order to justify restoration efforts. They argue that, given the ready availability of large areas of primary forest adjacent to Río Chiquito that are already preserved, biodiversity, existence and ecotourism values associated with an increase in forest cover would be minimal. Therefore the only realistic source of the necessary added value for restoration would be carbon sequestration. They further argue that this value is unlikely to exceed $200-$300 per ha, and conclude that restoration is not justified. However it would be interesting to revisit this conclusion using more recent estimates of carbon values.

Worral *et al* (2009) look at restoration of degraded peat in UK uplands; the study has a detailed science base, but slightly less detail on the cost estimates and benefit values. They conclude that peat restoration in UK uplands may be justified by carbon impacts, but that this depends on the value of carbon. The model was run for a decade from 1997–2006 and applied to an area of 550 km^2 of upland peat soils in the Peak District. The study estimates that the region is presently a net sink of 62 ktonnes CO_2 equivalent (CO_2e) per year; management interventions targeted across the area could increase this to 160 ktonnes CO_2e/yr (96% of cases where re-vegetation was possible gave a CO_2e benefit, but in only 20% of areas where drain or gully-blocking was possible was a CO_2 equivalent benefit observed). Taking into account the costs of restoration (£1500/ha, ±50%) and a value of carbon based on the then Defra guidelines (£26/tonne CO_2e), the study suggests that 51% of those areas, where modelled interventions resulted in a carbon benefit, would show a profit from carbon offsetting within 30 years.

It should be noted that the restoration may also have significant additional benefits not covered in Worral *et al*, including improvement in raw water quality (leading to lower treatment costs for drinking water), cultural services (recreation and aesthetic values) and biodiversity conservation. Some of these factors are covered, but at a much less detailed spatial resolution, in Tinch *et al* (2009), which presents six case studies, including the Bleaklow plateau, an ongoing post-fire restoration project within the area covered by Worral *et al*. For Bleaklow, the combined carbon, recreation and non-use benefits of restoration are estimated by Tinch *et al*. to exceed restoration costs by a ratio of approximately 2.8:1 over 50 years

2.3.4 Water supply

Studies assessing the costs and benefits of different options for coastal and riverine flood protection, often contrasting 'hard' engineering with 'soft' nature conservation options, are increasingly common and sophisticated. Many of the studies do link back to the same meta-analyses of wetland habitat values (Brander *et al*, 2006; Woodward and Wui, 2001). It is also possible to contrast the results of scheme appraisals with and without the inclusion of environmental values (see Tinch and Ledoux 2006). A commonly-discussed example here is the Catskills watershed supplying New York. (Chichilnisky and Heal 1998; Heal *et al* 2005; Hanley and Barbier 2009). The study estimated that costs of replacing this natural service would be $6-8bn, compared with $1-1.5bn in land purchase, protection and restoration to maintain the natural service. A number of more recent studies cover similar topics, in industrialised and developing countries.

Case 28: Upper Tuul watershed, Mongolia

This study (Emerton *et al*., 2009) looks at the conservation of the Tuul basin, a catchment area of almost 50,000 km^2 from which Ulaanbaatar derives its water. Ecological conditions in the upper watershed have a direct link to the availability of surface water and groundwater downstream in Ulaanbaatar. A recent study shows that as the ecosystem is degraded and land cover is lost, average runoff will increase and the river's mean annual maximum and low flows will be intensified. Diminished discharge would lead to a lowering of the groundwater table of between 0.24 metres (under a continuation of the *status quo*) and 0.4 metres (under a scenario of rapid degradation). In 25 years' time, daily water supply in Ulaanbaatar would be reduced by some 32,000 to 52,000m^3 respectively. In contrast, conservation and sustainable use of the upper watershed would protect current river flow and groundwater levels. Weighing up the gains (sustained water supplies to Ulaanbaatar) and losses (reduced land values in the upper watershed), conservation of natural habitats in the Upper Tuul is the most economically beneficial future management scenarios. The conservation and sustainable use scenario yields a Net Present Value, over 25 years, of $560 million. This is higher than the NPVs generated under either a continuation of the status quo or a scenario of rapid ecosystem degradation.

The Upper Tuul case is a good example of a simple, partial CBA, but based on science modelling permitting the valuation of a subset of goods and services and yielding clear, policy relevant conclusions. lthough the valuation methods used could be criticised, this is unlikely dramatically to change the results. There is a clear conclusion that water supply function alone could justify the costs of conservation, but that for local people this would involve losses, hence effective conservation is likely to require compensation financing.

Kramer *et al* (1997) present analysis of upland forests in a watershed in Madagascar suggesting that the economic benefits of biodiversity conservation far outweigh costs in Madagascar. They estimated the flood protection value of these forests at $126,700 per year. Sustainable management of a network of 2.2 million hectares of forests and protected areas over a 15 year period was estimated at $97 million (including opportunity costs forgone in future agricultural production) but would result in total benefits of $150–180 million. About 10–15 percent of these benefits are from direct payments for biodiversity conservation, 35–40 percent from ecotourism revenues, and 50 percent from watershed protection, primarily from maintaining water flows and averting the impacts of soil erosion on smallholder irrigated rice production. The study considered potential winners and losers from forest conservation and pointed to the needs for equitable transfer mechanisms to close this gap, but emphasized that conservation will help to maintain or improve the welfare of at least half a million poor peasants. It contributed to a government decision to increase forest protected areas to more than 6 million hectares in Madagascar.

Emerton and Bos (2004) present evidence from a variety of studies that have assessed the value of ecosystem services by estimating the costs of replacing these through artificial substitutes. In some cases, this can support CBA, but the focus is generally on a single service (water supply). But it can be argued that it is enough to show that conservation is beneficial in terms of a single service, if the other ecological impacts are likely to be positive. The authors report a study of the annual costs of replacing the ecosystem services provided by the Martebo mire, Gotland, Sweden, through technologies that would be required to replicate them; costs have been estimated at $350,000 to $1 million. These services (and their replacements) included peat accumulation (replaced by artificial fertilisers and re-draining of ditches), maintenance of water quality and quantity (installing pipelines, well drilling, filtering, quality controls, purification plants, treatment of manure, pumps, dams), moderation of water flow (pumps and water transport), waste processing and filtering (sewage plants), food production (increased agricultural production and import of foods), fisheries support (fish farming). Certain goods and services could not be replaced. Although this covers a wider range of services, the use of replacement cost methods only is problematic: these are not estimates of the value of these services, and so could not be used in a CBA assessing whether or not restoration would be cost-beneficial.

WWF (2008) includes the example of water tanks used for irrigation in Andhra Pradesh, India. These are ancient village earth dams (1-10 ha in size) which functioned as storage tanks in the past, but have now deteriorated due to sharp population rise, mismanagement and full diversion of river water. Andhra Pradesh Government proposed a US$4 billion Polavaram Dam on the lower Godavari River, that would displace 250,000 people and inundate key habitats (including 60,000 ha of forest) to supply irrigation water. A WWF pilot project in 2004 assessed the costs and benefits of restoring tanks. Between 2005 and 2006, in Sali Vagu sub catchment 12 tanks with an area of 11ha and serving 42,000 people were restored through de-silting: 73,000 tons of silt were removed. The cost was $28,000 in cash from WWF and $75,000 from farmers in cash inputs and labour. The increased water supply and groundwater recharge resulted in less groundwater pumping. Water tables rose, reactivating some wells that had dried up (wells worth an average value of $2,330 each). An additional 900 ha was irrigated and the nutrient rich silt was spread over 602 ha, increasing crop production by Rs 5.8 million ($69,600) per annum. Irrigation of additional lands decreased the need for electricity for groundwater pumping, and wages paid for de- silting supplemented farmers' incomes. Fish production in ponds gave net profit of Rs 160,000 ($3,700). The tank restoration project also created artificial habitats for migratory and water birds (this was not valued). In the Maner River basin there are 6,234 water tanks covering 588 km^2 that could be de- silted at an estimated cost of Rs 25.5 billion (US$635 million). These could store an extra 1,961 Mm^3 of water (compared to estimated water use in the basin today of 2,000 Mm^3 per year) at a cost of US$0.32/$m^3$. Further, this water would be stored widely across the basin where more people could access it. By contrast, the government's proposed $4 billion Polavaram Dam would store 2,130 Mm^3 irrigation water at a cost of US$1.88/$m^3$.

United Utilities (UU) 'Sustainable Catchment Management Programme' (SCaMP), is a wider ranging example of water catchment restoration. SCaMP is a flagship conservation initiative in the UK, involving a partnership of UU, local farmers and a wide range of other stakeholders, formed to invest in conservation activities in 20,000ha of water catchment land in the North West of England, aiming to secure improvement in SSSI condition while protecting water quality. In Tinch *et al* (2009) the ecosystem service changes expressed in monetary terms were greenhouse gas regulation, recreation, and non-use cultural/heritage values. Biodiversity non-use values were not added due to the risk of double counting with the cultural values. The net present value estimate was -£4.8 million over 50 years, rising to £0.4 million over 100 years, but this did not include the water quality benefits: these are likely to be very significant, but very difficult to specify at this stage; there is also reluctance on the part of some key stakeholders to put a value on the water quality benefits, because of the political and legal situation surrounding water charging and investments in the UK. So while it is widely expected that the scheme is cost-beneficial, value estimates have not yet been used to demonstrate this convincingly. Tinch (2009) illustrates, for the SCaMP project, an alternative approach to assessing the costs and benefits of impact, developed by IEEP and others in the EC project 'Financing Natura 2000: Cost estimate and benefits of Natura 2000', which stops short of economic valuation of different service changes (see also Case 34: 'PRIOLO life project (Azores Bullfinch)').

2.3.5 Flood protection services

Case 29: Blackwater Estuary, England

Liusetti (2008) is one example of CBA methods applied to managed realignment schemes (i.e. the planned and actively realised shifting of a flood defence line, generally involving creation of intertidal or wetland habitats, and distinct from simple abandonment of defences) in the UK. There are several others (e.g. Tinch and Ledoux 2006, Tinch and Provins 2007, Everard 2009), but these were entirely based on value transfer, whereas Liusetti *et al*. (2008) involves primary valuation and has strong links to simultaneous research into fishery production functions for bass, and sediment burial estimates from biogeochemistry research, all within the same research project (ComCoast).

The case assesses four different options for the estuary, with varying levels of managed realignment/habitat creation: 'hold the line' (HTL), 'policy targets' (PT; meeting existing targets), 'deep green' (DG), 'extended deep green' (EDG). Market prices are used for coastal defence work (costs avoided) and fish production function, and (after adjustment for subsidies) for the value of agricultural land. Three carbon price estimates are used for the carbon, methane and nitrous oxide fluxes. Stated preference is used for a 'composite environmental benefit' that is intended to cover a wide range of impacts without double counting: recreation, aesthetics, water quality, and biodiversity. The study breaks total WTP down into use and non-use components, and the aggregation methods allowed for distance-decay and non-linear relationship with wetland area. Thus the estimates for the composite environmental benefit showed the diminishing marginal utility generated by provision of additional areas of high environmental quality: in the PT scenario (81.6ha wetlands) the value estimate is £6.3m/yr of which £4.4 is use value; in the DG scenario, with 10 times more wetlands, the value is only a little higher at £7.7m/yr, of which £5.8m is use value, while in the EDG scenario, with 30 times more wetland than PT, value is £8.3m/yr of which £6.4m is use value.

Results of the CBA show that managed realignment can be cost-beneficial if account is taken of non-marketed benefits, in particular for conservation and recreation. With a constant 3.5% discount rate, the highest Net Present Value is the 'deep green' scenario (£106m over 25 years, £192m over 100 years); much higher values arise if using a declining discount rate, and this can make the 'extended deep green' scenario preferable (simply because the lower discounting of long-term future makes it easier for long-term environmental benefits to outweigh near-term costs). The study has a good grounding in scientific analysis, including uncertainties, and is exemplary in exploring sensitivities to different time horizons, discount rates, values and assumptions.

Case 30: Polder Breebaart, The Netherlands

Buter *et al.* (2007) considers a smaller scale choice between traditional and overtopping resistant dykes. The 'traditional' dyke requires heightening to maintain 1:4000 year risk of failure. Overtopping resistant dyke is a cheaper option: an overtopping resistant layer on existing dyke, and new secondary dyke, with a zone between providing flood storage. This involved change from agriculture into a brackish tidal area used for nature oriented recreation. The case study develops this and also considers other possible forms of land use.

The case is dominated by the significantly higher costs of the 'traditional dyke heightening' options: the alternative is more cost-beneficial irrespective of land use. The model also determines that housing and greenhouses are less beneficial land uses, due to the need for terps (artificial hills). But the resolution between other options (agriculture, aquaculture, nature conservation with and without recreation) is low.

2.3.6 Species conservation

Case 31: Wild goose conservation, Scotland

This case study (MacMillan *et al*., 2004) demonstrates the use of stated preference and farmer-reported costs in a cost-benefit framework to assess the appropriate level of conservation activity. The study estimates the willingness-to-pay of the general public for specified goose conservation measures (4 options) and compares this with the costs of crop losses, estimated from a survey of farmers.

In principle the methods allow estimation of marginal costs per goose (via regression analysis of costs against goose numbers, available from counts), and also marginal benefits (via the WTP functions from the contingent valuation study). So in principle an 'optimal' number of geese could be identified. However the values could be quite approximate – in particular, a deliberative valuation workshop suggested that the contingent valuation estimates were over-estimating true WTP by about 3.5 times - and it is more conservative to limit assessment to the assessment of the policy options considered. Achieving a 10% increase in endangered species of goose had the highest benefit-cost ratio of 700:1, but achieving a 10% increase in all goose species also performed well (113:1). Despite some uncertainties regarding the precise interpretation of value estimates, the study is able to give robust policy support for measures to improve goose conservation, because the benefit-cost ratios are so high that even significantly scaled back value estimates would not result in net loss.

Case 32: Large Blue butterfly conservation, Germany

Watzold *et al* (2008) is particularly notable for its combination of ecological and economic methods. The study seeks to assess the optimal conservation level of Large Blue butterflies (protected by the EU Habitats Directive) via payments to conserve specific times and sequences of mowing regimes on which the species depends. The scenarios are based on modelling of the ecological impacts of 112 different mowing regimes, under different conservation budgets, with results expressed in numbers of butterflies per ha after 20 years. Valuation of the benefits is based on contingent valuation: to make it understandable to respondents, the numbers of butterflies per ha were converted to expression of changes in the extinction risk and the chances of seeing the butterflies in the wild. Opportunity cost estimates are based on detailed analysis of mowing regimes and associated costs and returns. Sensitivity analysis was conducted by adding random components to farmer responses to incentive payments, and to butterfly population dynamics.

Putting it all together gives a clear policy conclusion in this case. The initial marginal benefit of the first steps on conservation is €65,000/ha, while for the subsequent increments in conservation area it falls to €3000/ha, but since the marginal costs are only around €123/ha, conservation is optimal for all the areas considered (a 'corner solution'); even allowing for uncertainty, the lower bound of the confidence interval for the marginal demand exceeds the marginal supply by at least a factor of 15 in the entire range of proposed conservation projects, and so the highest proposed level of butterfly conservation (64 ha of meadow area) is the best choice. Higher levels of conservation area would be possible, since 64ha is 8% of the total area, but this is already sufficient to provide good conservation ('very low risk of extinction') and this is the best outcome considered in the stated preference study – therefore, under the estimates used in the study, there would be no marginal benefit to further increases in conservation area.

Case 33: North Wind Weirs salmon habitat restoration, USA

Batker *et al.* (2005) is ostensibly about excavation and replanting of 2 acres at North Winds Weir in the Green/Duwamish and Central Puget Sound Watershed. This aims to expand salmon habitat in the transition zone from fresh water to salt water through excavating and replanting native vegetation. But the effects of this relatively small project could be felt throughout the watershed, since 'Transition zone habitat is essential to salmon and may be so scarce that salmon extinction could result without increased transition habitat.'

Cost-benefit calculations for this project estimated the value of the site-specific ecosystem service improvements at $13,388 – $47,343 per year, totalling a present value of $384,000-$1.36m. As the site is in a high development value area, land acquisition costs ($1.9 million) plus estimated restoration costs ($1.79 million) amounted to $3.69 million, significantly greater than the benefit.

However, this figure did not take account of the off-site impacts, and in particular the fact that transition habitat is critical for salmon conservation in the whole watershed (and it can only be located where freshwater meets tidal salt water, between 5.5 and 7 miles from the river mouth). Taking its rarity into account, the authors estimated that it would be worth paying up to $19m per acre for the restoration. The study is therefore a good example of how the costs and benefits can have different spatial/marginal relationships – here, although the opportunity cost of the land is potentially high, it is considered easier for these uses to move than for the conservation use to move; the area is argued to be critical natural capital.

In this case, the restoration project went ahead, with contaminated soil removed in 2008/9, construction work in 2009, and planting throughout 2010. It highlights the importance of ensuring that the boundaries – both spatial and temporal - of any cost-benefit calculations allow the full effects of any decision to be taken into account. Here, this required critical natural capital to be evaluated in the assessment by considering the interdependence of this project and many other actions leading to salmon conservation in the watershed. This is reflected by considering that a significant part of benefits throughout the watershed, and/or a large amount of other conservation expenditure, would be lost without this critical project. But it is noted that further expenditures are also critical. Hence this is also an example of analysing a single part of a much larger strategic project.

Case 34: LIFE Priolo (Azores Bullfinch) project, The Azores

Cruz and Benedicto (2009) look at the costs and benefits of restoring and protecting threatened vegetation in order to protect the Priolo, a Critically Endangered species of bird. The main method is that developed under the 'Financing Nature 2000' project, involving grading of the relative importance of services and impacts but the study stops short of full CBA. However there is also the presentation of monetary estimates based on value transfer methods.

This study is interesting in the presentation of an alternative methodology, specifically designed for evaluating Nature 2000 sites, that grades the significance of different impacts. The extension of this to monetary valuation is only partly successful, giving an indicator of certain total value categories, but not providing strong cost-benefit analysis directly related to specific impacts from project expenditure. This illustrates some important points: firstly it is essential to have a clear baseline, and only to value the changes in services that result from a project; secondly the valuation methodologies must be carefully applied and reported.

2.3.7 Pollution control

Case 35: Eutrophication in Swedish coastal zone

Söderqvist *et al*. (2004) presents cost-benefit calculations for reducing eutrophication in the Stockholm archipelago. The benefits of the reduction of eutrophication, leading to a one-metre increase in water transparency, were estimated to be about SEK 60 million per year (travel cost method) and SEK 500 million per year (contingent valuation method). To achieve such reduction, it was assumed that a 40 per cent reduction in nitrogen load was needed, through a combination of measures including increased sewage water treatment and reduced fertilizer use. The total costs of such measures were estimated to be SEK 57 million per year. There is a risk of double counting if combining the travel cost (which accounts only for recreation values) and contingent valuation (which accounts for a wider range of values, including non-use, but could also cover recreation), but it seems that the costs could be justified purely with reference to the recreation values, and that taking a full range of values into account the BCR could be 8:1 or more. However it should also be noted that the location near the capital city means the use values are going to be much higher than in less populated regions, so this result could not simply be transferred to other parts of the Baltic.

Case 36: Polluted site restoration, Estonia

Karmokolias (1996) is a cost-benefit analysis of renovating the Kunda cement factory to meet World Bank/IFC environmental guidelines. The factory was the only major source of pollution in the area and at full production, emitted 129,000 tons of dust a year as well as SO_2 and NOx emissions. Water pollution also existed. The kilns that were in operation were installed between 1961 and 1972. The objective of the renovation was to modernize the plant, increasing production to 900,000 tonnes per year (up from 500,000) while meeting much stricter environmental standards.

So the motives were not primarily connected with conservation, but the case is interesting for the range of benefits associated with the reduction of dust emissions, including human health benefits, increased timber production, increased agricultural production, tourism benefits, and further benefits from SO_2 and NO_x emissions, as well as enhanced property values and reduced cleaning and painting costs. Further ecosystem service benefits are also likely, but not estimated. Overall the rate of return on investment estimated for the projects was almost 25%; in fact the raw materials savings are enough to give a positive benefit-cost ratio.

2.3.8 Agricultural systems

Although there has been an emphasis on conserving pristine or little-exploited ecosystems, recent authors (see e.g. Marris 2009) have noted that converted ecosystems may nonetheless have high biodiversity and/or ecosystem service values (these have been termed ‘Novel Ecosystems’).

Case 37: Conservation in Sumatran oil palm plantations

Bateman *et al*. (2009) develop a spatially explicit cost-effectiveness model for optimising conservation efforts within plantation land. They draw on detailed ecological data made available by a plantation (over 1000 kilometres of species observation transects) and model both the potential of different areas for supporting endangered mammals, and the opportunity costs of this conservation (based on detailed assessments of oil production and costs). These models were integrated into a spatial cost-effectiveness analysis indicating the optimal locations within a plantation for conserving biodiversity at least opportunity cost. This was then integrated with a further study of the price premium potential of conservation-grade palm oil, based on a stated preference survey of UK consumers’ WTP for certified conservation grade palm-oil products.

Consideration of the possible premiums yields insights into the optimal design of schemes for delivering biodiversity within existing plantations. At a plantation size of 30000ha, the lowest conservation area considered (5000ha) was beneficial to the plantation owner at a 15% price premium, but higher conservation areas required higher premiums. For smaller plantations, it is not possible to set aside adequate conservation land and still make enough of a price premium on remaining production to turn a profit. Conversely, larger holdings can set aside more land for conservation and still profit. One implication is that it is the larger holdings, often singled out for criticism over bad ecological performance, who might be most encouraged to engage in conservation under a price premium scheme. The authors conclude that conservation efforts could aim to benefit from conservation-grade and/or Fairtrade certification, by creating an economic incentive for a majority of plantations to see conservation as an economically beneficial undertaking.

Case 38: Ranch overstocking, Zimbabwe

Kreuter and Workman (1994) highlight the distinction between short-term financial returns and long-term economic returns, even when focusing only on the narrow economic profitability of an activity (here ranching). If for policy or other reasons economic actors are myopic and do not consider the long-term or uncertain impacts of their actions on the future profitability of their own activities, this can lead to inefficient outcomes. Cost benefit analysis taking this into account can highlight the need for policy instruments to address the problem, and the efficient levels to be aimed for. This rather simple example does this, but also flags up some important uncertainties requiring further ecological analysis (notably the relationship between different multi-species grazing regimes and vegetation dynamics). A more detailed, explicitly dynamic model would be needed for a full cost benefit analysis to be feasible.

Several studies take a restricted approach to CBA and focus on comparing carbon values to opportunity costs; see for instance the Damania *et al.* (2008) study concerning populations of critically endangered Sumatran tiger.

Case 39: Landrace conservation, India

Krishna *et al.* (2010) is the only paper that compares the 'actual' benefits and opportunity cost of cultivating traditional varieties versus improved varieties (Unai Pascual, pers. comm.). It is based on two complementary analyses: a supply-side analysis (240 farm households) using production function and hedonic methods to estimate the market (use) value arising from the landrace attributes of cultivated aubergines; and a demand-side study (629 consumer households) estimating the full consumption value of landraces via stated preference methods.

The models show the difference in yields (hybrids 112 Q/acre, landraces 95 Q/acre, a 15% difference)[18] and that the per-unit cost of hybrid cultivation is lower by about 29 per cent. However there is also an existing price premium for landrace varieties (around Rs 69/Q) thought to be enough to compensate for the yield disadvantage (cost difference Rs 45/Q) and the modelled net revenues are higher for landraces. In the demand analysis, the median WTP was estimated as Rs 6.00/kg for the consumer households favouring landrace products and as Rs 4.50/kg for all households. The current informal markets provide farmers a price premium of 31% (Rs 1.18/kg), and the increment attributed specifically for the landrace attribute is only 16% (Rs 0.69/kg). The authors conclude that there is potential for labelling to improve the price premium for farmers, facilitating landrace conservation.

The study illustrates how detailed microeconomic modelling can be used to demonstrate the production and consumption costs and benefits of different options for food production with different conservation implications. However the full environmental and distributional implications of the choices are not explored, and would provide additional input for decision makers. In particular, it is assumed that landrace conservation is beneficial, but there is no assessment of these conservation benefits.

Several studies have examined the costs and benefits of biological control. McConnachie *et al*. (2003, Table 1) review 10 benefit-cost studies of successful biological control programs, including four insect pests, four terrestrial weeds, and two aquatic weeds. For terrestrials, the benefit-cost ratios range from 1.9:1 to 24:1. Van Wilgen *et al*. (2004) estimate the costs and benefits of biocontrol of six invasive weed species in South Africa, where biocontrol has been practiced since 1910. They estimate benefit-cost ratios ranging from 8:1 for red Sesbania to 709:1 for jointed cactus. The estimates are sensitive to assumptions about the rate of spread with a 3 percent decrease in benefits for each one percent decrease in the rate of spread. Olson (2006) presents a review of relevant literature.

2.3.9 Urban nature

Case 40: Urban Nature, Nijmegen

[18] 1 Q = 1 quintal = 100kg, and is the standard measure for agricultural production in India.

Kirchholtes and Ruijgrok (2008) present CBA of urban nature for the Lindenholt neighbourhood in the city Nijmegen in the Netherlands. hey considered the current 'reference' situation, a 'grey' scenario in which all the green infrastructure is replaced with paved areas, and an 'optimised' scenario in which the existing infrastructure is replaced with a grid structure of trees designed for conservation benefits (18,000 trees, compared with 2,500 in the reference scenario). The construction costs were estimated as the difference in investment costs between the reference design and the two alternative designs. Maintenance costs were also estimated. On the benefits side, estimated impacts included production function estimates for the health impacts of particulate matter and NOx, noise impacts, flooding impacts, water treatment costs, enjoyment of the environment, recreation, climate regulation, reduced energy costs due to wind shelter effects, and impacts on travel times.

In comparison to the reference scenario, both alternatives had higher costs – in the case of the 'grey' scenario this is because the initial investment costs of paving were much higher than for simple planting, and the reduced annual maintenance costs did not overcome this initial disadvantage. The big difference however was in the benefits side, with almost €268m gain under the 'optimised' scenario, and €212m loss under the 'grey' scenario. Almost all of these impacts arose through the modelled impact on particulate matter (PM10), with the impact on aesthetic enjoyment also being significant (though an order of magnitude lower), especially in the 'grey' case.

It is interesting to note that, although carbon sequestration was valued at €49.5 per tonne, total values here were nevertheless insignificant in comparison to the PM10 impacts. Carbon values can dominate forestry assessments in rural areas, but in urban centres, the health impacts are orders of magnitude more important.

Overall the study provides strong support for the health-related benefits of urban trees. The conclusions are likely to be transferable to other urban situations, probably quite widely. But the conclusions are very broad-brush, in that the marginal impacts of additional trees are not assessed: the comparison is from the current situation (about 2500 trees) to 'all grey' or 'all trees' (18000 trees). Since most practical decision contexts are likely to involve more modest changes in tree numbers/proportions, further analysis would be desirable to map the marginal values as the tree density increases.

2.4 Overall Results

Based on the above analysis, conclusions can be drawn under two broad areas: (1) relating to methodological issues, and (2) to the weight of evidence available in the literature for different biomes and decision contexts. As stated above, Tinch *et al.* (2010) find very few studies that might be classified as perfect. Given this, an assessment of what the evidence tells us must be conditioned by how reliable the literature sources are. The authors highlight various issues in this regard.

Many of the estimations of conservation benefits in the CBAs reviewed use stated preference valuation methods. There are many known biases that good survey design can ameliorate, of which Tinch *et al.* (2010) focus on two. First, there can be a problem associated with the timing of benefits and stated preference responses.

Generally in stated preference research it is accepted that there is a problem with 're-contracting' if asking people to give willingness to pay amounts for a long run of years (for example a tax increase every year for 10 years) so it is often considered preferable to focus on one-time payments. However conservation activities may require many years to take effect, and inevitably studies often relate to the long-term protection of species or resources. Second, it is not always clear exactly what impacts/costs/benefits are covered by the responses. Hence there is the risk that respondents consider (and state a value for) not only a specific impact but also various changes that may be perceived as linked. This means that there is an associated double-counting risk if separate assessments are made of the values of such changes.

Linked to this second point is the question of risk and uncertainty, and in particular the treatment of probability of conservation expenditure success are crucial but understudied as stated by Wainger *et al.* (2010): 'Few systems capture the probability that the restoration will succeed in restoring ecosystem services…this shortcoming can profoundly affect expected benefits'. There could also be differences between pre-project-implementation and post-project-review studies (i.e. ex-ante and ex-post studies) vis-à-vis 'optimism bias'. Optimism bias refers to the tendency in at least some proportion of CBA studies to underestimate costs and overestimate benefits. The funding agency for the CBA may well have a desirable outcome that the study is intended to validate; the independence of the CBA practitioner may then be compromised.

Although these methodological issues are important, they should equally not be over-stated: the studies in the short-list have been screened so as exclude any that have fundamental methodological flaws.

Beyond these specific methodological issues, there are three broad caveats that should be considered:

1. Although a good range of literature has been reviewed in Tinch *et al*. (2010), the review cannot claim to be exhaustive.
2. It is not possible to claim that even a comprehensive review would cover a representative set of conditions: CBA studies are not commissioned, carried out, or published, randomly. For example, in proposing a new protected area, one would expect the most likely (highest value, least opportunity cost) areas to be proposed ahead of other sites; CBAs carried out on these priority sites would not be representative of more general decisions relating to 'randomly selected' areas of land. The reverse situation could also occur with some policy instruments – for example, the provisions for demonstrating 'disproportionate costs' under the WFD could lead to an increased number of studies commissioned largely in order to demonstrate that costs (of achieving good status) exceed benefits for certain water bodies (see Case 26: 'River Bouvade restoration, France').
3. For any given decision context, there are not many studies available: it is hard, even on the basis of three or four studies of similar resources, to draw general conclusions that may be transferred to other situations.

Having stated these caveats, Tinch *et al.* (2010) provide some general conclusions.

2.4.1 Purpose of the Studies

Rather than representing a single methodological approach, there are in fact many different interpretations and presentations of CBA in use. Some studies, for example, do not present aggregated net present value figures, in some cases focusing on a single year, but nevertheless clearly belong to the general 'CBA family' through the characteristics of measuring and comparing costs and benefits in monetary terms.

One reason for the use of many different approaches is that there are also different purposes and uses for valuation and CBA evidence. These include, for example, the following broad categories:

1. Specific project appraisal
2. Broader-scale policy appraisal and impact assessment
3. Monitoring and review of decisions
4. Demonstrating 'Value for Money'
5. Seeking funding
6. Prioritisation of investments
7. Planning and location decisions
8. Pricing decisions: fees, payments
9. Understanding, communication and advocacy
10. Curiosity-driven research

Each of these may call for different specific methods and coverage, and different requirements for accuracy and research expenditure commensurate with the context. This can also depend on the audience for the results – for example, the requirements under 'seeking funding' depend heavily on the organisation(s) from which funding is sought, and the funding criteria used.

One implication of this is that there is no 'perfect' format for a CBA. There are general observations that can be made, for example about high and low quality approaches to measuring certain values, the need to avoid double counting, the need for clear baselines and so on. But beyond that the detail of each CBA needs to be judged with reference to the decision context and scale of the exercise. Even a CBA that seems to have many omissions, from the strict perspective of the assessment criteria for CBAs covering the full impacts of projects and interventions, may nonetheless be a practical and useful study within the context of a specific problem or decision context.

So for example we have some studies that seek to inform political decisions at a national scale, such as Case 1: 'UK Marine Conservation Zones', informing the Impact Assessment for the UK Marine and Coastal Access Bill with an assessment of the benefits and costs of the whole proposed marine conservation zone network. At the other end of the scale, Case 2: 'Lyme Bay no dredging zone' focuses just on a single activity restriction for a small area off the south coast of the UK.

Case 3: 'Natura 2000 sites in Scotland' focuses on review of policy and demonstrating value for money at a national scale. As noted in Appendix 1, the results of this study may be useful at that scale, broadly justifying the conclusion 'Nature 2000 is money well spent', but give no insight into decisions relating to specific sites, management decisions or compensation payments.

Slightly differently, Case 10: ‘Old growth forests, Finland III’ seeks to explore optimal levels of conservation of a particular resource at a national scale. The results are useful in drawing broad policy conclusions about the scale of increased protection that would be optimal (about double the actual level) and the level of compensation payments that may be required to achieve this, but do not give site-specific detail.

Other studies aim at securing funding for particular objectives: for example Case 15: ‘European cork forest conservation’, from the European Cork Industry Association, essentially argues for public expenditures to protect cork forest ecosystems. Conservation NGOs are already pushing the case for cork forest conservation (see e.g. WWF, 2006), so the focus on revenues and employment may be interpreted partly as a pitch to a political agenda favouring ‘sustainable growth’ and competitive economies.

Case 5: ‘Uganda lowland forest protection’ is interesting in exploring the possible implications of setting higher entrance fees for forest parks, considering both the willingness of tourists to pay fees, and the potential effects on forest and bird conservation.

Case 26: ‘River Bouvade restoration, France’ and Case 27: ‘River Gardon downstream restoration, France’ are interesting in that both were carried out in the context of the EC Water Framework Directive, but in the first case the conclusion was to demonstrate disproportionate costs, while the second demonstrated high benefits, and losses associated with any delay in the restoration.

This list could be extended, but the general point is simply this: the purpose of CBA is highly variable, and this can have a strong influence on the specific form it takes, including the resources invested in it. The purpose is also one factor influencing the coverage of ecosystem goods and services within CBA, as discussed in the next section.

2.4.2 Coverage of services

There are many different approaches to assessing and valuing ecosystem services within an overall CBA framework, ranging from highly reductionist attempts to value each individual service (and flagging a subset that cannot be valued), to more holistic attempts to value composite environmental quality goods via stated preference or even revealed preference surveys. Partly this depends on the purpose of the assessment (for example, if funding criteria focus on financial factors, assessment may be restricted to marketed goods and services) and partly on the availability of data, resources and methods.

Some studies focus only on direct use values. This can be interesting because it can help to explain how management of an area is likely to evolve without funding or other interventions from outside the immediate user community, perhaps showing where external financing for conservation and compensation for opportunity costs would be required in order to achieve conservation goals (though demonstrating the desirability of such intervention would need valuation of the wider-scale benefits), and so on.

Case 15: 'European cork forest conservation' is an example of a partial CBA, looking at some costs and revenues, and noting but not assessing the many non-market benefits of cork forest conservation actions. In addition to the pitch at an economy-focused political agenda, noted above, this may also be a way of avoiding any criticism based on lower confidence in non-market value estimates. The results suggest that, even with this restricted set of goods and services under consideration, conservation expenditures seem justified.

This point about lower confidence in certain forms of value or valuation method, in particular for certain services, notably non-use values associated with conservation and biodiversity, is quite general. Relatively few studies seek to value the non-use aspects, and those that do generally treat the uncertainty, and the risk of double counting, with considerable care and conservatism. At one extreme, Case 1: 'UK Marine Conservation Zones' omits them, although they are estimated by the authors (McVittie and Moran 2008), but instead suggests they might be used instead of the detailed service value estimates, as a kind of alternative way of reaching the same conclusion. That is, rather than add them and risk the criticism that this entails double counting, or the criticism that the non-use values are not reliably estimated, they argue instead that the costs can be justified either by the direct ecosystem service values alone, or they can be justified by non-use alone. Strategically, this approach makes sense, but of course that only holds where a positive net present value does not depend on summing the use and non-use values to overcome the costs.

Case 29: 'Blackwater Estuary' and some others take a different approach that explicitly accounts for the double counting by using a composite environmental good, covering both non-use and use values associated with general environmental quality

2.4.3 Spatial scales

Benefits and costs arise at different spatial scales, and CBAs do not always cover all of them. Some for example look only at the local benefits and costs (and ignore impacts outside local boundaries); some compare the costs and benefits at different scales (see e.g. Case 6: 'Rajiv Gandhi National Park', Case 7: 'Dandeli Wildlife Sanctuary', Case 9: 'Old growth forests, Finland II', Case 28: 'Upper Tuul watershed, Mongolia').

Others draw a similar line, without strictly restricting consideration to a local area, by differentiating between global/non-use values (carbon, global biodiversity) and the local ecosystem goods and services: for example Case 4: 'Cardamom Mountains Wildlife Sanctuaries Project'; Case 12: 'Guyana forest protection'.

These studies are particularly interesting because they demonstrate the possible need for compensation or side-payments if the globally-optimal solution is to be obtained in a stable fashion. Where local or regional populations have incentives to exploit resources that, from a global perspective, should be conserved, some transfer from global funds to local actors is likely to be essential. This is particularly important in the context of greenhouse gas emissions and carbon values, and in particular REDD+, and an increasing number of studies are targeting this line of argument. Uncertainty regarding the global value of carbon is of course a key issue here – for example, Case 4: 'Cardamom Mountains Wildlife Sanctuaries Project' shows a positive Net Present Value, but only through assuming rather a large carbon value, that is arguably unrealistic.

More generally, however, the overall weight of evidence reviewed here suggests that there is considerable scope for cost-beneficial protection of forest resources on the basis of carbon-based, and potentially biodiversity-based, side payments. However Case 14: 'Hypothetical forest area, Cambodia' is important in this context by illustrating that while partial CBA of REDD+ can support schemes aiming at legal logging with lower carbon and other impacts than alternatives, full consideration (valuation) of ecosystem services and biodiversity values is required in order to encourage greater use of full protection in priority areas, and to ensure that REDD+ schemes take into account the full range of impacts, and do not blindly target carbon reductions at the expense of conservation and local ecosystem service interests.

2.4.4 'Bottom Line' results

With few exceptions, the studies analysed for this report find positive benefit-cost ratios for conservation decisions. In many cases the benefits far exceed costs; in others, benefits exceed costs despite the omission of important categories of benefit that are difficult to assess – notably, but not only, non-use values. By contrast, there are also cases in which the benefits exceed costs only because of the inclusion of non-use values. This may be due to selection bias, as noted above.

One set of partial exceptions to this is the group of studies, discussed above, that focus on local values, in most cases contrasting that with regional, national or global values. Here there are examples where the (opportunity) costs outweigh the benefits locally, though generally the benefits outweigh costs at the larger spatial scale, supporting the policy conclusion that to achieve globally-optimal conservation, local side-payments will be required.

There are also studies that explore the sensitivity of the general 'benefits exceed costs' conclusion to changed assumptions. Again, by and large, the beneficial nature of conservation appears to be robust. One important sensitivity however is to the discount rate, in particular where a conservation or restoration project requires up-front capital expenditure, with much longer-term benefits. Case 24: 'Skjern river, Denmark' is one such example, where strongly positive outcomes at lower discount rates turn to negative around a 7% discount rate.

Some studies do not seek to show a benefit-cost ratio for a specific project, but rather to determine where an optimal outcome lies – that is, the point at which the marginal BCR becomes 1. Case 10: 'Old growth forests, Finland III' is one example, finding that marginal costs and benefits of forest conservation would be balanced at approximately 2– 2.5% of private forests protected, at a conservation payment of around $6000 per ha. Total benefits (sum of consumer and producer surplus) at this level are $1.5bn to $3bn, depending on assumptions. Since about 1% of forestland is already protected by current environmental regulations, a doubling of currently protected areas would be justified based on these results.

Case 22: 'Agriculture-forest conversion in Wales' demonstrates a slightly different approach, looking at the choice of where agricultural land might be converted to forests. By mapping the different ecosystem service values within a GIS, this study allows prioritisation of the most cost-beneficial areas for conversion – the values vary from strongly positive to somewhat negative, depending in particular on the distance from population centres. This kind of approach is likely to become much more prevalent with increasing use of GIS, remote sensing data, and enhanced computing methods. It targets important spatial planning decisions at a strategic scale. The analysis has to be iterative, since the marginal values change as land conversion takes place, in particular since (in this case) the marginal value of forests for recreation is strongly dependent on the availability of other forests for recreation in the surrounding area.

One study with a particularly interesting twist on the 'bottom line' is Case 33: 'North Wind Weirs salmon habitat restoration'. Here, the specific local assessment of service values and opportunity costs suggests that development would be the better option; however the broader consideration of the vital role of this habitat in restoration efforts for an entire estuary and river system suggests that conservation is overwhelmingly beneficial. In some respects this leads to a relatively simple point about ensuring any CBA sets assessment boundaries (both spatial and temporal) such that the full effects of any decision can be taken into account. But there are also more nuanced points about the care required in representing critical natural capital in a cost-benefit framework, and also about the interdependence of projects and assessments – the critical value of this small area is realisable only if other important projects are also undertaken; it is one piece in a bigger picture.

2.5 Overall Conclusions

The broad picture suggests that there are few clear examples of 'near-perfect' CBA studies, but several examples that are 'good enough' in the context of providing a useful aid to decision making within a given context; and it must be stressed that CBA should not be seen as a replacement for deliberation, but rather as one part in it.

Within each type of valuation approach, there is highly variable quality and reliability, in terms of sample sizes, but also in terms of methodologies, assumptions and data analysis. In many cases, the nuances of valuation quality are hidden because the values derived from value transfer methods, or sometimes from 'off the shelf' standard unit values. There are, of course, examples of CBA attempts in which the assumptions or methods are rather inappropriate and this cast doubt on the usefulness of the specific study. It is important to recognise that the weaker CBA studies do not discredit the usefulness of the CBA method overall, but rather highlight the need for care and attention to detail in commissioning, carrying out, and interpreting cost-benefit work. Further, the conclusions herein are based on studies that have been screened for quality and reliability.

Almost all of the studies support a positive decision regarding the conservation of the study good, irrespective of whether that is forest conservation, or watershed protection, or river and floodplain restoration, and so on. However, this needs to be interpreted with care. The studies that have been carried out, and subsequently published, cannot be assumed to be representative of all possible conservation decisions. By and large, we would expect the most likely, most beneficial, least costly conservation actions to be taken first. So the finding could simply reflect the fact that these are the decision contexts that have been assessed.

Nevertheless, tentatively, we can suggest that the overall strength of the benefit-cost results, coupled to the fact that these strongly positive ratios are often derived on the basis of partial analysis, omitting changes in some key services and non-use values that are likely to be positive, tend to suggest that there may be ‘plenty more where that came from’. In other words, even accepting that these positive studies may relate to the most beneficial areas for conservation, it seems likely that the next most likely tranche will also be beneficial.

Strengthening this conclusion is the observation that, in many parts of the world, natural capital continues to be eroded, while populations rise, such that we can expect that the marginal value of conservation, and the marginal value of ecosystem services on which populations depend, are likely to rise. However, with rising populations to feed, clothe, house and employ, the opportunity costs of conservation will also rise. This may increasingly drive a wedge between local and global benefit-cost ratios for conservation, as some studies discussed above have already demonstrated. There is therefore a need to draw on the results of these studies as a justification for driving forward with policy measures aimed at securing globally optimal conservation outcomes by ensuring adequate compensation for local stakeholders, and further integrating these concerns with other policy areas such as climate policy, trade and development aid. This is in part the rationale for Part III of this Quantitative Assessment, i.e. the assessment of global scenarios and options.

PART II REGIONAL ASSESSMENT

3 Approaches to valuing ecosystem services at a regional level

There have been relatively few attempts to value changes in biodiversity and ecosystem services at a large scale (above the site level), due to limitations in the data and tools available. However the last decade has seen rapid advances in a number of areas, and several assessment models are now available and under further development. This chapter aims to providing an overview of some of the main models[19] that can be applied for large-scale assessments. It discusses some of the approaches that can be used for assessing and valuing changes in biodiversity and ecosystem services at a large geographical scale, with a focus on model-based approaches. Some of the main approaches are noted in Table 3, below.

The research has covered, in particular, global integrated assessment models such as the IMAGE-GLOBIO model suite, and some models for ecosystem services at the regional/watershed scale. The emphasis is on models that produce results in economic terms rather than purely biophysical models, though this distinction is not clear cut since most models aiming at monetary valuation first compute a biophysical measurement, and generally monetise only some services. Sector-based models have not been considered. The focus has been on Integrated Assessment (IA) models that seek to build a holistic analysis of global or regional systems.

The Society for Integrated Assessment defines IA as ‘the scientific ‘meta-discipline’ that integrates knowledge about a problem domain and makes it available for societal learning and decision making processes.’[20] Within this very broad definition, IA has primarily been developed for treating issues of large-scale, long-term environmental management. This approach can be useful for studying links between conservation of biodiversity and provision of ecosystem services, because the causal chains are complex, involving processes of hydrology, ecology, economics and so on.

There are a number of IA models available that can be used for assessing the impacts of environmental change on ecosystem services. These modelling frameworks have internally consistent and calibrated sub-models or modules, representing relevant components of the ecological-economic system. Examples include the Millennium Ecosystem Assessment (MA, 2005), The Global Biodiversity Outlook (2006), the Global Environment Outlook 4 (UNEP 2007), and so on. But none of the assessments, and none of the models, is ‘perfect’. In what follows, we do not attempt to make a comprehensive assessment of these models, but rather focus on some key features, and assess the usefulness of these approaches in the context of answering key questions about biodiversity and ecosystem service provision at the global to regional scales.

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[19] The ambition of this research was not to be comprehensive, but rather to assess which approaches and models can be considered to be ‘state-of-the-art’.

[20] http://www.tias.uni-osnabrueck.de/integrated_assessment.php (The Integrated Assessment Society)

Table 3 Summary of selected global and regional integrated assessment models and biodiversity models

Name	Description	Modelling	Economic valuation
GUMBO (Global Unified Metamodel of the Biosphere)	Global simulation meta-model for exploring possible future planetary scenarios.	234 state variables, 930 variables total, and 1715 parameters. Gumbo models five sectors or spheres: Atmosphere, Lithosphere, Hydrosphere, Biosphere, and Antrophosphere (human systems); also divided into 11 biomes. No spatial component. GUMBO is the first global model to include the dynamic feedbacks among human technology, economic production and welfare, and ecosystem goods and services within the dynamic earth system.	Calculates the marginal product of ecosystem services in both the production and welfare functions as estimates of the prices of each service. Boumans *et al*., 2002 Werners *et al*., 2004 Costanza et al., 2007a http://ecoinformatics.uvm.edu/projects/the-gumbo-model.html
IFs (International Futures simulator)	Large-scale, long-term, integrated global modeling system. Designed to facilitate exploration of global futures through alternative scenarios. Developed for educational purposes, increasingly used in policy analysis and international assessments, including GEO-4 and for assessing UN MDGs.	Represents demographic, economic, energy, agricultural, socio-political, and environmental subsystems for 183 countries interacting in the global system. Integrated with a large database containing values for its many foundational data series since 1960.	Not designed for valuation. Moderate degree of human-natural integration. Hughes & Hillebrand, 2006 http://www.ifs.du.edu/ Constanza, 2007
IMAGE (Integrated Model to Assess the Global Environment)	IMAGE 2.4 is an ecological-environmental meta-model. Aims to explore long-term dynamics of global change as result of interacting demographic, technological, economic, social, cultural and political aspects of human activity, to assess sustainability issues such climate change, biodiversity and human well-being.	*Socio-economic systems:* Demography World economy Agricultural economy and trade Energy supply and demand *Earth systems:* Atmosphere- Ocean System Atmospheric chemistry Managed land Natural vegetation.	Not designed for economic valuation. Moderate degree of human-natural integration. Widely use in global assessments. MNP, 2006 Publications on IMAGE applications are available at: http://www.pbl.nl/en/themasites/image/publications/index.html http://www.mnp.nl/en/themasites/image/index.html

Name	Description	Modelling	Economic valuation
GLOBIO (Global Methodology for Mapping Human Impacts on the Biosphere)	GLOBIO is a modelling framework to calculate the impact of five environmental drivers on land biodiversity for past, present and future (these are often derived from IMAGE)	GLOBIO is based on cause-effect relationships, derived from the literature. Models mean species abundance and habitat types. Used to explore impacts of environmental drivers on land biodiversity in past, present and future, relative importance of the environmental drivers, trends under future scenarios, effects of policy response options.	Not designed for economic valuation. Alkemade *et al*. (2009) http://www.globio.info/publications http://www.globio.info/ MNP, 2006
IMPACT – WATER (International Model for Policy Analysis of Agricultural Commodities and Trade)	IMPACT represents a competitive world agricultural market for 30 crop and livestock commodities, Specified as a set of 115 countries and regions within each of which supply, demand, and prices for agricultural commodities are determined. Linked through trade. IMPACT-WATER adds the WSM model incorporates water availability as a driving variable with observable flows and storage to examine the impact of water availability on food supply, demand and prices.	Income; Population growth (to determine food and non-agricultural water demand); Crop productivity; Change in available agricultural area over time, Climate parameters; Irrigation and water supply information; Trade policies.	Not designed for economic valuation, or for consideration of biodiversity. IMPACT-WATER is the only model that takes into account water availability for food production (other models assume that water for irrigation is available). Rosegrant *et al*., 2005 http://www.ifpri.org/themes/impact.htm
MIMES (Multiscale Integrated Model of Ecosystem Services)	Simulation models for spatial modelling of human-environment system at different scales. MIMES project aims to integrate participatory model building, data collection and valuation.	Suite of linked models for use at various scales – similar approach to GUMBO, but with spatial analysis added. Imports/exports of various kinds can included.	Joint economic and social valuation of ecosystem services. Marginal product of ecosystem services in both the model's production and welfare function. Can be expressed in monetary units or in physical flows and land areas. Boumans and Costanza, 2007 http://www.uvm.edu/giee/mimes2/
InVEST (integrated valuation of ecosystem services and tradeoffs)	Tookit for modelling and mapping the delivery, distribution, and economic value of ecosystem services.	Set of production function tools for modelling ecosystems at the regional/watershed scale. Comparison of scenarios for land-use/land-cover. Accounts for supply and demand aspects of services.	Economic value of ecosystem services can be added in many modules, value transfer added to a production function. Nelson *et al*. 2009 Nelson *et al*. 2008 http://www.naturalcapitalproject.org/InVEST.html

Name	Description	Modelling	Economic valuation
Ecopath with **Ecosim**, **Ecospace**, **EcoOcean**, **EcoVal**	Mass- or energy-balance modelling of ecosystems. Focus on fisheries management.	EwE has three main components: **Ecopath** – a static, mass-balanced model of an ecosystem; E**cosim**– a time dynamic simulation module for policy exploration; and **Ecospace**– a spatial and temporal dynamic module primarily designed for exploring impact and placement of protected areas.	Economic value of ecosystem goods and services under different management scenarios with EcoVal, a decision support component. Ecopath: Christensen & Pauly (1992) Ecosim: Walters *et al*. (1997) Ecosim II: Walters *et al*., (2000) Ecospace: Walters *et al*. (1999) EwE overview: Pauly *et al*., 2000 Christensen *et al*. (2000) Christensen *et al*., (2005) EcoOcean: Alder *et al*., (2007) http://www.ecopath.org/about
UK NEA (UK National Ecosystem Assessment)	The UK National Ecosystem Assessment (UK NEA) is the first analysis of the UK's natural environment in terms of the benefits it provides to society and continuing economic prosperity. Due to report in 2011.	Demographic, Economic, Socio-political, Technological and behavioural. Aim: to enable the identification and development of effective policy responses to ecosystem service degradation.	Detailed approach to marginal valuation across a range of ecosystems and services. Modelling of service production coupled with production function or value transfer methods. UK NEA (2010) http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx Bateman *et al* (2010)

3.1 GUMBO (Global Unified Metamodel of the Biosphere)

Global simulation models are best considered not as a means of predicting outcomes with any accuracy, but rather as a tool for exploring possible consequences under different scenarios and assumptions. Models at a very local scale, in contrast, can aim for more accurate predictions of expected outcomes under specific conditions. Regional assessments may fall somewhere between these extremes.

The main global model emphasising ecosystem services as a component of human welfare is GUMBO[21] (Boumans *et al*. 2002). This is a simulation model of the integrated earth system, aiming to assess the dynamics and values of ecosystem services. It is a metamodel combining simplified forms of several existing dynamic global models in both the natural and social sciences. The current version of the model contains 234 state variables, 930 variables total, and 1715 parameters. GUMBO itself is not spatially explicit, but MIMES (see below) seeks to add this. GUMBO is constructed as a stand-alone dynamic systems model, but the modelling makes use of relationships based on outputs of more complex and computationally demanding models. It is a compromise model aiming to be complex enough to include the production and interconnections among the major global ecosystem services, while at the same time remaining simple enough to be distributed and run on a desktop.

GUMBO is sub-divided into five 'spheres' (Atmosphere, Lithosphere, Hydrosphere, Biosphere, and Anthroposphere) and 11 biomes, covering the entire surface area of the planet. Among the spheres and biomes, there are exchanges of energy, carbon, nutrients, water and mineral matter. Ecosystem goods and services are represented by 10 aggregate categories for the output from natural capital. These combine with renewable and non-renewable fuels, built capital, human capital (knowledge and labour), and social capital to produce economic goods and services, and social welfare. GUMBO calculates the marginal product (i.e. value) of ecosystem services in the production and welfare functions. The model is calibrated with historical data on 14 key variables with observations over 1900 to 2000. The model is run with scenarios for a base case and four others reflecting different assumptions about the style of global government (globalised versus balkanised) and about the capacity of the planet and its resources (optimistic versus pessimistic). Model users can change the assumptions/parameters within the scenarios and observe the results.

The economic component of GUMBO draws together three groups of inputs – the production of ecosystem goods, the production of ecosystem services, and the economic production based on socio-economic stocks of social capital, knowledge, labour force and built capital. These feed in to the overall production of goods and services for satisfying human needs; waste is modelled as a negative feedback. The total production is divided into personal consumption, and savings rates for the main capital stocks, including natural capital. A key feature of GUMBO is modelling dynamic processes including feedbacks among human technology, economic production and welfare, and ecosystem goods and services. These linkages make it possible to estimate the costs and benefits associated with specific changes, by calculating the marginal product of ecosystem services in both the production and welfare functions.

[21] See also http://ecoinformatics.uvm.edu/projects/the-gumbo-model.htm and http://ecoinformatics.uvm.edu/GUMBO/GUMBO.ppt

GUMBO gives global projections for key aggregate variables, and results are at the broad strategic and advocacy levels. For example:

- All the scenarios show agricultural production, human population, and ecosystem service values, peaking before 2050 and then declining significantly (though the details differ across scenarios).
- The overall value of ecosystem services, in terms of their relative contribution to both the production and welfare functions, is shown to be significantly higher than GWP (4.5 times in the preliminary version of the model); this ratio also increases then falls over time.[22]
- 'Skeptical' investment policies are shown to have the best chance (given uncertainty about key parameters) of achieving high and sustainable welfare per capita. This means increased relative rates of investment in knowledge, social capital, and natural capital, and reduced investment in built capital and consumption.

Although GUMBO gives estimates of the global value of ecosystem services under different scenarios, it is a very broad-brush, strategic approach and offers no spatial detail. Other models discussed below add a spatial component, but of course this comes at a price in terms of model complexity and data requirements.

3.2 GLOBIO 3 and the IMAGE-GLOBIO suite

The GLOBIO3 model (Alkemand *et al*, 2009) has been developed to assess human-induced changes in biodiversity, in the past, present, and future at regional and global scales. The model is built on simple cause–effect relationships between environmental drivers and biodiversity impacts. Biodiversity is measured by the 'mean species abundance' (MSA), which is the mean abundance of original species relative to their abundance in undisturbed ecosystems.

There is no direct economic valuation component in the IMAGE-GLOBIO suite. The modelling gives the impact of changes in drivers on biodiversity and biomes, and these can then be linked to economic values through further analysis. In this current report (Part III) , this has been done through application of per hectare values for the different biomes, taken from the valuation literature.

Changes in drivers used in GLOBIO are often, but not necessarily, derived from the IMAGE 2.4 model (MNP, 2006), coupled with GTAP (Global Trade Analysis Project) for the economics side. This approach models several aspects of human impacts on the natural world. IMAGE is a global model using data at different resolutions depending on the category. The relationships modelled are broad estimates, calibrated against historical data, but knowledge gaps remain and indirect (second order) impacts are not fully represented.

The model is discussed in more detail in subsequent chapters of this report.

3.3 MIMES (Multi-scale Integrated Models of Ecosystem Services)

MIMES (Multi-scale Integrated Models of Ecosystem Services) is being developed by the University of Vermont's Gund Institute for Ecological Economics. MIMES extends the GUMBO approach to allow spatially explicit modelling, at various scales. The approach

[22] http://ecoinformatics.uvm.edu/GUMBO/GUMBO.ppt

aims to facilitate the understanding of the links between economic services and human welfare, determining the full value of ecosystem services, and explaining the way changes in the function and value of those services might occur under different management scenarios. The website[23] lists two key outcomes:

- A suite of dynamic ecological economic computer models designed to integrate understanding of ecosystem functioning, ecosystem services, and human well-being at a range of spatial scales; and
- Development and application of new valuation techniques adapted for ecosystem services and integrated with the modelling.

Values can be calculated and expressed in monetary units or in physical flows and land areas. The MIMES suite is open source and users can contribute their own sub-models, and select which sub-models they wish to used. The meta-model then requires an 'interaction matrix' to link outputs and inputs across component parts; the demands for input and supply of output for each sub-model then have to then link up across the meta-model.

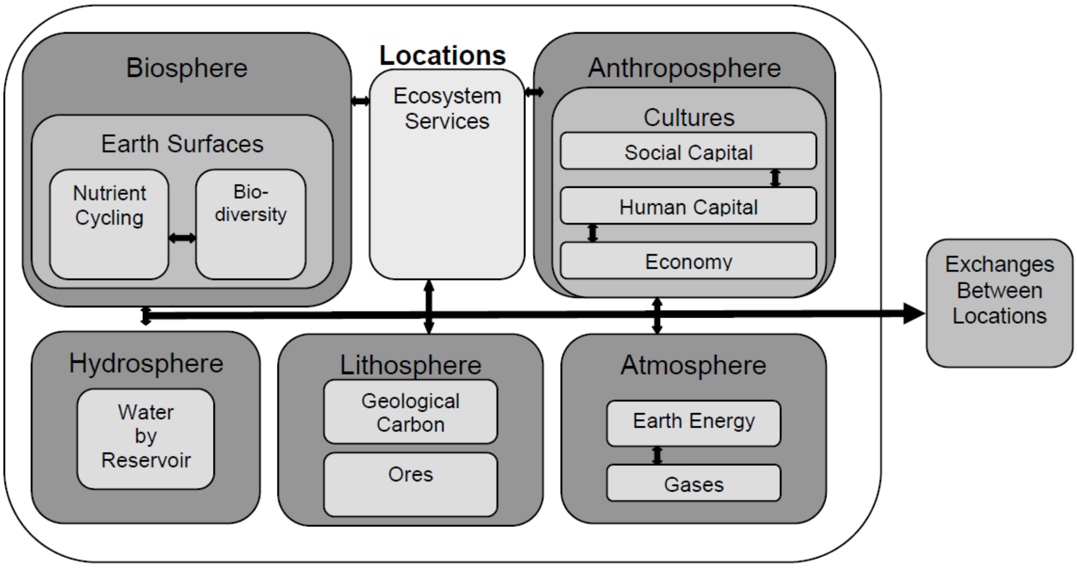

Figure 1: MIMES structure

(Source: Boumans and Costanza, 2007)

3.4 ARIES (ARtificial Intelligence for Ecosystem Services)

ARIES (ARtificial Intelligence for Ecosystem Services) is a web-based modelling platform for ecosystem services assessment, planning and valuation, under development by the University of Vermont, Conservation International, Earth Economics, and UNEP-WCMC (Villa *et al.*, 2009). It is built on University of Vermont's Ecosystem Services Database, which contains spatially-explicit, peer-reviewed valuation data as well as methods of analysis, publications and project models.

ARIES maps the potential provision of ecosystem services and their users using ecological process models or Bayesian models. In essence this means drawing on the best evidence available to make appropriate probabilistic assumptions about the generation of ecosystem

[23] http://www.uvm.edu/giee/mimes2/index.html

services, in circumstances where full data do not exist. Identifying and mapping beneficiaries has been a key step in development of the ARIES methods. The models are still under development.[24]

ARIES aims at economic valuation of services: at present this is based on simple benefit transfer from broad scale studies (linking land use-land cover types to per-area economic value derived from studies like Costanza *et al*. 1997) but longer term plans are to link to a database of primary valuation studies, enabling more sophisticated transfer approaches,[25] based on a recognition of the role of local demand and supply factors in determining value. The value transfer engine under development for ARIES uses the quantitative assessment of flows and estimates of critical thresholds to adjust values from source studies for transfer to an area under investigation.

3.5 InVEST / Natural Capital Project

The Natural Capital Project, a partnership of Stanford University, The Nature Conservancy and World Wildlife Fund, is developing suites of tools for modelling ecosystem service changes and associated values. A more detailed paper focusing on the Natural Capital Project has recently been prepared for DG ENV by Tinch and Mathieu (2010).

InVEST is essentially a set of production function models that relate biophysical and land use/ land cover (LULC) characteristics to production and delivery of ecosystem services. The input maps assign each cell a LULC type, which can be a natural (unmanaged) cover or a managed cover, at any level of classification detail. Results are given in relative or absolute biophysical terms (e.g. index of habitat quality, tons of carbon sequestered) and in some cases this can be extended to economic terms (annual flows or net present values). Modules currently available include: carbon sequestration and storage; pollination of crops; managed timber production; water supply (for reservoirs, hydropower); water purification (nutrient retention); and sediment retention (for reservoir maintenance). A biodiversity model enables analysis of links and tradeoffs between biodiversity and ecosystem services. Other modules are at various stages of development but not yet available. There is now also a Marine InVEST project, developing spatially explicit, process based models for mapping and valuing services provided by coastal and ocean ecosystems (Ruckelshaus & Guerry, 2009). Production function models currently in development within Marine InVEST include: food from commercial fisheries; food from aquaculture; protection from coastal erosion and flooding; wave energy generation; and recreation.

The predictions of ecosystem service provision from InVEST models can be used to compare the values arising in different scenarios modelled. This approach can help to demonstrate the relationships between services, and to identify management options that optimise service provision across the range of services considered, and over time. But the InVEST models are not dynamic models of land use change – the future scenarios are optional, and are separately created by users / stakeholders. This means that climate change and other driving forces are not directly modelled but must be dealt with through developing scenarios that take into account the influence of these factors on land use/land cover, and by supplementary analysis. Much more detailed projections and valuations could be achieved via econometric modelling of land use change, rather than defining scenarios

[24] See http://www.ariesonline.org/modules.html (accessed 24 January 2011) for an up-to-date list of modules available and under development.

[25] http://www.ariesonline.org/approach.html

directly through stakeholder processes, as discussed further below. But of course the resource and data requirements are much greater and the policy objectives are different.

As discussed in Tinch and Mathieu (2010), there are few numerical results currently published or available from the Natural Capital Project, although a number of studies are underway. Detailed results are available for an application to the Willamette Basin, Oregon (Nelson *et al*., 2009), and for the State of Minnesota (Polasky *et al* 2010). Nelson *et al* (2008) take a different approach that does seek to *predict* patterns of land-use change, and the comparison between the two studies of this area illustrate the different levels of complexity in the approaches.

Both InVEST and ARIES map the provision and beneficiaries of multiple ecosystem services, and can estimate monetary values. The main difference is that InVEST determines ecosystem service provision and value via ecological and economic production functions, linking spatially explicit maps of habitat types to specific service outputs. ARIES assigns ecosystem service provision and value directly according to the habitat and management characteristics, with the ecosystem service provision and values drawn from other site-based studies.

The main advantage of production function modelling over methods based on transfer of value estimates is that the services are explicitly modelled for the area under assessment, rather than being inferred simply on the basis of land use/land cover. However, this is of course a more complex and involved modelling process. From the perspective of global or regional assessments, the spatial scale and the level of detail and modelling required for the production function approach limits the complexity of the production functions that can be estimated.

3.6 Ecopath with Ecosim, Ecospace, EcoOcean, and EcoVal

Ecopath with Ecosim (EwE)[26] has been developed by the Fisheries Centre at the University of British Columbia, designed in particular for fisheries management purposes (though in principle the modelling approach could be applied in terrestrial environments). Ecopath is based on building, parameterising and analysing mass-balance trophic models of ecosystems. The ecosystem is modelled based on 'boxes' or 'groups' that may be a group of (ecologically) related species, a single species, or a single size/age group of a given species. Ecopath does not assume a steady state but rather estimates parameters based on the assumption of mass balance over an arbitrary period, usually a year. Ecopath provides an 'instantaneous' estimate of biomasses, trophic flows and instantaneous mortality rates for that period. Boxes need not be at equilibrium for the reference year, provided the modeller can provide an estimate of the rate of accumulation or depletion for each box for the reference period. The model requires the energy input and output of all living groups to be balanced, based on equations for each group.

Ecopath results in a description of the average state of an ecosystem, that can then be used to parameterise systems of coupled difference and differential equations to model changes in biomasses and trophic interactions in time (Ecosim) and space (Ecospace), though different Ecopath models may be needed for systems that have undergone major change

[26] www.ecopath.org

Ecosim is a complex simulation model for evaluating the impact of different fishing regimes on the biological components of ecosystems. Results of simulations can be used to modify the parameters iteratively until convergent results are achieved. Ecospace is a dynamic, spatial module incorporating key elements of Ecosim simulations. EcoOcean is a further development, used to explore the GEO-4 scenarios, based on marine ecosystem models for each of the 19 FAO marine areas. The model simulates changes in Ecopath model systems based on varying fishing effort levels for different fleet types. Ecoval is a decision-support method for comparing the ecological, economic and social benefits of different states of a marine ecosystem.

The EwE approach is data-hungry, but has been widely and successfully applied. There are however problems with the reliability and availability of input data and the results do depend on expert judgement. The model focuses on fisheries management and further extensions may be possible to enhance the representation of marine ecosystem services other than food provision, including but not limited to biodiversity.

3.7 UK National Ecosystem Assessment

The UK NEA (see Bateman *et al*, 2010) is based on 'linking the drivers of change (direct and indirect) to changes in biodiversity and ecosystem goods and services, and human wellbeing'. The valuation component takes a bottom-up approach to valuing ecosystems and their services. The objective is to 'consider the web of natural world relations underpinning each ecosystem service and acknowledge the role which human and manufactured capital play in combining with such services in the production of welfare bearing goods' ; this differs from most previous approaches to ecosystem service valuation by paying explicit attention to the inputs of human and manufactured capital, which mean that not all of the final value of the services can be attributed to ecosystems alone. The NEA takes a marginal approach to valuation, assessing scenarios against a 'business as usual' baseline, taking into account available knowledge on environmental change, the likely path of market forces, and ongoing and planned policy initiatives.

3.8 Mapping service values and biodiversity

The relationships between biodiversity, nature conservation and various ecosystem services are of great policy relevance, but are poorly understood, in part because it can be difficult to distinguish between correlation and causation. Various approaches have been taken to modelling these relationships, including several global / large-scale mapping analyses.

Costanza *et al* (2007b) use multiple regression analysis at the site and ecoregion scales to estimate relationships between biodiversity (plant species richness as a proxy) and net primary productivity (NPP) (as a proxy for ecosystem services). At the site scale, 57% of the variation in NPP was correlated with variation in biodiversity after effects of temperature and precipitation were accounted for. At the ecoregion scale, biodiversity was positively correlated with NPP only for the higher temperature areas, accounting for approximately 26% of the variation in NPP after effects of temperature and precipitation were accounted for. These results back up the idea of positive links between biodiversity and ecosystem functioning from experiments. Danovaro *et al* (2008) present results from a global study of 116 deep–sea sites, showing that several indicators of deep-sea ecosystem functioning increase exponentially with higher measures of biodiversity.

Turner *et al* (2007) set out to assess the extent to which biodiversity conservation priorities and ecosystem service values match in space, with a view to examining the global potential for simultaneously safeguarding biodiversity and ecosystem services. They used a global map of ecosystem service values (from Sutton and Costanza 2002, based on a standard land-cover map and unit value estimates from Costanza *et al*. 1997) combined with published biodiversity conservation priority maps to assess the concordance. Nine different biodiversity priority 'templates' were assessed, by comparing the total observed ecosystem service values from within the template, with the service values derived by randomly sampling (without replacement) sets of cells totalling the same area. This method leads to generally positive concordance of biodiversity priorities with areas of high ecosystem service value, with eight of the nine templates demonstrating significantly higher service values than random areas. The templates' service values exceed that of comparable random areas by an average of 71.6 percent.

This global approach assumes constant marginal values of ecosystem services within biomes (it does not account for within-biome variation) and relies on values based on simple value transfer from the Costanza *et al*. (1997) study, based on land cover, not modelling of actual service flows.

The results of such assessments are promising but further research is needed to develop ways of quantifying the value of these impacts on function. Costanza *et al* (2007b) tentatively conclude that a 1% change in biodiversity in the high temperature range (which includes most of the world's biodiversity) corresponds to approximately a 0.5% change in the value of ecosystem services.

There is also contradicting evidence. Naidoo *et al* (2008) take a more detailed approach, though also at a global scale, considering four ecosystem services for which spatial proxies could be developed at a resolution of 0.5° or finer (Carbon sequestration; Carbon storage; Grassland production of livestock; Water provision)

Even for this small subset of important ecosystem services, no global spatial data were available on economic values, preventing an attempt to convert the ecosystem service estimates to monetary values. Mean per-unit-area ecosystem service production was calculated for each ecoregion. Analysis showed little correspondence among services: no pair of services had a correlation coefficient >0.2. Naidoo *et al* compared two strategies: sequentially selecting areas to maximize total ecosystem service provision, and sequentially selecting areas to maximise species protection. On average for all taxa, for levels up to 90% of species representation, optimizing for individual ecosystem services conserved only 22–35% as many species for a given area as did optimizing for species. This is no more than were conserved by selecting ecoregions at random. Conversely, maximizing species representation for a given area captured only 17–53% of maximum ecosystem service provision, which is no greater than those from a random selection of ecoregions. These results varied with the service considered and the area examined; as such, this limits the comparison made.

Overall these results suggest that biodiversity and ecosystem services are not strongly correlated, on average. However looking at the spatial detail, Nelson *et al* identified win–win areas important for both types of targets, and areas of trade-off. Naidoo *et al* further argue that, although global analysis can inform broad-scale priorities, actual conservation investments are usually made at much smaller scales. Their assessment of a California ecoregion, a 'win-win' area in the global analysis, actually had less than 25% of its planning units in the win-win category. The flip side to this is that there will also be win-win areas within the ecoregions not identified as win-win overall. Therefore finer scale analyses remain essential for targeting specific conservation action.

Konarska *et al*. (2002) also demonstrate the scale-dependence of mapping exercises via empirical analysis of ecosystem service values for the US using both 1-km and 30-m resolution land cover data. They found that the total estimated value of ecosystem services more than doubled by going from the coarser scale land use data to the finer scale, because smaller, more valuable ecosystems, in particular wetlands, were under-represented in the coarser-scale data.

Raudsepp-Hearnea *et al* (2010) extend analysis of these themes by identifying ecosystem service 'bundles' via analysing the spatial patterns of 12 ecosystem services in a mixed-use landscape consisting of 137 municipalities in Quebec, Canada. They identified six types of ecosystem service bundle, and showed how these could be linked to areas on the landscape characterized by distinct socio-ecological dynamics. The results reveal tradeoffs at the landscape scale between provisioning and almost all regulating and cultural ecosystem services, and they show that a greater diversity of ecosystem services is positively correlated with the provision of regulating ecosystem services.

3.9 Evaluation of modelling approaches

Naidoo *et al*. (2008) note that, in contrast to global estimates of *biodiversity*, the spatial estimation of global ecosystem service *values* remains quite crude. Global simulation models such as GUMBO take a highly aggregated approach, lacking spatial detail, to explore rather broad future scenarios for key parameters at a global level. These models are appropriate for very strategic, horizon-scanning purposes. Spatially explicit global simulation models such as GLOBIO are only feasible at quite a coarse spatial resolution, and are also appropriate for strategic assessments, while breaking down impacts at broad regional scales.

Moving from the global to regional and local/watershed scale models usually involves more detailed data on land use and land cover. Future scenarios may also involve spatial detail, and more complex or nuanced management options. These models can be very demanding in terms of data and expert inputs, and also often involve significant interaction with stakeholders, to inform scenarios and/or model parameterisation. Static scenario-based models, such as InVEST, seek to develop production functions for services at the landscape scale, and to evaluate different scenarios developed outside the model. More complex approaches seek to model the human decisions underpinning land-use and land-cover change over time, involving often complex feedbacks from environmental and policy conditions to actor decisions (e.g. Nelson *et al* 2008). Monetary valuation of ecosystem service outputs is sometimes attempted, though rarely for all services. Often, as in InVEST, monetary valuation is an 'optional extra' that can be added on to certain of the biophysical outputs.

Statistical mapping approaches are also used. Costanza *et al* (1997) used a bottom-up approach to extrapolate global values from local studies, and these estimates, though very controversial, are still widely used. Sutton and Costanza (2002) used them to build a map of ecosystem service values across the globe, linking a standard land-cover map with the ecosystem service value estimates. Turner *et al* (2007) linked this map with maps of biodiversity priority areas to demonstrate possible win-win scenarios for conservation and ecosystem service protection. At a more regional/local scale, the ARIES model seeks to link up detailed modelling of land use and land cover with value transfer methods, at present simple unit transfer, but more complex modelling of marginal values is envisaged for the future.

There is a distinction between *predicting* the future and *envisioning* possible future scenarios, although this can be blurred to the extent that modifying control variables in predictive models allows exploration of trade-offs among different possible futures, and scenario models need to be grounded in some idea of what feasible paths might be. But predictive models need to be dynamic while scenario-based approaches can be static, taking the distribution of state variables and demands for a particular period as inputs. The challenges in full predictive modelling are significant, in particular because of the need to model human behaviour, including the ways in which it depends on environmental conditions, and policy.

It is also important to consider where feedbacks from changes in biodiversity and ecosystem services impact on socio-economic development. Consideration needs to be given to situations where changes in ecosystem services feed back to the decisions that human actors and societies make. This is analogous to the 'dumb farmer' critique of simple models of agriculture under climate change that do not take into account changes in crop choices; adding the link from changing climate to changing farmer behaviour radically changes (and improves) the predictions of the models.

None of the approaches cited above is perfect. Nelson *et al* (2008) note that 'we are still at an early stage in the analysis of the provision of ecosystem services and biodiversity conservation from landscapes, and much remains to be learned.' But they note that, in their modelling, the greater model complexity and data did not greatly change model results or policy advice. Adding ever greater ecological detail to models may not be the most productive direction for research. If the models available are able to give a reasonable appreciation of the direction and approximate extent of change, then that is enough for most purposes, especially at large scales.

On the other hand, there are major data gaps in service coverage, in particular at the economic valuation level. Although some services are widely covered at many scales, including carbon sequestration and storage, water supply, and food and timber production, others such as pollination, disease control and most cultural services are much less addressed. Naidoo *et al* (2008) note that 'one of our most striking findings is simply how few ecosystem services we were able to include in our analyses' at global scale. Efforts to expand the range of services that can be estimated in models, and valued, should be a priority.

One of the problems with basing global conclusions on a micro-level approach to ecosystem service valuation has been that there is likely to be a selection bias in the sites chosen for study: research focuses on areas with more biodiversity and ecosystem services. This could be less of a problem for studies at the regional scale, if we adopt a strategic approach to ecosystem service studies at large scales. Selection bias may still arise, if the focus is on regions more dependent on ecosystem services, but generally the regions will cover a wide range of habitats, human settlements and activities. On the other hand, as noted above, the coarser resolution can result in the reverse problem, with areas of particular rarity and high ecosystem service value being under-represented in coarse data sets.

Nelson *et al* (2008) stress the importance of monitoring and evaluation of the consequences of past decisions, leading to more confidence in predictions and policy advice from landscape-level analyses. Cornell *et al* (2010) similarly stress the importance of learning from the past, in the context of the IHOPE[27] project: the website states that 'Recent recognition that current earth system changes are strongly associated with the changes in the coupled human-environment system make the integration of human history and earth system history an important step in understanding the factors leading to global change and in developing coping and adaptation strategies for the future.' Diamond (2005) identified the 12 most serious environmental problems facing past and future societies, that often have led to the collapse of historical societies and also stressed that the interplay of multiple factors is almost always more critical than any single factor. Building these synergistic features and discontinuous threshold effects into Integrated Assessment models, and into valuation (for example through valuing resilience), presents major challenges for the future.

Global level modelling suffers from the difficulty of representing complex relationships at this scale. Detail about the specific relationship between features of natural systems and specific services is lost. The regional scale or watershed scale reduce this problem to some degree, but full production function modelling of services across a region or watershed is a major undertaking, demanding considerable expertise and huge quantities of data. Models based on land use-land cover scenarios coupled with databases of values for different land types are relatively simple, demanding less data and expertise than models seeking to develop explicit local production functions for ecosystem services. But production function methods can be more sensitive to local conditions and to subtler changes in the health of ecosystems, and can be better suited to accounting for spatial interactions in values.

It is important therefore to bear in mind that the level of accuracy to be demanded of a modelling process needs to be commensurate with the scale and complexity of the situation and the decision context. While we might set the bar quite high for local appraisals, where a specific conservation or development decision depends on assessment of the costs and benefits of each option, global and regional models need to be seen more as strategic tools yielding broader assessments of general trends in values over wide areas and in some cases long timescales. Each type of model has its strengths and weaknesses, and role to play in addressing the many problems facing biodiversity policy and the management of ecosystem services at all physical and governance scales.

Further research in this field should lead to more complete suites of production function tools that are sufficiently general to be applied anywhere in the world, but sufficiently flexible to be tailored to local conditions (Nelson and Daily 2010).

Many of the models discussed above are still in development, and the evidence base remains thin. One priority for the future will be to develop and assess the growing evidence base from which we can learn (especially with monitoring and ex-post evaluations) and from which value or ecosystem service transfer might be applied.

[27] http://www.aimes.ucar.edu/ihope/

PART III GLOBAL ASSESSMENT

4 Introduction to the Global Assessment

We provide a brief synopsis of the methodology applied for the global assessment in Section 1 so as to present the arguments against such an analysis as well as contra-arguments vis-à-vis the approach that we apply. We do not repeat this discussion here but do set out the methodology in detail, the results and the discussion.

The overall aim of this segment of the report (Part III) is to provide policy-makers with a defensible indicative monetary estimate of the economic and environmental benefits of a suite of macro-level scenarios to 2030 or 2050 (depending on the scenario being considered) and comparing these, where possible, with the regulatory implementation costs.

A summary of the change scenarios under evaluation is provided in Table 4. The macro-level scenarios are not policy options *per se*. In fact it is difficult to define precise policy options to realise the scenario in some cases, for instance 'dietary change'. There is an associated issue in that it is not possible to assign any cost estimate in the absence of a definable policy.

The IMAGE-GLOBIO bio-physical analysis is restricted to terrestrial biomes. Specifically, our analysis of these eight change scenarios values marginal changes in three biomes: (1) temperate forests and woodland; (2) tropical forests; and (3) grasslands. In Section 10 we provide a complimentary analysis of results for wetlands, mangroves, coral reefs, and lakes and rivers**.** *This complimentary analysis does not value the change scenarios*; it provides value estimates for projections of policy inaction, i.e. business-as-usual (BAU).

The eight scenario options set out in Table 4 focus on terrestrial ecosystems; a final ninth scenario option pertaining to the restoration of marine fish stocks (with compensating expansions in aquaculture) is dealt with separately in Section 11. The scenarios were chosen as they have been discussed in recent assessments such as the Global Biodiversity Outlook[28].

The methodological pathway to developing benefit estimates is the same for each of the eight scenarios set out in Table 4 and defines the structure of the sub-sections within Part III, where each stage is discussed in detail:

(1) *The IMAGE-GLOBIO bio-physical analysis* (Section 5)

The IMAGE-GLOBIO bio-physical model estimates: (i) ecosystem extent, (ii) Mean Species Abundance (MSA)[29] and (iii) carbon storage and carbon sequestration for 0.5 degree by 0.5 degree (*circa* 50km by 50km) grid cells under the assumptions inputted. The application of the with-change scenario projection determines the changes in each of these three parameters. Thus there is a 2000 base year (the status quo), a business-as-usual (without-

[28] http://gbo3.cbd.int/
[29] MSA is a measure of 'habitat intactness' and is discussed in Section 5.

policy) 2030 or 2050 baseline projection, and a 2030 or 2050 with-change scenario projection (e.g. with-Protected Areas expansion).

(2) *Summary of change scenarios* (Section 6)

The change scenarios in PBL (2010) and summarised in Table 4 are set out in this section.

Table 4 Summary of options scenarios

Sector/Trend	Baseline	Change scenarios
1. Agricultural productivity	The current trend of slowing productivity growth persists at a level of 0.64% productivity growth per annum. Cumulative agricultural productivity increase is of 25.6% by 2050, measured in terms of yields.	Investment in Agricultural Knowledge Science and Technology (AKST) leads to 40% additional crop productivity and 20% additional livestock productivity compared to the baseline.
2. Reducing post-harvest losses in the food chain	No change to current practices in which large amounts of agricultural products are 'lost' in the supply chain (around 30% of total food supplies).	Supply chain losses (i.e. post-harvest losses) are reduced by half, to 15% of total food supplies.
3. Forest Management	Forests continue to be exploited through conventional logging practices.	Replacement of conventional selective logging practices by reduced impact logging (RIL) and an increase in forest plantations.
4. Protected Areas	Current system of protected areas in maintained, with 14.6% of terrestrial areas having protected status, albeit differences between eco-regions. No further policy interventions.	(1) Expansion of protected areas to 20% of each ecological region. (2) Expansion of protected areas to 50% of each ecological region. †
5. Reduced Deforestation	No incentives to promote further measures to promote a reduction in deforestation rates are assumed. Deforestation and forest degradation continue (as *per* 'Forest Management')	Protection of all dense forests (closed tree cover) from agricultural expansion.
6. Mitigating climate change	Biofuel developments are modest, and land needed for biomass fuels is of the order of 0.5 million square km in 2050.	1) GHG concentration limited to 450 ppm CO_2-equivalent by including an expansion in bio-energy. Bio-energy land requirement of 4 million square km by 2050. 2) GHG concentration limited to 450 ppm CO_2-equivalent without bio-energy expansion.
7. Global dietary patterns	Livestock production doubles as a consequence of population and increased per capita consumption, driven notably by increased consumption in developing countries.	1) Transition to 'Willett diet' with reduced meat consumption 2) Complete substitution of all meat by plant-based protein consumption.†
8. Global agricultural trade	No change in the structure of global agricultural trade regime. Tariffs and subsidies remain as barriers to international trade.	Tariff, non-tariff barriers and subsidies are gradually removed between 2010 and 2020 resulting in full liberalization of trade in agricultural commodities. †

† These are extreme options developed by PBL (2010), care should be taken when interpreting our subsequent analysis as they are likely to be non-marginal in nature.

(3) *Generation of a database of biome-level primary valuation studies* (Section 7)

A comprehensive literature review, search of extant benefits transfer databases and a structured process of biome-level expert input realise a database of valuation estimates that are screened under various selection criteria. Valuation data is entered into matrices of biomes/ecosystem services. The biomes included in our study are temperate forest and woodland, tropical forest, grassland, inland wetland, mangroves, coral reefs, and lakes and rivers. Although only the first three of these is used in the analysis of change scenarios, database development (Section 7) and value function estimation (Section 8) are described for all seven biomes. Only the first three are included in the analysis of change scenarios.

Value estimates need to be standardised in terms of monetary units per unit area of ecosystem per year: we use purchasing power parity (PPP)-adjusted US$ in 2007 prices per hectare per year. 2007 prices are used as this is the most complete recent year vis-à-vis data across all countries.

(4) *Estimation of biome-level value functions* (Section 8)

The database is used for the development of biome-level value functions. Each observation used for a particular biome has some value (in PPP-adjusted US$ in 2007 prices per hectare). A biome-level value function explains the variation in value estimates. The explanatory variables that capture site characteristics might include: general characteristics (e.g. site size, ecosystem services provided); context characteristics (e.g. abundance of the ecosystem in the region, accessibility); and socio-economic characteristics of beneficiaries (e.g. size of relevant population, income). The value functions do not include MSA directly as an explanatory variable of ecosystem service value but some include variables that represent the underlying determinants of MSA in the IMAGE-GLOBIO model (land use intensity, fragmentation, and site size).

Whether these (and other) variables are included in the biome-level value functions varies on a biome-by-biome basis depending on the relevance of each explanatory variable to each biome and on statistical significance in the meta-regression model. The spatial variables that were considered in the GIS analysis are also set out in this section.

(5) *Estimates of costs for change scenario* (Section 9)

The scenario options are not policies *per se*. In this section we attempt to map scenarios onto policy options, and then provide cost estimates for these policies when available. No primary research has been carried out for the cost estimation – only extant data and literature are presented.

(6) *Indicative results for inland wetland, mangroves, coral reefs, and lakes and rivers biomes* (Section 10)

This section provides indicative results for policy inaction for those biomes not modelled directly by IMAGE-GLOBIO. These results are *not* part of the assessment of the eight change scenarios set out above; they represent a separate adjunct analysis.

(7) *Analysis of marine change scenario* (Section 11)

The analysis for the marine biome is set out and results presented. Again, this is an adjunct analysis that is *not* used as part of the assessment of the change scenarios, owing to the fact that the IMAGE-GLOBIO bio-physical analysis does not include the marine biome.

(8) *Results: Analysis of each change scenario* (Section 12)

There are two elements to the estimation of benefits: (1) valuing overall changes in land-use, and (2) valuing changes in carbon storage. With regards to land-use change, for each landscape patch (i.e. site) the value per hectare under the baselines and alternative scenarios are calculated. This is done by substituting in the site-specific variable values into the value function. The value of a change in a specific site is calculated by multiplying the average value (average across two scenarios) for that site by the change in area at the site. The values for changes in each ecosystem site are then aggregated to regional and global level to give the annual benefit value at these respective scales to determine the benefits of the change scenario vis-à-vis land-use change. Note that in some cases land-use change can have net costs if there is a switch from higher-value to lower-value land-use types.

The integrated assessment module of IMAGE-GLOBIO also estimates changes in carbon storage arising from some of the eight change scenarios relative to the baseline. These are juxtaposed with the value changes associated with shifts in land-use. These two categories of benefits are then set against any cost estimates derived in Section 9. We set out the benefit-cost ratio where applicable and apply sensitivity analysis to determine the reliability of this result. A schematic representation of the methodology used to value land-use change arising from the change scenarios is provided in Figure 2.

In Figure 2, there are two versions of the 'change in ecosystem extent' from the IMAGE-GLOBIO results: (1) the without-change 'baseline' scenario for 2050; and (2) with-change, 'option on', scenario for 2050[30]. It is this *change* that is valued, and we discuss in the Results section instances where this change might reasonably be considered as marginal and other instances where it is not considered marginal. This change arises over time from 2000 to 2030 or 2050, depending on the option scenario. So although there are estimates for total value in 2000 and 2030/2050 underpinning our analysis, we do not use (or reveal) these totals for the three biomes (temperate forests and woodlands; tropical forests; grasslands) i.e. we do not mimic the approach of Costanza *et al.* (1997) or indeed that of Braat and ten Brink (2008).

[30] Two change scenarios (REDD and protected areas) have a 30 year time horizon and so 2030 is used in these cases, not 2050.

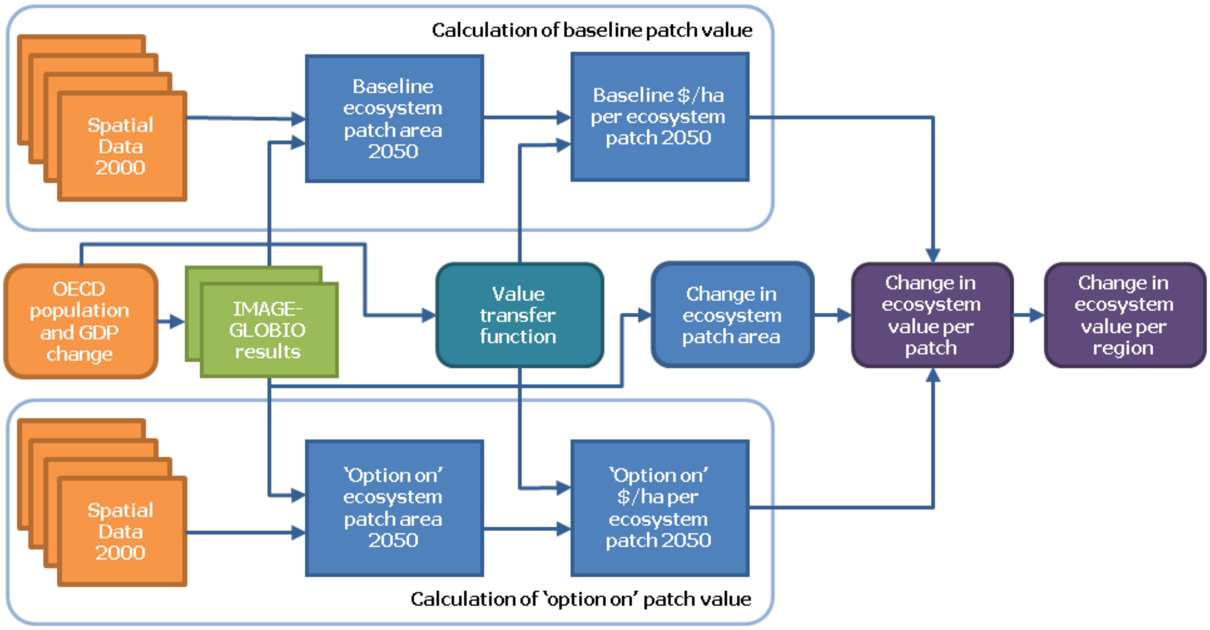

Figure 2 Schematic representation of value transfer methodology.

The methodology applied leads to an estimate of the change in 2030/2050. In the Results section we allow for the fact that this change is the accumulation of year-on-year changes across the entire study period (2000 to 2030 or 2050) by apportioning this change across the entire period. Our method for doing so is to assume a linear trajectory; this is discussed further in Section 12.

An overall discussion and summary of future research needs is set out in Section 13.

5 Bio-physical modelling: IMAGE-GLOBIO background and basis of the results

5.1 Outline of the IMAGE-GLOBIO model

The GLOBIO3 model is part of a modelling framework developed from existing approaches (IMAGE-NCI[31] and GLOBIO2) to evaluate the 2002 targets set by the Convention on Biological Diversity (CBD) and World Summit on Sustainable Development (WSSD), primarily the, 'significant reduction in the current rate of biodiversity loss at the global, regional, and national level; as a contribution to poverty alleviation; and to the benefit of all life on Earth' (Alkemade *et al.*, 2009). The model uses a number of cause-effect relationships to link environmental drivers to biodiversity impacts. The two primary outputs from IMAGE-GLOBIO are mean species abundance (MSA) and ecosystem extent, i.e. remaining natural area of a biome and remaining natural area of high quality (PBL, 2010).

MSA as estimated by IMAGE-GLOBIO is a composite indicator that indexes the abundance of original species remaining in disturbed ecosystem patches relative to the abundance in a pristine, undisturbed state. MSA is calculated for five drivers:

1. land use;
2. nitrogen deposition;
3. infrastructure;
4. fragmentation; and
5. climate change.

In the IMAGE-GLOBIO model these five drivers are assumed to affect the abundance of original species remaining relative to the pristine, undisturbed state, i.e. MSA. The extent to which MSA is affected varies across the five drivers. The overall MSA is obtained by combining the five elements (MSA for the land use type, MSA for the level of nitrogen deposition etc.) so as to derive a single MSA figure. Changes in these drivers are derived from the IMAGE model (MNP, 2006)[32].

Based on consultation and feedback from the authors of PBL (2010), we do not in this report discuss the cause-effect relationships which determine MSA. The full PBL (2010) report is available at: http://www.rivm.nl/bibliotheek/rapporten/500197001.pdf. We do not use the MSA measure *per se* in our analysis. As such we focus below on the aspects of the IMAGE-GLOBIO bio-physical modelling that impinge on our results directly.

5.2 Input data in IMAGE-GLOBIO

Data for land cover and land-use changes in IMAGE-GLOBIO come from the IMAGE model at a resolution of 0.5 by 0.5º grid cells. The spatial detail is increased by calculating the proportion of each land cover type within each grid cell from the Global Land Cover 2000 (GLC2000)[33] map (Alkemade *et al.*, 2009); GLC2000 data is at a resolution of 1 by 1km. Figure 3 illustrates how 21 of the 22 GLC2000 land classes[34] are mapped onto the IMAGE-

[31] Natural Capital Index (NCI) module of the Integrated Model for Assessment of the Global Environment (IMAGE).
[32] http://www.pbl.nl/en/themasites/image/index.html
[33] http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php
[34] GLOBIO does not include water bodies

GLOBIO land-use categories and MSA values. The 10 GLC2000 forest classes are converted into 4 land use categories using national data on forest use[35] with fractions assigned on a regional basis.

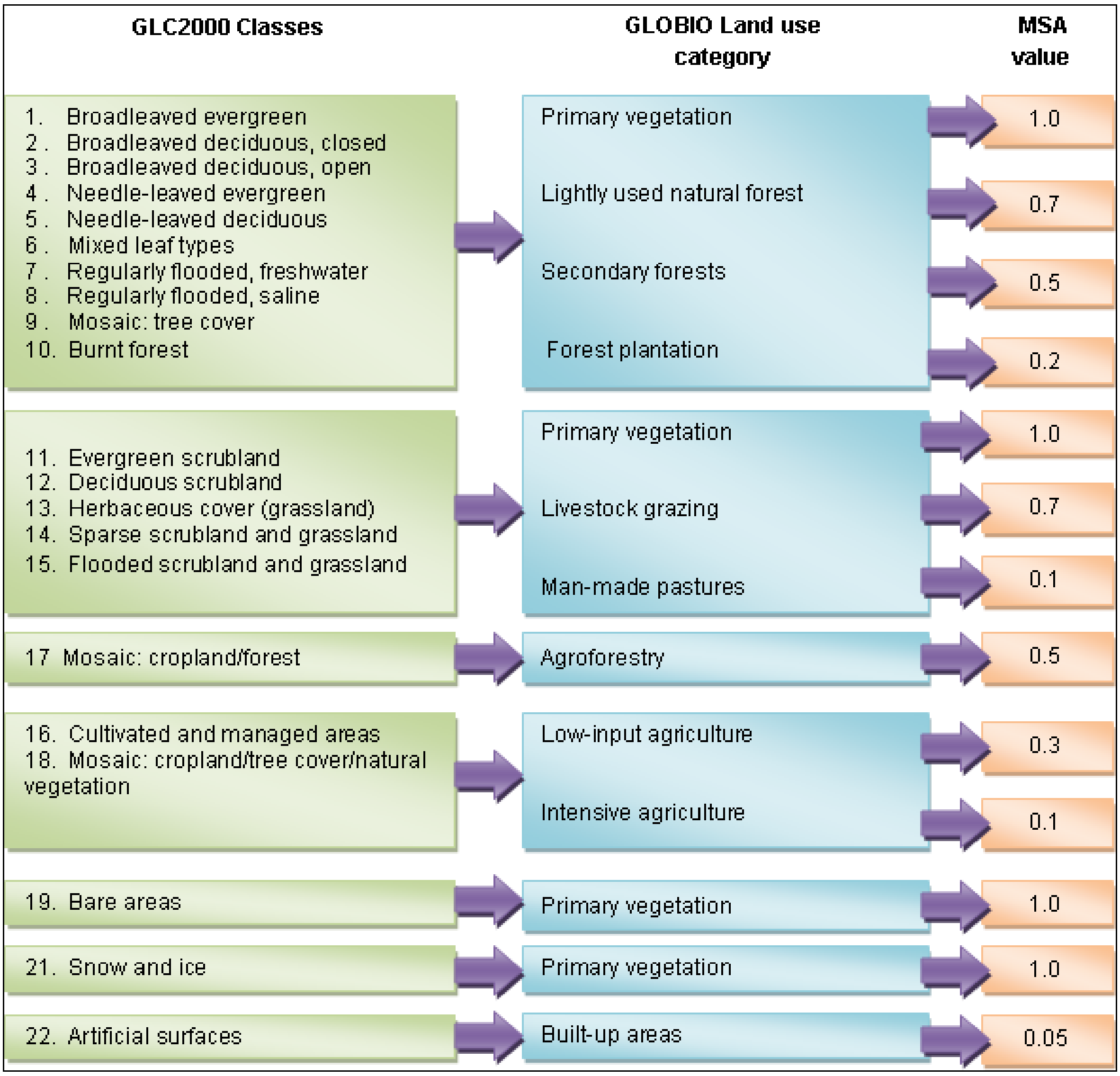

Figure 3 Conversion of GLC2000 classes to IMAGE-GLOBIO land use categories
Source: Adapted from Alkemade *et al.* (2009)

The five scrubland and grassland classes are converted into three IMAGE-GLOBIO categories. Livestock grazing area is based on estimates from IMAGE, and herbaceous areas are assigned to 'pasture' if those areas were originally forest in an IMAGE-generated potential vegetation map.

The cultivated and managed areas class is categorised as either low-input or intensive agriculture based on regional distributions of intensity[36]; where no estimates of distribution

[35] FAO (2001) Global forest resources assessment 2000. Main report. FAO Forestry Paper 140, Rome: FAO ftp://ftp.fao.org/docrep/fao/003/Y1997E/FRA%202000%20Main%20report.pdf

are available intensive agriculture is assumed. The GLC2000 class of mosaic of cropland and tree cover is treated as a 50/50 mix of low-input agriculture and lightly used forest.

IMAGE is also used to calculate N deposition, based on agricultural and livestock production, and global mean temperature change (Alkemade *et al.*, 2009). Infrastructure data is derived from a GIS map of linear infrastructure (roads, railways, power lines and pipelines) derived from the Digital Chart of the World database[37]. Buffers representing low, medium and high impact zones were calculated for each biome (Alkemade *et al.*, 2009).

Land-use change within the IMAGE model is derived from an extended version of the GTAP agriculture and trade model (PBL, 2010). The outputs from GTAP include sectoral production growth rates, land-use and the degree of intensification. Exogenous trends in crop yields (due to technology, science and knowledge transfer) are adjusted through a process of iteration between IMAGE and GTAP in which the effects of climate change and land conversion are calculated in IMAGE (PBL, 2010).

5.3 Model base year and baseline scenario

The baseline for the IMAGE-GLOBIO projections is based on the OECD 'Environmental Outlook to 2030' report[38] (OECD, 2008), and runs from a base year of 2000 to 2050. The IMAGE-GLOBIO modelling exercise uses a number of specific baselines for different policy options. The main characteristics of the OECD baseline are:

- Population growth from 6 to 9 billion following the UN medium scenario.
- Per capita incomes increase in all regions, particularly in dynamic emerging economies such as the Brazil, Russia, India and China (BRIC) countries.
- Global economic output increases fourfold (approx 2.8% per annum) with attendant shifts in consumption patterns including increased luxury foodstuffs and livestock products.
- Technical progress and the productivity of labour converge on long-term industrialised nation trend.
- Agricultural productivity increases by an average 1.8% per annum. This is insufficient to keep pace with either increasing population or changing consumption patterns necessitating an increase in land under production.
- Global energy use increases from 400 EJ to 900 EJ primarily from fossil fuel sources. Global average temperature increases 1.6ºC above pre-industrial levels.
- No new policies are introduced on the environmental and global trade.
- No new measures taken to promote bio-fuel use or to reduce CO_2 emissions from deforestation and forest degradation.
- No incentive to promote sustainable forestry. Demand for timber, pulp and firewood increase with economic and population growth.
- The size of protected areas will remain constant (approx 14% of terrestrial area).

[36] Dixon J, Gulliver A, Gibbon D (2001) Farming systems and poverty: Improving farmers' livelihoods in a changing world. FAO and World Bank, Rome and Washington DC ftp://ftp.fao.org/docrep/fao/003/y1860e/y1860e00.pdf

[37] http://www.maproom.psu.edu/dcw/

[38] http://www.oecd.org/document/20/0,3343,en_2649_34283_39676628_1_1_1_37465,00.html

Further details with regards the underlying assumptions are provided in PBL (2010); the authors note that regional differences will occur with regards the effects of these drivers.

5.4 Baseline scenarios: a synopsis of IMAGE-GLOBIO results

The results of the baseline scenario on biodiversity indicators modelled in IMAGE-GLOBIO suggest that, for the period 2000 to 2050, globally:

- The extent of natural areas will decline by 8% (10 million km2). Natural areas are those not used for urban construction, agriculture or infrastructure.
- Biodiversity, as measured by MSA, declines from 71% in 2000 to 62% in 2050. PBL (2010) note it is unlikely that MSA globally drops below 30-35% as 20% of the terrestrial land area is inaccessible; land converted to agriculture has a lower MSA limit of 5%.
- Wilderness areas (MSA > 80%) will decline by over 11% (15 million km2).

Within these global projections, some regional variations occur with MSA forecasted to drop below 60% in South Asia, China and the OECD countries due to higher economic and population growth and higher proportion of usable land taken into production or development (PBL, 2010). The highest rates in the decline of MSA are expected in South Asia and Sub-Saharan Africa where wilderness area will decline from 30 to 12% and 55 to 33% respectively (PBL, 2010). The changes in land cover are used as an input to the valuation of the eight change scenarios (discussed in Section 6) and so these results are presented in more detail in the Results section (Section 12) of this report. Further details pertaining to MSA change are provided in PBL (2010).

5.5 Use of IMAGE-GLOBIO output in the Quantitative Assessment

The output from IMAGE-GLOBIO most relevant to this study is the estimates of changes in the extent of different biomes. These changes are more readily applied to the (per hectare) values obtainable (directly or indirectly) from the valuation literature.

The assessment in our study has made best use of the available data and models at the global scale. In this respect it is a major challenge to consistently use the available data throughout the entire study given the different requirements on the data during the assessment process (Verburg, 2011).

In the IMAGE-GLOBIO framework, the quantitative modelling of scenarios of future land change is based on macro-economic models that calculate sectoral changes in commodity supply and demand. These models are based on aggregated production statistics linked to agricultural areas based on statistical data representing harvested areas. Changes in agricultural area reported under the scenario conditions will affect the extent of the other ecosystems: expansion of agricultural area occurs at the expense of semi-natural lands.

In the assessment of the extent of forest, grassland and other ecosystem areas for the valuation presented in Section 12, use is made of a combination of the best available global scale data to represent the spatial extent of the ecosystems. The GLC2000 data are used as a base dataset in this assessment. However, auxiliary data are used to also distinguish wetlands, mangroves and coral reefs that are not distinguished in the GLC2000 legend.

When comparing the agricultural areas in the statistical data as used within the IMAGE-GLOBIO assessment (PBL, 2010) and the land cover data as used in the spatial analysis of extent for our study, some deviations become apparent. These are explained by differences in classification and scale of analysis. Land cover data for agricultural areas also include fallow lands, small landscape elements, farm houses *etc*. In most cases the area covered by agriculturally dominated landscapes is larger than the actual harvested area as used within the scenario calculations in the economic modelling framework set out in Section 12.

In our analysis we have chosen to transfer the relative changes (2000 to 2030/2050) from IMAGE-GLOBIO to our valuation estimation. This way we can account for the differences in data representation between the data sources as we are not using changes in absolute areas. In terms of valuation this leads to, in most cases, a conservative estimate (i.e. an *under-estimation* of the benefits of applying the option-scenario) as the agricultural area is smaller in the statistical data (i.e. as applied in PBL, 2010) as compared to the land cover data (applied herein). Table 5 presents the areas of the three terrestrial biomes for the seven regions used in our analysis.

Table 5 Baseline areas of terrestrial biomes by region for current study and IMAGE-GLOBIO analysis.

	Grassland		Tropical Forest		Temperate forest	
	TEEB QA	IMAGE-GLOBIO	TEEB QA	IMAGE-GLOBIO	TEEB QA	IMAGE-GLOBIO
OECD	1,419.7	1,557.4	87.3	19.1	966.3	1,287.6
Central and South America	425.5	667.3	696.4	677.2	78.9	238.7
Middle East and North Africa	146.4	136.2	1.0	0.0	7.2	4.5
Sub-Saharan Africa	769.2	964.9	640.1	275.6	29.7	462.4
Russia and Central Asia	595.3	719.4	0.0	0.0	835.7	932.0
South Asia	172.4	211.7	252.6	211.1	51.0	143.9
China region	398.4	453.3	8.5	2.4	194.1	218.2
Total	**3,926.9**	**4,710.3**	**1,685.9**	**1,185.4**	**2,162.8**	**3,287.3**

A final point with respect to the use of IMAGE-GLOBIO outputs in this study pertains to the scope of the IMAGE-GLOBIO model, *viz*. it only models terrestrial ecosystems with respect to the change scenarios presented in Table 4. It is for this reason that we provide a complementary analysis for coastal biomes and open oceans.

6 Summary of change scenarios

PBL (2010) outline eight change scenarios pertaining to terrestrial ecosystems and one associated with marine ecosystems. A first group of options aims to assess the impacts of transformations to current agricultural and food production practices:

1. *Agricultural productivity*: Closing the yield gap
2. *Reducing post-harvest losses in the food chain*:
3. *Fisheries*: Aquaculture fish replacing partly marine capture fisheries
4. *Forest Management*: Reduced impact logging replacing conventional logging, and increased plantation establishment.

A second group of options aims to explore the impacts of scenarios that are explicitly linked to general conservation policies:

5. *Protected Areas*: Expansion of protected areas to (i) 20% and (ii) 50% per Eco-region.

A third group of options addresses the impacts of climate change mitigation policies:

6. *Reducing Deforestation*: Effects of stopping conversion of forested areas for biodiversity and greenhouse gas emissions[39].
7. *Bio-energy*: Climate change mitigation through bio-energy (bio-energy) development

Further, one option illustrates possible effects of a consumption-based transformation by addressing meat consumption and livestock production:

8. *Global dietary patterns*: Adoption of a healthier diet ('Willett diet') characterised by lower meat and higher vegetable consumption. As an extreme option PBL (2010) also consider the complete substitution of meat by plant-based protein consumption.

Finally as part of their analysis of baselines PBL (2010) consider two scenarios of *Agricultural productivity stagnation* and *Global agricultural trade liberalisation*. We estimate results for these scenarios, with the former characterised as an extreme variant of the 'closing the yield gap' option above where there is no investment in agricultural knowledge, science and technology.

Underpinning this choice of change scenarios is the premise that halting both biodiversity loss and the loss in provision in ecosystem services (ESSs) is not merely a 'conservation' issue; economic development and biodiversity are inextricably linked and should be analysed as an entity (MA, 2005). Promoting conservation policies without providing credible alternatives aimed at tackling the causes of habitat destruction through land conversion would likely be doomed to fail (Goklany, 1998). The choice of policy options reflects the main drivers of biodiversity loss. They show the close relationships, synergies and trade-offs between different global objectives that will become more prevalent in the years to come.

[39] We refer to this change scenario in our later analysis as a variant of Reduced Emissions for Deforestation and forest Degradation (REDD). The change scenario as described by PBL (2010) is not a full REDD type policy as forest degradation can continue. However our analysis is restricted to the valuation of reduced deforestation as this is reflected in land cover changes.

Relations between these different objectives include synergies, tradeoffs and complementarities (Caparos and Jacquemont, 2003).

The need to consider development and conservation hand-in-hand becomes clear when considering these options which address the main drivers of biodiversity loss. For instance: (1) closing the yield gap requires a substantial investment in agricultural knowledge, science and technology, implying technology transfer to the developing world; (2) communities in developing countries depend on fish catch for their livelihoods and therefore this scenario includes the development of aquaculture to compensate for reduced catches from wild stocks.

It is also important to mention what is *not* explored in the selection of nine scenario options. Two levels of policy action can be identified: (1) focusing on altering the main indirect drivers (e.g. population growth and economic expansion); or (2) keeping indirect drivers constant, analyze potential impacts of policies aiming to address more direct impacts (e.g. land-use change; land use intensity; and management). Our study focuses on the latter category, with the exception being the option that considers dietary change. What our study does not consider is a radical transformation of BAU scenarios, such as halting economic expansion as per the 'de-growth' agenda (see for instance Latouche, 2006). Neither does it model the possible impacts of population control from a neo-Malthusian perspective (for example: Daly and Cobb, 1989; Daly, 1992). While the 'de-growth' perspective can be theoretically interesting to explore, it arguably is not on the current political agenda (Chasek *et al.*, 2006), in part because it tends to compete with other objectives, such as the economic development of the South. Thus, our analysis broadly assumes that the *current global economic and political frameworks remain intact*, although it is the case that a switch to no meat consumption or expanding protected areas to 50% of each eco-region would constitute a radical transformation. It is for this reason that the results for these options are presented with caveats applied.

We provide a synopsis for each change scenario in turn[40], but refer the reader to PBL (2010)[41] for further details. Option 3 on fisheries is dealt with separately in Section 11. This is because the IMAGE-GLOBIO modelling applies to terrestrial ecosystems; although there is an impact on terrestrial ecosystems arising from this option, *viz*. agricultural production for aquaculture feed, the bio-physical modelling of fishing effort and catch projections for this change scenario is carried out separately.

6.1 Agricultural productivity

The first option addresses the potential impacts of a transformation in agricultural production practices. Since the industrial revolution, agricultural productivity has increased more than ten-fold world-wide, primarily as a consequence of the intensification of Western agricultural production; intensification has also occurred in parts of the developing world, particularly

[40] The same ordering of change scenarios and shorthand notation (the italicised terms in 1-9, e.g. *agricultural productivity*) is applied throughout Part III of this report, but omitting *Fisheries* (option number 4).

[41] http://www.pbl.nl/en/publications/2010/Rethinking_Global_Biodiversity_Strategies.html

following the green revolution (Evenson and Gollin, 2003). Yet disparities exist globally between regions and there is evidence of the growth rate of agricultural productivity levelling off (van Vuuren *et al.*, 2009). Many propositions have been advanced for explaining this trend: reduced investment in agricultural R&D (Pardey *et al.*, 2006); a general decrease of policy focus on, and support of, agriculture (Bello, 2010; McIntyre *et al.*, 2009); and land degradation and desertification (Bai *et al.*, 2008) as a consequence of poor land management or over-intensification of agricultural practices (Steinfeld *et al.*, 2008; Vitousek *et al.*, 1997).

The baseline scenario for this change scenario, based on Rosegrant *et al.* (2009), Van Vuuren *et al.* (2009) and FAO (2006), assumes that the current levelling-off of agricultural productivity growth persists: productivity growth is projected to be 0.64% growth per year, or cumulated growth in productivity of 25.6% to 2050 relative to productivity in 2000. More specifically, productivity growth per year is assessed to be about 1% for cereals, 0.35% for soybeans, roots and tubers, 0.8% for fruits and vegetables, 0.74% for livestock and 0.29% for dairy.

Yield differentials can be due to a variety of factors such as soil and climatic conditions. From a point of view of bio-physical constraints, Van Ittersum *et al.* (2003) synthesize the factors determining attainable crop yields by separating (1) growth defining factors, (2) growth limiting factors and finally (3) growth reducing factors. A ‘yield gap’ refers to the difference between potentially attainable yields (given a variety of bio-physical conditions) compared to actual agricultural yields. As stated by Fisher *et al.* (2010): ‘Yield gaps exist because known technologies that can be applied at the local experiment station are not applied in farmers’ fields having the same natural resource endowment’ .

In the baseline change scenario, while the year-on-year increase in yield remains constant, the rate of growth in food demand is projected to outstrip yield growth as a consequence of (1) population and economic growth, and (2) increased demand for meat (livestock) - itself an outcome of dietary transformations brought about through economic growth in developing countries (FAO, 2008). These parallel developments are expected to put pressure on land conversion of natural areas; expansion is projected to be of the order of 10%, occurring mostly in the tropical and sub-tropical zones (OECD, 2006).

The following option is modelled in our study: globally, productivity growth is spurred by investment in Agricultural Knowledge, Science and Technology (AKST), increasing productivity growth by 40% and 20% for crop and livestock respectively, relative to the baseline.

6.2 Reducing post-harvest losses in the food chain

A second change scenario linked to agricultural production deals with agricultural supply chain losses. These are defined as a loss in any stage of the supply chain following harvesting and up to calorific intake. These losses are estimated to represent about 30% of total food production, while reaching virtually 50% in specific geographical regions, and for specific agricultural products (Stuart, 2009). On the one hand, a considerable part of supply chain losses can be attributed to socio-economic and institutional factors in developing countries, such as poor market development linking farms to the distribution sector, or a lack of (or inappropriate) storage methods and facilities (Kader, 2005; FAO, 2008). On the other hand, post-harvest losses are no less considerable in developed countries, being attributed to different factors such as distribution sector inefficiencies and consumers' practices, including disposal of out-of-date produce (Stuart, 2009).

The PBL (2010) baseline scenario assumes no change to current practices. The change scenario sets out a reduction of food supply chain wastes and losses by 15% of total supplies, which would roughly correspond to halving the estimated current losses; this is mimicked by adjusting price and income elasticity curves to reduce total food demand. No distinction is made between different types of losses (e.g. during harvest inefficiency, pre-marketing, or post-marketing storage) or type of foodstuff). The option is based on a current joint study between PBL, LEI and IFPRI.

In short, this option assesses the possible impacts of simple efficiency increases in the food chain in order to decrease the necessity of further land conversion, and thus pressure on remaining natural areas, but without necessitating an increase in agricultural production and productivity compared to baseline developments.

6.3 Forest management

Conventional logging practices in current forest exploitations are significant contributors to forest degradation and biodiversity loss through over-exploitation of forest resources (Siry *et al.*, 2005). In the baseline, forests continue to be exploited through conventional logging practices, as no further incentive schemes or polices are assumed. The effects of current practices are projected to be exacerbated given population and economic growth and the resultant increase in demand for timber products for raw materials and fuel.

Beyond forest exploitation trends and current practices, sustainable forest management (SFM) practices can potentially respond to the need to maintain timber extraction to meet consumption demands whilst conserving ecosystems and reducing greenhouse gas emissions (Kant *et al.*, 2004). Yet SFM is often argued to be an elusive concept (Pearce *et al.*, 1999 and 2003), encompassing ‘one hundred faces’ (Wang, 2004). It is therefore critical to determine specific policy options. To this end, two combined policies are assessed as the scenario option: (1) A transition from conventional logging (CL) (or selective logging) practices to reduced-impact logging (RIL) for the exploitation of remaining (semi-)natural forest stands; (2) an expansion of plantation forestry to substitute for reduced logging of semi-natural forest.

The rationale for this combined policy is as follows: a transition from CL to RIL will more than likely either induce a decrease in global timber products supply (Pearce *et al.*, 2003) or imply the exploitation, albeit through RIL, of larger forest surfaces. This neglects the possibility that through RIL, productivity may also increase as less damage to remaining forest biomass is attained. This in turn may lead to better re-growth. In order to counter possible adverse effects, the ambitious development of forest plantations contributes to meeting future demand; forest plantations are generally more productive than natural or semi-natural forest logging (Brown *et al.*, 2000).

6.4 Protected areas

This change scenario explores the effects of increasing protected area coverage to 20% and to 50% of 65 terrestrial ecological regions (Olsen *et al.*, 2001). The 20% target was developed from earlier work for the second Global Biodiversity Outlook (sCBD and PBL, 2007). The current protected area system is estimated to represent 14.6% of all terrestrial area (Coad *et al.*, 2009). But despite the (pre-Nagoya) 10% objective set by the CBD[42], figures vary across biomes and even more across eco-regions. According to Coad *et al.* (2009), while the 10% objective has been achieved for 11 out of 14 global biomes, only half of global eco-regions reach this protection level.

In the baseline, no further policy beyond the 10% objective of the CBD is assumed. The current system of protected areas representing 14.6% of terrestrial areas is maintained. The two scenario sub-options evaluated in this study are the expansion of protected areas to 20% of each ecological region and to 50% of each ecological region.

This results in an even representation of protected areas not only per biome but also per eco-region within each biome. While expansion of protected areas can severely limit potential agricultural expansion in the context of the baseline projection of rising food demand, land scarcity is also thought to provide incentives for spurring agricultural productivity (Lambin *et al.*, 2001). The model suite allows for changes in land use intensity when protected area expansion limits the area available for agriculture. Finally, this option assumes that further anthropogenic pressures on ecosystems such as nitrogen deposition and climate change impacts continue to exert their effects within protected areas, thus impacting on biodiversity.

6.5 Reduced deforestation

Approximately 20% of greenhouse gas emissions come from deforestation and forest degradation, whether directly or indirectly through land-use change; thus, reducing emissions from deforestation and forest degradation (REDD) appeals as a prominent way to reduce greenhouse gas emissions from anthropogenic origin, and/or increase natural carbon sinks of global forest areas (IPCC, 2007; Houghton, 2009). Beyond emissions abatement, REDD is also believed to present ‘co-benefits’ for biodiversity preservation since 40% to 50% of the global genetic pool is located in global forests, particularly in tropical forests (Karousakis, 2009; Kitayama, 2008).

The original REDD objectives have been replaced by a so-called ‘REDD-plus’ objective, which: (1) takes further into account co-benefits, (2) aims to develop schemes spurring participation of local communities and (3) extends international transfer mechanisms to participating developing countries (Angelsen, 2009).

We do not model REDD or REDD-plus *per se*, as PBL (2010) does not model degradation. However the analysis in Section 12 focuses on value changes derived from land-cover changes. As such our methodology would only partially address quality changes vis-à-vis degradation, i.e. if degradation were to have been modelled in the bio-physical analysis in PBL (2010) then our valuation results would not change markedly to those presented. As such we term this option a variant of REDD.

The baseline assumes no additional actions compared to current standards: in short, deforestation and forest degradation continue due to additional pressures of population and economic growth, with subsequent land-use change for agriculture and logging practices

[42] The change scenario obviously talks to the outcomes established at the Nagoya CBD COP (see http://www.cbd.int/nagoya/outcomes/) vis-à-vis protected areas but equally the analysis in our report is based on current protection *achieved* as opposed to protection level aspired to.

(OECD, 2006). The scenario option assumes the protection of *all* forests with closed tree cover[43] (i.e. 100%) from agricultural expansion.

The analysis of reduced deforestation through IMAGE-GLOBIO allows the assessment of potential tradeoffs between climate change mitigation objectives and biodiversity preservation (or the extent of ‘co-benefits’). On the one hand, protecting forest areas can reduce greenhouse gas emissions thus addressing a critical pressure on global biodiversity. On the other hand, protecting solely forest ecosystems leads to the possibility of agricultural expansion to other natural areas (through a ‘leakage’ phenomenon). The change scenario assumes that *all* forests (areas of closed tree cover excluding savannah, scrub and wooded tundra) are protected form conversion to agricultural land from 2010 onwards (PBL, 2010).

6.6 Mitigating climate change with bio-energy

Bio-energy development and expansion is considered as a prominent potential component of climate change mitigation strategies, in the main because of its potential to be used as a fuel for the transport sector. In the baseline scenario, no further action to reduce greenhouse gas emissions is taken beyond current standards and agreements. In the scenario option, climate change is limited to greenhouse gas concentrations of 450 parts per million of CO_2 equivalent in the atmosphere with ambitious bio-energy development: Bio-energy land requirements increase from 0.5 million km^2 by 2050 in the baseline to 4 million km^2 for the change scenario.

This option aims to make the possible synergies and tradeoffs between climate mitigation and biodiversity explicit; the indirect effects of bio-fuel policies include land-use change impacts (Stehfest *et al.*, 2010; Crutzen *et al.*, 2007) and their resulting effects on biodiversity levels (Gallagher, 2008; van Oorschot *et al*., 2010) and spillover effects in terms of greenhouse gas emissions (Crutzten *et al.*, 2007; OECD, 2008).

Variants of this change scenario with either no bio-energy (i.e. GHG mitigation is achieved through changes in the energy mix) and with bio-energy and more efficient agricultural production (i.e. simultaneous and rapid improvement in agricultural productivity) were assessed by PBL (2010). We do not consider those variants in our analysis.

6.7 Global dietary patterns

The idea of a dietary transformation through a reduction of meat consumption has been particularly in vogue with climate change mitigation policies (Gerber *et al.*, 2009; FAO, 2006). Beyond climate change mitigation concerns, dietary change has also be justified on biodiversity grounds (FAO, 2006). Overall, the environmental and economic concerns that drive the contemporary debate on the livestock sector are as follows: (1) the livestock sector requires a considerably larger amount of land than agriculture for a similar calorific intake; (2) in the context of food security in developing countries, a rise in grain and vegetable supplies decrease their market prices; and (3) a reduction in meat consumption is thought to be a valuable strategy for a reduction of greenhouse gas emissions from ruminants, land-use change and avoiding agricultural production intensification, including reductions in nitrogen inputs and resultant nitrous oxide emissions.

[43] Woodlands are not included because of lower tree cover; in PBL (2010) some tropical woodlands are lost while tropical dense forests are preserved resulting in an overall loss of the tropical forest biome extent.

Current projections, as assessed in the baseline, forecast an increase in global meat consumption, both as a consequence of economic development and a 'westernisation' of emerging countries' diets (Pingali, 2004). These effects will lead, according to baseline projections, to a global doubling of per capita meat consumption by 2050.

The following two sub-change scenarios are evaluated:

(1) A global transition to vegetarianism through the complete substitution of meat protein intake by plant-based protein consumption, i.e. the phasing out of the livestock sector worldwide. This implies a radical transformation of global agricultural production practices.

(2) a transition to a lower-meat consumption diet with more fruit, vegetable and whole grain consumption based on the 'Willett diet' (see PBL, 2010).

6.8 Global agricultural trade

A variation on the baseline considered by PBL (2010) is liberalisation of global agricultural trade. Trade liberalisation has been considered to be a spur for (1) efficient global agricultural production, and (2) for developing countries economic growth in the context of economic globalization (Anderson *et al.*, 2006). It is indeed often argued that trade liberalisation could be a critical factor for poverty reduction in the developing world (Watkins and Fowler, 2002; Oxfam, 2005) and spur global food security, which is particularly important for developing countries (Rosegrant, 2000). Under this developmental rationale, and considering that agriculture is currently by far the most protected sector globally, the World Trade Organization (WTO), as well as some emerging economies, has insisted on the progressive liberalisation in agricultural commodity markets during the Uruguay and Doha rounds; however, their efforts remain relatively unsuccessful (Daudin, 2003).

Despite the fact that advocates of agricultural trade liberalisation focus primarily on economic development, it also entails potential environmental impacts both in terms of greenhouse gas emissions and in terms of land-use change and its resultant effects on biodiversity - as a consequence of global production relocation (Verburg *et al.*, 2008). Hence, the interest in evaluating the impacts of liberalisation lies precisely in an aim to make explicit the potential tradeoffs and synergies between the competing goals of the international community.

The baseline scenario (OECD, 2006) assumes that the current structure of global agricultural trade persists, with no dismantling of current tariff and non-tariff barriers. The change scenario assumes these barriers are progressively dismantled by 2015 thus contributing to a global structural transformation of agricultural production[44]; under this change scenario no barriers to agricultural trade remain after 2015.

6.9 Summary on change scenarios

Table 4 above in Section 4 presents a summary of the options so as to provide a context for the Part III 'route-map' set out in Section 4. It is repeated below (Table 6).

[44] The effects of free (or indeed freer) trade on biodiversity and ecosystem services is much disputed and the analysis herein is partial, in part because the global scale bio-physical modelling cannot fully account for local pollution havens, displacement effects, scale effects *etc*.

Table 6 Summary of Policy Options focused on terrestrial ecosystems

Sector/trend	Baseline	Change scenarios
1. Agricultural productivity	The current trend of slowing productivity growth persists at a level of 0.64% productivity growth per annum. Cumulative agricultural productivity increase is of 25.6% by 2050, measured in terms of yields.	Investment in Agricultural Knowledge Science and Technology (AKST) leads to 40% additional crop productivity and 20% additional livestock productivity compared to the baseline.
2. Reducing post-harvest losses in the food chain	No change to current practices in which large amounts of agricultural products are 'lost' in the supply chain (around 30% of total food supplies).	Supply chain losses (i.e. post-harvest losses) are reduced by half, to 15% of total food supplies.
3. Forest Management	Forests continue to be exploited through conventional logging practices.	Replacement of conventional selective logging practices by reduced impact logging (RIL) and an increase in forest plantations.
4. Protected Areas	Current system of protected areas in maintained, with 14.6% of terrestrial areas having protected status, albeit differences between eco-regions. No further policy interventions.	(1) Expansion of protected areas to 20% of each ecological region. (2) Expansion of protected areas to 50% of each ecological region.
5. Reduced Deforestation	No incentives to promote further measures to promote a reduction in deforestation rates are assumed. Deforestation and forest degradation continue (as *per* 'Forest Management')	Protection of all dense forests (closed tree cover) from agricultural expansion.
6. Mitigating climate change	Bio-fuel developments are modest, and land needed for biomass fuels is of the order of 0.5 million square km in 2050.	1) GHG concentration limited to 450 ppm CO_2-equivalent by including an expansion in bio-energy. Bio-energy land requirement of 4 million square km by 2050. 2) GHG concentration limited to 450 ppm CO_2-equivalent without bio-energy expansion. †
7. Global dietary patterns	Livestock production doubles as a consequence of population and increased per capita consumption, driven notably by increased consumption in developing countries.	1) Transition to 'Willett diet' with reduced meat consumption 2) Complete substitution of all meat by plant-based protein consumption.
8. Global agricultural trade	No change in the structure of global agricultural trade regime. Tariffs and subsidies remain as barriers to international trade.	Tariff, non-tariff barriers and subsidies are gradually removed between 2010 and 2020 resulting in full liberalization of trade in agricultural commodities.

† This variant of the change scenario is not considered in our subsequent analysis

7 Databases of biome-level primary valuation studies

7.1 Introduction

In Section 7 we outline the data collection and development of biome-level primary valuation studies. We summarise the data sources used to develop the benefit databases for each biome and provide commentary on the respective valuation databases subsequently used in the value functions (Section 8).

7.2 Benefit database development

The valuation studies used for the QA benefit transfer were identified from existing databases held by IVM (for water related biomes) and the TEEB valuation database (de Groot *et al.*, 2010) developed at Wageningen (forest and grassland biomes). The TEEB database contains 1,298 individual entries across 14 biomes (see Table 7) with temperate and tropical forests accounting for 105 (8%) and 260 (20%) of values respectively. Woodlands studies account for 3% of studies in the database, and grasslands are just under 5% of studies. Several entries may arise from a single study as each entry represents the values for a specific ecosystem service.

Table 8 presents the distribution of studies across regions in the TEEB database[45].

The major task in database development was to undertake a thorough review of the biome values obtained from the TEEB database so as to determine the suitability of the values for their inclusion and to identify additional variables not contained in the TEEB database but that would be of use to the QA analysis. The site co-ordinates listed in the TEEB database were also checked prior to the calculation of site-specific spatial data for use in the QA value function estimation (Section 8).

Following completion of the review a number of studies in each biome were considered unsuitable for inclusion in the QA database. The primary reason for rejection was that the values contained in a study were themselves derived through benefit transfer; only primary valuation estimates are included. Benefit transfer commonly occurred where an existing study was used to provide values for specific ecosystem services (e.g. bio-prospecting) or where global or regional values were downscaled to a specific country or site. Other reasons for rejection include the value being for an entire country rather than an identifiable site, or there being insufficient information to identify the site size or the benefiting population. In some cases additional values were found, for example where the published paper aggregated a number of individual site values or where additional values were stated in the paper.

Some additional analysis was undertaken on the selected values - conversion of all values to the common unit of value, *viz*. 2007 US$/ha/annum. The data used for the currency conversions and deflations were obtained from the World Bank's World Development Indicators dataset (World Bank, 2010). These calculations involved first estimating the year of study value per ha per annum in local currency units (if reported in another currency such as US$ these were converted to local units using the appropriate purchasing power parity

[45] The TEEB database is to be made available on the web but the *url* location is not as yet available.

exchange rate). Values given in perpetuity or over a specific time period were converted into present value terms using the discount rates quoted in the study (if none was quoted an appropriate local discount rate was identified through an online search). If values were given in per-household terms then these would be aggregated using relevant local, regional or national household estimates[46] (studies were rejected if the relevant population over which to aggregate could not be identified). The aggregate values were then divided by site area. Finally, per ha values in local currency units were adjusted to 2007 values using appropriate national GDP deflators and then converted to US$ using the relevant purchasing power parity exchange rate[47].

Table 7 Frequency of studies per biome in the TEEB database.

Biome	Frequency	Percentage
Marine	26	2.0%
Coastal	65	5.0%
Inland Wetlands	272	21.0%
Fresh water	41	3.2%
Temperate and Boreal Forests	105	8.1%
Woodlands	43	3.3%
Grasslands	62	4.8%
Desert	3	0.2%
Cultivated	39	3.0%
Urban	4	0.3%
Multiple Ecosystems	32	2.5%
Coral Reefs	160	12.3%
Tropical Forest	260	20.0%
Coastal wetlands	186	14.3%
Total	1298	

Table 8 Frequency of studies by region in the TEEB database.

Region	Frequency	Percentage
Africa	231	18%
Americas	142	11%
Asia	340	26%
Europe	172	13%
Latin America and the Caribbean	178	14%
Oceania	105	8%
Global studies	130	10%

[46] Estimates were obtained for household numbers in Denmark, Finland and Australia (Queensland) from national statistical agency online databases.
[47] The reason for converting a reported US$ estimate to local currency using the appropriate PPP exchange rate and then back to 2007 US$ was so as to track changes in the local currency, which is arguably more methodologically defensible for values elicited from local residents. Those studies that elicited values from foreign visitors were not subject to this two-stage conversion.

In addition to the variables contained in the studies themselves, we added a number of site-specific spatial variables from a range of biophysical and socio-economic datasets (see Section 8) to the dataset used in this study. These site-specific variables are used in value function estimation and also for the subsequent value transfer.

7.3 QA forest biome database description

Following the review of the TEEB database 58 temperate forest and 103 tropical forest values were selected for inclusion in the QA database. A further 16 values were obtained for the woodlands biome; given this small number these 16 studies were included with the temperate forest biome for value function development and transfer[48]. Table 9 summarises the ecosystem service categories represented by the values for temperate and tropical forest biomes; site locations and services are illustrated in Figure 4. There is a clear difference between the biomes with a higher representation of provisioning and regulating services in the tropical forest biome. The main provisioning services considered are non timber forest products (NTFP), particularly food resources, and the provision of raw materials. The regulating services cover a range of ESSs including climate regulation, moderation of extreme events, regulating water flow, waste treatment, erosion prevention and pollination. The wide range of services included in the tropical biome studies is due primarily to the nature of those studies which purposefully set out to estimate values for all ESSs provided. By contrast, nearly half of the temperate forest biome values relate to cultural services, specifically recreation.

We can speculate that the reason for these differences between studies for the forest biomes is that in temperate regions ‘natural’ forests have been more fully exploited. The motivation for a primary valuation study is often the potential conversion from forest to other land uses (e.g. agriculture). Tropical forests are relatively under-exploited (at least in respect of our study sites) and thus more complete information on service provision is needed to balance trade-offs in land-use decisions. The other major difference between service coverage between the biomes is that there is a higher proportion of studies (17% versus 4%) relating to supporting services in the temperate forest studies; these all relate to gene pool protection, i.e. an approximate proxy for biodiversity conservation. Regulating service values make up a fifth of the temperate forest values; however the coverage is less even across services when compared to tropical forest values.

Table 10 summarises the valuation methods used in the studies and distribution of valuation methods used between the two forest biomes reflects the categories of ecosystem services that the studies cover (an introductory explanation of valuation methods is provided in Pascual *et al.* 2010). Contingent valuation was used in over half of the temperate forest studies reflecting the dominance of recreational values collected. By contrast 40% of the tropical forest values were collected using market values - for example for NTFP these would reflect either the market values of selling those products or the market cost of substitutes. 23% of tropical forest values reflect production function or factor income values for regulating services. The locations of the study sites for each of the forest and woodland biomes are illustrated in Figure 4 (note that multiple values may have been obtained for individual sites).

[48] We refer in this section and throughout this report to two forest biomes: (1) temperate forests and woodland and (2) tropical forests. However Table 9 and Table 10 provide the disaggregated analysis for completeness, i.e. temperate forests and presented separately to woodlands.

Table 9 Ecosystem service categories covered by the temperate and tropical forest studies.

Ecosystem service category	Temperate Forest		Woodlands		Tropical Forest	
Provisioning services	8	14%	10	63%	43	42%
Regulating services	11	19%	1	6%	32	31%
Cultural services	28	48%	2	13%	22	21%
Supporting services	10	17%	2	13%	4	4%
Total economic value	1	2%	1	6%	2	2%
Total	58		16		103	

Table 10 Valuation methods used by forest biome.

Valuation method	Temperate Forest		Woodland		Tropical Forest	
Contingent valuation	32	55%	2	13%	10	10%
Contingent ranking	3	5%	0	0%	0	0%
Choice experiment	0	0%	1	6%	2	2%
Group valuation	0	0%	0	0%	1	1%
Hedonic pricing	0	0%	0	0%	0	0%
Travel cost	1	2%	0	0%	8	8%
Replacement cost	1	2%	2	13%	1	1%
Factor income / Production function	2	3%	0	0%	24	23%
Market price	11	19%	11	69%	41	40%
Opportunity cost	0	0%	0	0%	0	0%
Avoided cost	7	12%	0	0%	15	15%
Other/unknown	1	2%	0	0%	1	1%
Total	58		16		103	

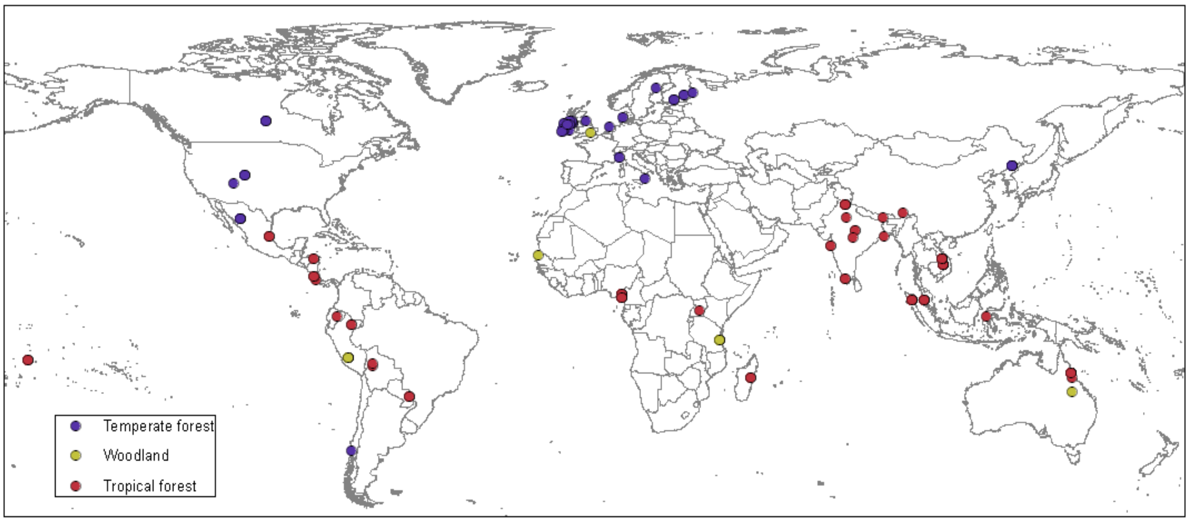

Figure 4 Forest biome site locations and services.

7.4 QA grassland biome database description

We collected and reviewed 27 studies that estimate values for ecosystem services from grasslands. Of these studies, there are 11 that provide both original value estimates (not benefit transfers) and complete information on all the explanatory variables that we include in the estimated value function (see Section 8). From the 11 selected studies we are able to code 19 separate value observations. We therefore obtain multiple value observations from single studies, with an average of 1.7 observations per study. Separate value observations from a study were included if they represent different study sites or ESSs.

The studies included in our analysis were published between the years 1995 and 2010. The locations of study sites included in the database are largely in Northern Europe, with studies in the Netherlands, United Kingdom, Sweden and Germany. We include one study from North America (Colorado, United States), two from Africa (South Africa and Botswana), and two from Asia (Israel and the Philippines). We have no information on the value of ESSs from grasslands in South America. A summary of ESS provision across these selected studies is provided in Table 11.

Table 11 Ecosystem service categories valued in grassland studies.

Ecosystem service	Number of observations	Percentage
Food provisioning	6	32%
Recreation and amenity	7	37%
Erosion prevention	3	16%
Conservation	3	16%

Table 12 provides a synopsis of the valuation methods used to estimate ESS values for grasslands. We find that the most commonly employed method is to estimate replacement costs for lost ESSs - food provisioning and erosion prevention. The contingent valuation and choice experiment methods have been used to value recreational uses of grasslands and wildlife conservation, the hedonic pricing method to estimate the amenity value of grasslands, and the net factor income method and market prices have been used to value food provisioning.

Table 12 Valuation methods used in grasslands studies.

Valuation method	Number of observations	Percentage
Contingent valuation	5	26%
Choice experiment	2	11%
Hedonic pricing	1	5%
Net factor income	1	5%
Replacement cost	6	32%
Market prices	4	22%

7.5 QA wetland biome database description

The wetlands dataset is an extension of the data described in Brander *et al.* (2006) and Ghermandi *et al.* (forthcoming). The dataset described here excludes value estimates for mangroves since these are treated as a separate biome in the present study.

We collected and reviewed almost 400 studies that estimate values for ESSs from wetlands. Of these studies, there are 131 that provide complete information on all the explanatory variables that we include in the estimated value function. From the 131 selected studies we are able to code 247 separate value estimates. We therefore obtain multiple value estimates from single studies, with an average of 1.9 estimates per study. Separate value estimates from a study were included if they represent different study sites, sample populations, ecosystem services, or valuation methods. These characteristics of value estimates are explicitly coded in the database.

The studies included in our analysis were published between the years 1974 and 2006. The geographic distribution of study sites included in the database provides a fairly wide global representation. There are a large number of wetland valuation studies in North America and Western Europe but also a reasonable number in Africa, South East Asia, and Australasia. The regions that are less well represented in the data are Latin America, Eastern Europe, the former USSR, and China. The geographic distribution of study sites is presented in Figure 5.

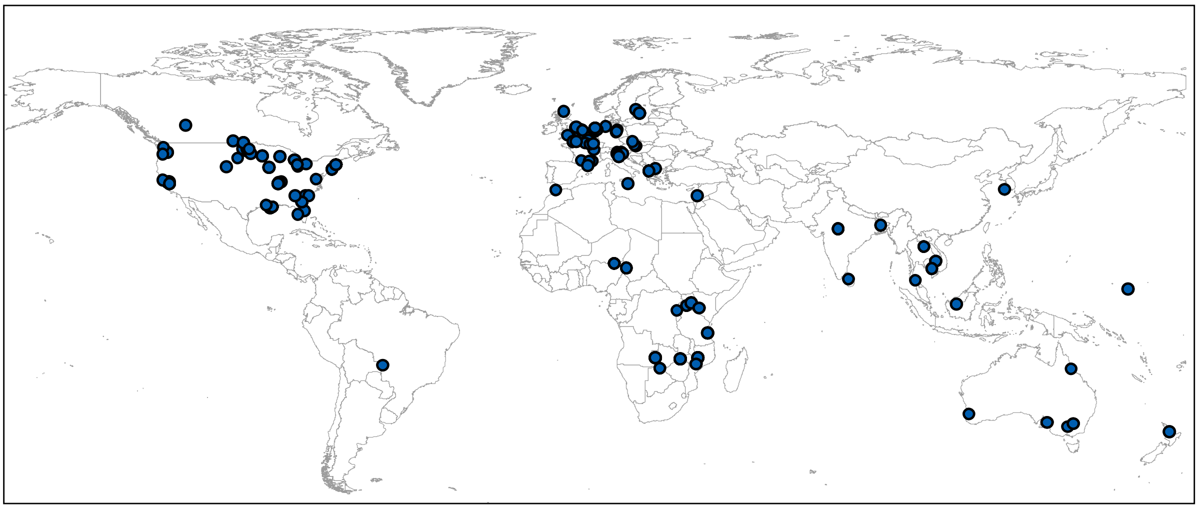

Figure 5 Locations of wetland value study sites

We find that a wide range of ESSs are well represented in the available literature. Table 13 presents the number of value estimates for the set of ESSs coded in our data. Many of the value estimates included in our data are for the provision of more than one ESS. The number of observations presented in the second column of Table 13 therefore sums to a number much larger than our total sample size. Cultural services such as recreation, amenity, and biodiversity conservation are particularly well represented in our data. There are also a large number of value estimates for regulating and provisioning services.

Table 13 Ecosystem service categories valued in wetland studies.

Ecosystem service	Number of observations
Flood protection	38
Water supply	33
Water quality	42
Habitat and nursery	47
Recreational hunting	52
Recreational fishing	45
Food and material provisioning	36
Fuel wood provisioning	11
Non-consumptive recreation	65
Amenity	34
Biodiversity conservation	32

Regarding valuation methods used to estimate ecosystem service values for wetlands we find that a wide variety of methods have been applied. Table 14 presents the number of value observations for each valuation method. The contingent valuation method has been the most frequently used and also the most widely applied in terms of ESSs valued. Market prices have mainly been used to value provisioning services, such as materials and food harvested from wetlands. The replacement cost method has largely been used to value regulating services, such as flood control and water quality.

Table 14 Valuation methods used in wetland studies.

Valuation method	Number of observations
Contingent valuation	62
Hedonic pricing	4
Travel cost method	38
Replacement cost	50
Net factor income	32
Production function	16
Market prices	53
Opportunity cost	9
Choice experiment	7

7.6 QA mangroves biome database description

The mangroves dataset is an extension of the data described in Brander *et al.* (2006) and Ghermandi *et al.* (forthcoming). The dataset described here excludes value estimates for other wetland types since these are treated as a separate biome in the present study. The data has been extended to include a number of recent mangrove valuation studies.

We found 48 mangrove valuation studies that provide complete information on all the explanatory variables that we include in the estimated value function. From the 48 selected studies we are able to code 111 separate value estimates. We therefore obtain multiple value estimates from single studies, with an average of 2.3 estimates per study. Separate value estimates from a study were included if they represent different study sites, sample populations, ecosystem services, or valuation methods. These characteristics of value estimates are explicitly coded in the database.

The studies included in our analysis were published between the years 1975 and 2009. The geographic distribution of study sites is presented in Figure 6. The data shows a reasonably good representation of regions with mangrove ecosystems. There are a large number of studies for South East Asia, Central America and the United States gulf coast. There are a more limited number of studies for East Africa, Australasia and the Pacific. We found no mangrove valuation studies for West Africa.

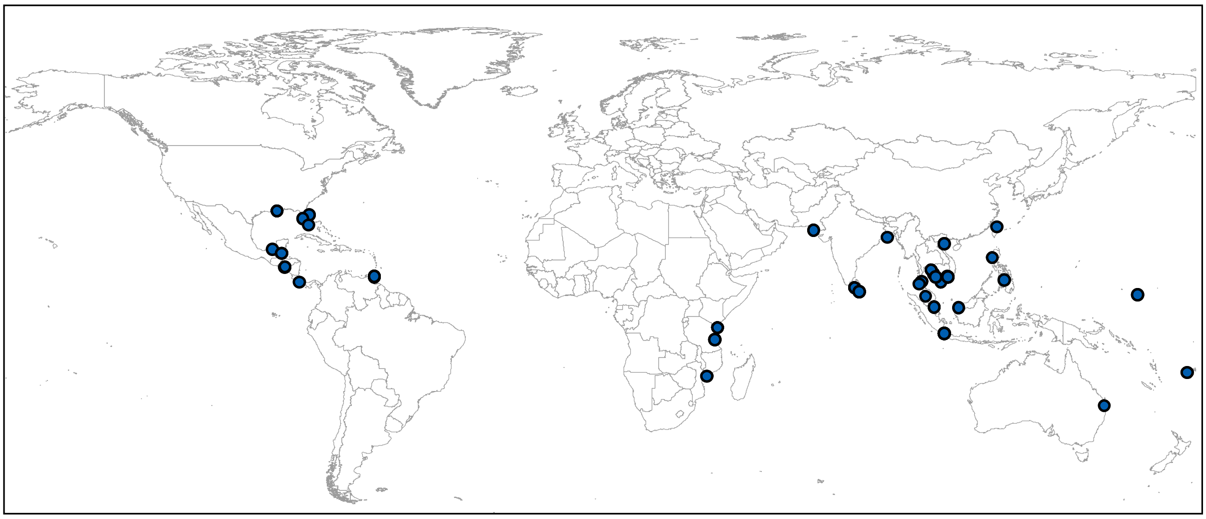

Figure 6 Locations of mangrove value study sites

The ESSs valued in this literature mainly relate to provisioning services (e.g., food, building materials and fuel wood) and regulating services (e.g., habitat and nursery functions for fisheries and coastal protection). Table 15 presents the number of value estimates for the set of ESSs coded in our data. Cultural services such as recreation, amenity, and biodiversity conservation are less well represented in our data.

Table 15 Ecosystem service categories valued in mangrove studies.

Ecosystem service	Number of observations
Coastal protection	16
Water supply	5
Water quality	6
Habitat and nursery	47
Recreational hunting	6
Recreational fishing	11
Food and material provisioning	37
Fuel wood provisioning	30
Non-consumptive recreation	14
Amenity	2
Biodiversity conservation	8

Regarding valuation methods used to estimate ecosystem service values for mangroves we find that estimates are often based on direct market prices of food and materials that are extracted from mangroves. The net factor income method has also been widely applied to estimate values for these services. The replacement cost method has largely been used to value coastal protection services. Table 16 presents the number of value observations for each valuation method. In some cases, estimated values are based on the application of more than one valuation method.

Table 16 Valuation methods used in mangrove studies.

Valuation method	Number of observations
Contingent valuation	10
Travel cost method	5
Replacement cost	16
Net factor income	23
Production function	12
Market prices	69
Choice experiment	2

7.7 QA coral reefs biome database description

The coral reefs dataset is an extension of the data described in Brander *et al.* (2007) and Brander *et al.* (2009). The data has been extended to include a number of recent coral reef valuation studies.

We collected and reviewed 168 valuation studies related to ecosystem services from coral reefs. Of these studies, 72 were found to provide complete information on all the explanatory variables that we include in the estimated value function. From the 72 selected studies we are able to code 163 separate value estimates. We therefore obtain multiple value estimates from single studies, with an average of 2.3 estimates per study. Separate value estimates from a study were included if they represent different study sites, sample populations, ecosystem services, or valuation methods. These characteristics of value estimates are explicitly coded in the database.

The studies included in our analysis were published between the years 1973 and 2010. The geographic distribution of study sites is presented in Figure 7. The data shows a reasonably good representation of regions with coral reef ecosystems. The Caribbean, Indian Ocean, South East Asia, and Pacific are all well represented.

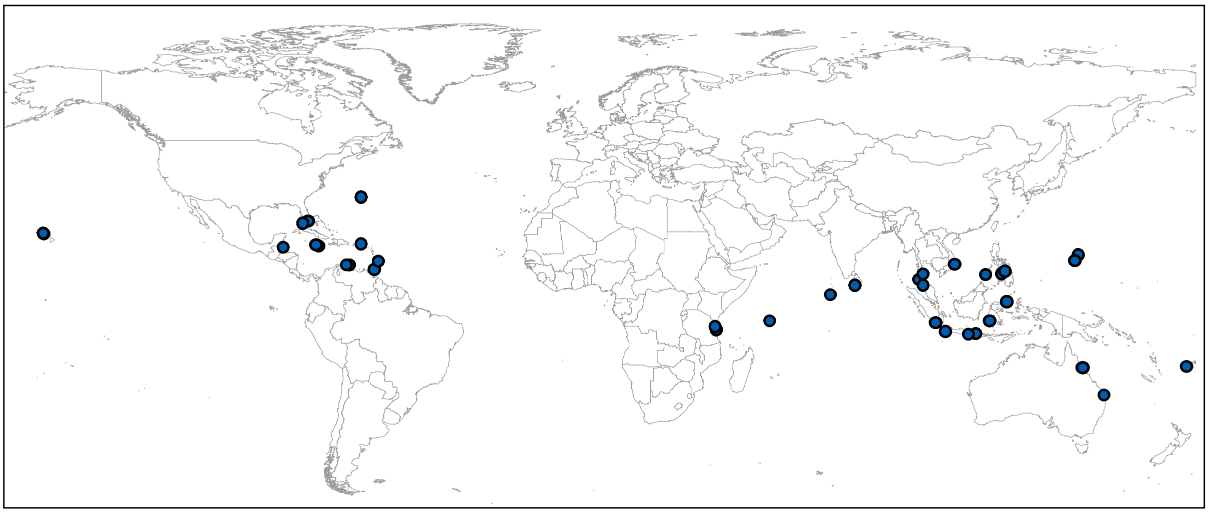

Figure 7 Locations of coral reef value study sites

The ESSs valued in this literature mainly relate to cultural services, and particularly to tourism and recreational activities. Table 17 presents the number of value estimates for the set of ESSs coded in our data. Regulating services such as coastal protection and habitat and nursery functions for commercial fisheries are also represented in the data. There are also a number of value estimates available for non-use values for coral reefs related to their preservation and existence.

Table 17 Ecosystem service categories valued in coral reef studies.

Ecosystem service	Number of observations
Recreational diving	71
Recreational snorkelling	60
Other tourism activities	49
Recreational fishing	15
Commercial fisheries	23
Coastal protection	14
Coral mining	0
Biodiversity	6
Research	4
Bio-prospecting	0
Non-use values	21

Regarding valuation methods used to estimate ESS values for coral reefs, we find that recreational values have largely been estimated using stated preference methods including both contingent valuation and choice experiments. The travel cost method has also been used to estimate the value of international tourism related coral reefs. The net factor income, market prices, and production function methods have been used to value coral reef inputs into commercial fisheries. The replacement cost method has largely been used to value coastal protection services. Table 18 presents the number of value observations for each valuation method.

Table 18 Valuation methods used in coral reef studies.

Valuation method	Number of observations
Stated preference	64
Hedonic pricing	5
Travel cost method	18
Replacement cost	5
Net factor income	19
Production function	10
Market prices	36

7.8 QA lakes and rivers biome database description

This section describes the value data used for estimating the value function for changes in *water quality* in surface water. This dataset is somewhat different from the data compiled for the other biomes in that the value information relates to changes in quality as opposed to quantity. The standardised measure of value in this case is willingness to pay per household per year for a change in water quality. This difference also has implications for the way in which the transfer and scaling up of values is conducted.

We collected and reviewed 154 contingent valuation studies that estimate values for ESSs related to surface water quality. Of these studies, there are 54 that provide complete information on all the explanatory variables that we include in the estimated value function. The most important missing information, resulting in a study being omitted from our analysis, was a clear description of the change in water quality that could be translated to a standardised measure. The standardised measure of water quality that we use is described in detail in Section 8.8 with the other explanatory variables. From the 54 selected studies we are able to code 388 separate value observations. We therefore obtain multiple value observations from single studies, with an average of 7.2 observations per study. Separate value observations from a study were included if they represent different study sites, sample populations, ecosystem services, elicitation formats, or estimation methods. These characteristics of value observations are explicitly coded in the database.

The studies included in our analysis were published between the years 1981 and 2006. The locations of study sites included in the database are largely in North America and Europe with a small number in Central America, South East Asia, China and Australia. The geographic distribution of study sites is presented in Figure 8. There is clearly a lack of available value information for large areas of the world, which raises questions of the representativeness of the compiled data and the validity of transferred values. Preferences for changes in water quality may differ across cultural and socio-economic context and we lack information about such preferences in South America, Africa and large parts of Asia.

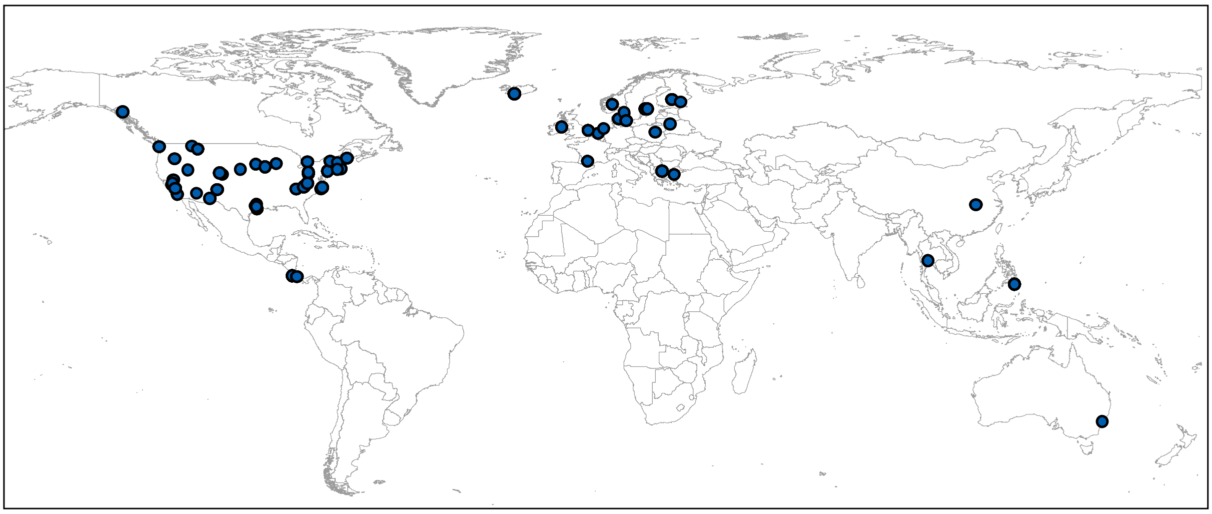

Figure 8 Locations of water quality value study sites

In terms of ESSs valued in this literature, we find that both use and non-use values are well represented in our data. Table 19 presents the number of value observations for the set of ESSs coded in our data. Many of the value observations included in our data are for multiple ESSs. The number of observations presented in the second column of Table 19 therefore sums to a number much larger than our total sample size. The ESS that is best represented in our data is non-use value related to preservation or improvement in water quality unrelated to any current or potential future use of the resource. Most contingent valuations comprise an element of non-use value in combination with use values but it is difficult to assess the size of this component of total value. Direct use values related to water are also well represented in our data. These are mainly related to recreational activities. Provisioning services such as drinking water and irrigation are less well represented. This may reflect the priorities for water use at the locations where valuation studies have been conducted. The focus of studies in North America and Europe has been on direct recreational uses and non-use values. The priority uses of surface water in developing countries, which are poorly represented in our data, are likely to be provisioning services such as drinking water. The values that we scale up are limited to the available value information and may therefore underestimate the importance of provisioning services in developing countries.

Table 19 Ecosystem service categories valued in lakes and rivers studies.

Ecosystem service	Number of observations
Drinking water	17
Irrigation	3
Nature conservation	80
Fishing	151
Boating	128
Swimming	119
Walking	10
Other recreation	29
Health	4
Amenity	21
Non-use	275

7.9 Overall summary on biome databases

The databases used in this study have been formed using (1) a TEEB-commissioned study which collected biome-specific and ecosystem service-specific valuations (de Groot *et al.*, 2010) and (2) proprietary databases developed in previous research (e.g. Brander, 2006) for coastal biomes. Irrespective of the source for the studies, each entry in the database for the current study was screened and data added vis-à-vis GIS coordinates to allow the development of (1) biome-level benefit functions (see Section 8) and (2) subsequent value transfer based on spatially-specific characteristics for individual patches of land that vary in extent based on what scenario option is applied (see Results in Section 12).

The approach adopted for estimating the benefits of each change scenario is in part determined by the outcomes of this database development. In particular, benefit transfer on the basis of transferring values for ESSs depends on the availability of sufficient observations for any given ESS-biome combination; this condition does *not* apply for most combinations. Table 20 provides a synopsis. Further, there are also methodological issues in such a benefit transfer exercise (see for instance Barbier *et al.*, 2008). Thus the only ESS treated independently is carbon storage (see results Section 12).

Table 20 Summary of biome databases

Ecosystem service category	Forests and Woodlands	Grassland	Wetland	Mangroves	Coral Reefs	Lakes and Rivers
Provisioning services	34%	32%	18%	40%	9%	3%
Regulating services	25%	16%	18%	12%	5%	0%
Cultural services	29%	37%	45%	18%	76%	52%
Supporting services	9%	16%	18%	30%	10%	45%
Total economic value	2%	0%	0%	0%	0%	0%

8 Biome-level value functions

8.1 Introduction

The aim of benefit function estimation is to produce a model that explains variation in site values (in this case US$/ha) in both a theoretically and statistically robust manner. That is, the explanatory variables should have some reasonable theoretical justification for both having an effect and the direction of that effect (the sign of the estimated coefficient); that effect should also have reasonable level of statistical significance.

An important decision in function estimation is the choice of functional form to be used. Common throughout the meta-analysis and benefit transfer literature is the use of either log or log-log functions. In log forms a natural logarithm transformation of the dependent variable (unit value) is used; in log-log the transformation is applied to both dependent and independent variables. There are a number of reasons why a log or log-log functional form is attractive (see Brander *et al.*, 2006). Often values follow skewed (non-normal) distributions with a small number of outlying values; a log transformation counteracts this by reducing the effect of extreme values and the resulting data more closely reflect a normal distribution and has a smaller variance. The use of a log-log specification allows the normalisation of both dependent and independent variables and has the further advantage that the estimated coefficients can be interpreted as elasticities, i.e. the coefficients represent the percentage change in the dependent variable (value per ha) of a small percentage change in the explanatory variable (Brander *et al.*, 2006).

For example the effects of the log transformation on per ha values for both forest biomes are illustrated in Figure 9 and Figure 10. In Figure 9 the majority of observed values occur towards the left of distribution, the highest value is US$15,345/ha and the mean is US$595/ha whereas the median value is US$29/ha. The natural logarithm transformation of the unit values is shown in Figure 10. However this also illustrates a problem with the transformation, i.e. original values that are less than 1 become negative following transformation. This is clearly problematic when considering values which should be non-negative. A similar problem arises with some environmental and socio-economic variables: if they have the value of 0 they cannot be calculated (i.e. $\ln(0) = -\infty$). To avoid this problem and to truncate the log transformed values at zero, one was added to all variables that were to be transformed and that had values within the zero to one range. This mean that any value that was zero remained so following transformation, values between 0 and 1 remained positive, and values of exactly 1 remained non-zero. Figure 11 illustrates the effect of this transformation.

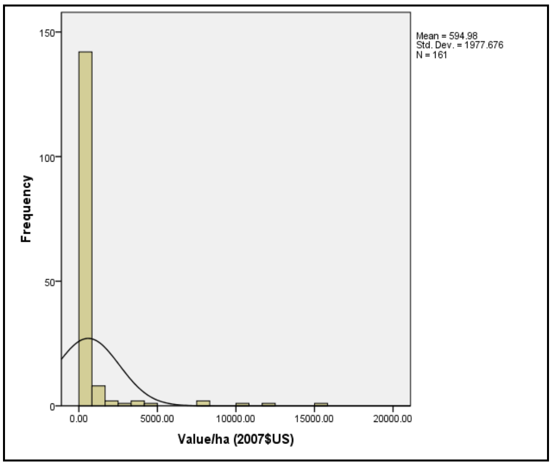

Figure 9 Distribution of forest biome values.

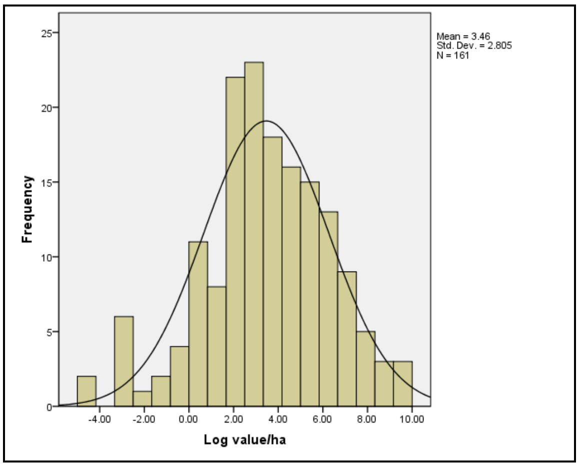

Figure 10 Distribution of natural logarithms of forest biome values

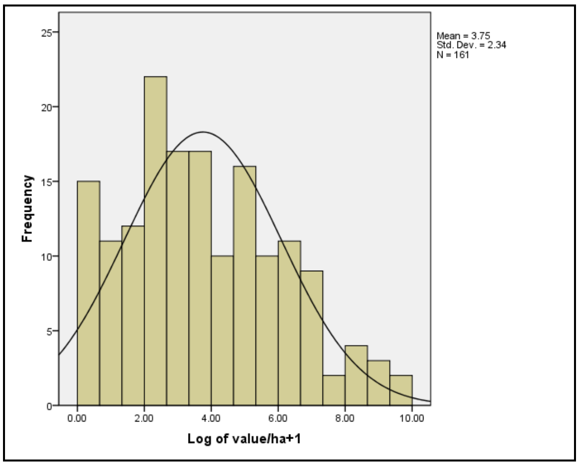

Figure 11 Distribution of truncated natural logarithms of forest biome values

In addition to functional form a major consideration in the development of benefit functions is the choice of explanatory variables. As noted above these should be theoretically valid and have a significant effect on per ha values. A further consideration with benefit transfer exercises is that they should also be observable for the sites to which benefits are to be transferred. It is common in meta-analyses of valuation studies to include study-specific variables that relate particularly to the methodology that was applied. The effect of different valuation methods or the different value elicitation approaches have been found to be significant explanatory variables; see Bateman and Jones (2003), Lindhjem and Navrud (2008), and Barrio and Loureiro (2010) for examples of meta-analyses of forest valuation studies where methodological variables were found to be significant. However, although such analyses are of theoretical interest and can be useful in guiding methodological development they are of little value in benefit transfer as such variables are essentially unobservable. Similarly site-specific variables that cannot be observed across transfer sites are of little use.

We discuss below possible reasons for the occurrence of positive or negative signs on various variables, on a biome-by-biome basis; the degree to which we have confidence in this interpretation of the results varies. For instance, we would certainly expect patch value to be positively linked with income. However, the accessibility variable for instance is more complicated to interpret; it shows the potential for the study site to generate positive on-site use values (e.g. for recreation) but also the ease by which the site might be exploited and degraded (which reduces ecosystem service values).

We use a range of spatially referenced variables that are derived from publically available data sources and are applied to the study sites by Geographical Information Systems (GIS) analysis of each site's location. Table 21 summarises the spatial variables estimated for the study sites that can also be applied to all transfer sites. GIS is used to transform and integrate a series of global spatial datasets into separate datasets that spatially cover the seven biomes under investigation. Note that spatial variables are applied at three different radii from the patch: 10km, 20km and 50km.

The GIS is used to transform the different spatial input data, such as global population, infrastructure, urbanization and human appropriation of net primary product (HANPP) into a dataset of specific spatial variables (e.g. area, abundance). The spatial data selection is based on the following criteria: (1) possible explanatory value for ecosystem value estimates; (2) completeness vis-à-vis global extent; (3) spatial and temporal consistency; and (4) credibility, i.e. well-documented and preferably scientifically-referenced data. Further details are provided in Appendix II which specifically pertains to the GIS analysis.

There are four chronologically executed stages to the GIS integration and analysis work. The first three pertain to the benefit function estimation:

1. spatial data selection, acquisition, transformation and integration of input data for spatial variables and biome maps;
2. import of study sites into the GIS data base as point locations, based on their estimated geographic coordinates; and
3. extraction of spatial variable values to point-based study site locations as input for meta-regression analysis.

The fourth chronological step (upscaling of spatial relationships resulting from the meta-regression analysis between ecosystem values and explanatory spatial variables to a global scale) takes place after the generation of the biome-level value functions. We thus return to GIS Step 4 in Section 8.9 after discussing the biome-level meta-regression.

Table 21 Spatial variables used in benefit function development.

Variable	Description	Comments	Source
Forests	Area (ha) of forest within specified radius of site	Measure of substitute and/or complimentary sites	The Global Land Cover Map for the Year 2000, 2003. GLC2000 database, European Commission Joint Research Centre. http://www-gem.jrc.it/glc2000.
Mangrove	Area (ha) of mangrove within specified radius of site	Measure of substitute and/or complimentary sites	Mangrove GIS data in shapefile format (V 3.0 1997), Mangroves of Western Central Africa GIS dataset in shapefile format (V 1.0 2006). UNEP World Conservation Monitoring Centre.
Grassland	Area (ha) of grassland within specified radius of site	Measure of substitute and/or complimentary sites	The Global Land Cover Map for the Year 2000, 2003. GLC2000 database, European Commission Joint Research Centre. http://www-gem.jrc.it/glc2000.
Wetlands	Area (ha) of wetlands within specified radius of site	Measure of substitute and/or complimentary sites	Global lakes and wetlands database GLWD. World Wild Life – WWF and Center for Environmental Systems Research, University of Kassel, Germany. http://www.worldwildlife.org/science/data/item1877.html
Rivers and lakes	Area (ha) of rivers and lakes within specified radius of site	Measure of substitute and/or complimentary sites	Global lakes and wetlands database GLWD. World Wild Life – WWF and Center for Environmental Systems Research, University of Kassel, Germany. http://www.worldwildlife.org/science/data/item1877.html
Coral reef	Area (ha) of coral reef within specified radius of site	Measure of substitute and/or complimentary sites	Coral reef 1km data in ESRI Grid format and Shapefile format (V 7.0 2003)
Gross cell product	Measure of gross value added (ppp US$ 2005) within specified radius of site	Measure of economic output that acts as proxy for ability (willingness) to pay for ecosystem services	Global Economic Activity G-Econ 3.3. http://gecon.sites.yale.edu/data-and-documentation-g-econ-project
Population	Population density (2000 persons/km^2) within specified radius of site	Measure of population likely to benefit from ecosystem services and/or proxy measure of pressure	Socio-Economic Data Center (SEDAC) Columbia University. http://sedac.ciesin.columbia.edu/gpw/global.jsp
Urban area	Area (ha) of urban land use within specified radius of site	Measure of presence of population likely to benefit from ecosystem services and/or proxy measure of pressure	Institute for Environmental Studies, University of Wisconsin-Madison http://www.sage.wisc.edu/people/schneider/research/data.html
Roads	Length (km) of roads within specified radius of site	Measure of accessibility and/or fragmentation of site	FAO - UN SDRN http://www.fao.org:80/geonetwork?uuid=c208a1e0-88fd-11da-a88f-000d939bc5d8
Net primary product (NPP)	Net primary product of actual vegetation (gC/m^2/yr) within specified radius of site	Proxy measure for production of ecosystem services of site and substitutes	Institut für Soziale Ökologie IFF - Fakultät für interdisziplinäre Forschung und Fortbildung der Alpen-Adria-Universität Klagenfurt Wien, Österreich. http://www.uni-klu.ac.at/socec/inhalt/1191.htm
Human appropriate of NPP	Human appropriation of NPP (gC/m^2/yr) within specified radius of site	Proxy measure of human exploitation of ecosystem services and/or land management – primarily agricultural land	Institut für Soziale Ökologie IFF - Fakultät für interdisziplinäre Forschung und Fortbildung der Alpen-Adria-Universität Klagenfurt Wien, Österreich. http://www.uni-klu.ac.at/socec/inhalt/1191.htm
Accessibility index	Index of accessibility based on distance in travel time to urban centres	Measure of accessibility and use of ecosystem services of site	Aurelien Letourneau, Wageningen University aurelien.letourneau@wur.nl

In the sections that follow (Sections 8.2-8.8) we present the benefit functions used in this study for each biome in turn. The dependent variable in every case except the lakes and river biome is US$/ha/annum in 2007 price levels; for lakes and rivers the dependent variable is willingness to pay (WTP) per household per year for a change in water quality in US$ 2007 price levels. In all cases the value functions are estimated by ordinary least squares regression (OLS) using SPSS 16.0[49].

What we do not consider is the change in habitat type described in the primary valuation studies; the US$/ha/annum value estimate for a particular (say) woodland site depends on what the proposed alternative land use is. We do not apply a filter vis-à-vis the alternative land use as to do so would imply having smaller sub-sets of data points for each biome (e.g. only those studies proposing woodlands conversion to pasture land), and in terms of our patch-level analysis there is insufficient spatial resolution to identify the nature of land use changes for each patch.

Note that only three biomes are pertinent to our analysis of the change scenarios set out in Section 6: temperate forests and woodlands; tropical forests; and grasslands. The other biomes described in Section 8 are discussed further in a complimentary analysis provided in Section 10. This restriction to the three terrestrial biomes arises as IMAGE-GLOBIO does not explicitly model changes to these other non-terrestrial biomes with respect to the change scenarios.

8.2 Temperate forests and woodlands

The average temperate forest and woodland value is US$892/ha/annum and the median is US$127/ha/annum. The dependent and explanatory variables for the temperate forest and woodland biomes are summarised in Table 22. A number of other explanatory variables that could be observed across both the primary valuation sites and transfer sites were tested and found not to be significant; these included location-specific variables such as regional dummies. The benefit function outlined in

Table 23 was found to have the best performance in terms of variable significance and goodness-of-fit. The estimated coefficients have the expected signs. The negative sign on the log of site area indicates that values per ha decline as the size of the site increases, i.e. diminishing margin values. The log of gross cell product within 50km is positive indicated that site values increase with income. The positive sign on the log of urban area within 50km of the sites suggests that values for natural areas increases with the local urban population; this would be expected given the predominance of recreational values in the temperate forest studies. The final independent variable included is the log of human appropriation of net primary product (NPP) within 50km of the study sites, a proxy for land-use intensity. The negative sign on the estimated coefficient could be interpreted to mean that more intensive land use surrounding forest sites reduces their value, but we accept that interpreting the sign on this variable is less straight-forward.

[49] We use OLS notwithstanding the fact that willingness-to-pay is truncated at $0; the truncation is due to the log transformation rather than being otherwise imposed on the data.

Table 22 Dependent and explanatory variables for temperate forests and woodlands.

Variable name	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VALUE_07	Site value 2007 US$/ha/annum	892.20	127.45	2544.55	0.07	15344.79
LN_VAL	Natural log of site value US$/ha/annum	4.68	4.86	2.32	0.07	9.64
LN_AREA	Natural log of the study site area	10.63	12.00	4.85	4.17	19.30
LN_GCP50	Natural log of Gross Cell Product within 50km radius	6.91	6.39	1.68	4.12	10.44
LN_URB50	Natural log of urban area within 50km radius of study site	2.60	3.22	1.91	0.00	6.09
LN_HAN50	Natural log of human appropriation of NPP within 50km radius of study site	13.63	14.00	0.94	11.18	15.02

Table 23 Temperate forest and woodland value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		28.627	6.124	0.000
LN_AREA	Natural log of the study site area	-0.420	0.076	0.000
LN_GCP50	Natural log of Gross Cell Product within 50km radius	0.247	0.150	0.104
LN_URB50	Natural log of urban area within 50km radius of study site	0.245	0.143	0.092
LN_HAN50	Natural log of human appropriation of NPP within 50km radius of study site	-1.610	0.417	0.000
N		69		
Adjusted R^2		0.348		

The coefficients are significant at the widely accepted 5 and 10% levels, although the significance of LN_GCP50 (Gross Cell Product) is marginally insignificant under these criteria. However, removal of such variables can serve to reduce the significance of those remaining or the overall model performance. The adjusted R^2 indicates that this model accounts for 34.8% of the observed variation in log per ha values.

8.3 Tropical forests

The average tropical forest value is US$444.98/ha/annum and the median is US$14.86/ha/annum. Table 24 summarises the dependent and explanatory variables for tropical forests and Table 25 outlines the benefit function. There are four independent variables in common with the temperate forest function; these have the same signs and interpretation. The additional variables include the area of forest within 50km of the site and the length of roads within 50km; both of these have negative signs. For the former variable we suggest that this can be interpreted as the effect of having substitute sites in the same area that can provide a similar range of ESSs. This might reflect the greater continuity of forest cover in the tropical forest sites as compared to many of the temperate forest study sites where forest cover was more fragmented. The negative sign on the log of roads within 50km variable suggests that this variable might be a proxy for the degree of forest exploitation.

The adjusted R^2 figure indicates that 39.2% of observed variation in values is explained by the model. With the exception of the LN_HAN50 and LN_RDS50 variables each variable is significant at either the 5% or 10% level.

Table 24 Dependent and explanatory variables for tropical forests.

Variable name	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VALUE_07	Site value 2007 US$/ha/annum	444.98	14.86	1612.03	0.01	11706.50
LN_VAL	Natural log of site value US$/ha/annum	3.28	2.76	2.22	0.01	9.37
LN_AREA	Natural log of the study site area	11.68	12.06	2.82	1.10	16.59
LN_GCP50	Natural log of Gross Cell Product within 50km radius	5.51	5.11	1.93	0.00	8.90
LN_URB50	Natural log of urban area within 50km radius of study site	2.07	1.10	1.79	0.00	6.42
LN_HAN50	Natural log of human appropriation of NPP within 50km radius of study site	14.06	13.97	0.84	12.28	15.72
LN_FOR50	Natural log of area of forest within 50km radius of study site	7.96	8.47	1.84	0.00	8.95
LN_RDS50	Natural log of length of roads within 50km radius of study site	8.52	9.10	2.18	0.00	10.02

Table 25: Tropical forest value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		12.960	4.071	0.002
LN_AREA	Natural log of the study site area	-0.230	0.070	0.001
LN_GCP50	Natural log of Gross Cell Product within 50km radius	0.402	0.173	0.022
LN_URB50	Natural log of urban area within 50km radius of study site	0.424	0.121	0.001
LN_HAN50	Natural log of human appropriation of NPP within 50km radius of study site	-0.394	0.292	0.181
LN_FOR50	Natural log of area of forest within 50km radius of study site	-0.336	0.202	0.100
LN_RDS50	Natural log of length of roads within 50km radius of study site	-0.204	0.131	0.124
N		102		
Adjusted R^2		0.392		

8.4 Grasslands

Table 26 presents the dependent and explanatory variables included in the grassland value function together with the definition and descriptive statistics for each variable. The mean of grassland value is US$216/ha/annum and the median is US$37/ha/annum. These values are low in comparison to those of the other biomes examined in this study.

Given the very limited sample size of grassland ecosystem service values, the number of explanatory variables that can be included in the value function is also low. The explanatory variables included in the value function are GDP per capita; the area of grassland within a 50km radius of the study site; the length of road within a 50 km radius of the study site; and the accessibility index.

Table 26 Dependent and explanatory variables for grasslands.

Variable label	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VAL_HA	US$/ha/annum 2007 prices	216.51	37.48	568.93	0.08	2427.57
VAL_LN	Natural log of US$/ha/annum	3.48	3.64	2.13	0.08	7.80
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	9.86	10.22	1.03	6.60	10.53
GRA50_LN	Natural log of area of grassland within 50km radius of study site	1.83	0.00	3.65	0.00	8.96
RDS50_LN	Natural log of length of roads within 50km radius of study site	8.65	9.42	2.42	0.00	10.06
SITES_AI	Accessibility index	0.40	0.36	0.39	0.00	0.95

The value function is presented in Table 27. The estimated coefficients on the explanatory variables all have the expected signs but are mostly not statistically significant[50]. Only the estimated effect of accessibility is statistically significant at the 10% level, although the GDP per capita variable is significant at the 12% level. The positive coefficient on the income variable (GDP per capita) indicates that grassland ecosystem services have higher values in countries with higher incomes, i.e., grassland ecosystem services are a normal good for which demand increases with income. The negative effect of grassland abundance (area of grassland within 50km radius) on value indicates that the availability of substitute grassland areas affects the value of ecosystem services from a specific patch of grassland. The negative effect of roads on grassland values captures the effect of fragmentation on the provision of ESSs from grassland. Grasslands that are more fragmented by roads tend to have lower values. The positive coefficient on the accessibility index indicates that grassland areas that are more accessible tend to have higher values. In this case, direct use values derived from grasslands (e.g., recreation and food provisioning) appear to dominate values that do not require access (e.g. wildlife conservation).

Table 27 Grasslands value function

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		-2.366	5.094	0.444
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	0.856	0.514	0.120
GRA50_LN	Natural log of area of grassland within 50km radius of study site	-0.029	0.142	0.839
RDS50_LN	Natural log of length of roads within 50km radius of study site	-0.225	0.213	0.309
SITES_AI	Accessibility index	2.590	1.322	0.072
N		17		
Adjusted R^2		0.27		

Confidence in the estimated value function for grassland ecosystem services is not high. Although the adjusted R^2 of 0.27 for grasslands (which indicates that the estimated model explains 27% of variation in the value of grassland) is not much worse than the R^2 of 0.35 that applies for the temperate forests and woodlands biome, all but one of the explanatory variables included in the grasslands model are not statistically significant. The signs and magnitudes of effect of the explanatory variable do, however, make theoretical sense. We therefore cautiously use this value function to estimate site specific values for grasslands. Transferred values are checked for estimates that lie outside of the range of values observed in the literature.

8.5 Wetlands

In sections 8.5-8.8 we present the value functions for those biomes for which change scenario outcomes were not modelled in IMAGE-GLOBIO; thus they do *not* constitute part of

[50] The presence of insignificant independent variables is of concern; however the estimated model is otherwise theoretically consistent. The lack of significance indicates low precision in the degree to which the coefficients predict the effect of the independent variables on per ha values. We would argue that rejecting the value function for this biome entirely would result in the omission of potentially significant values in our subsequent analysis.

the assessment of change scenarios. The analysis of these biomes is included however in an adjunct section (Section 10).

Table 28 presents the dependent and explanatory variables included in the wetland value function together with the definition and descriptive statistics for each variable. The average wetland value is US$4,774/ha/annum and the median is US$250/ha/annum.

The explanatory variables included in the value function are as follows: the area of the wetland study site; the GDP per capita of the country in which the study site is located; the area of lakes and rivers within a 50km radius of the study site; the area of wetland within a 50km radius of the study site; the population within a 50km radius; and the human appropriation of net primary product (HANPP) within a 50km radius.

The value function is presented in Table 29. The estimated coefficients on the explanatory variables all have the expected signs and are all statistically significant at the 5% level, except for HANPP, which is significant at the 10% level. The negative effect of the area of the wetland indicates diminishing returns to scale for wetland values. In other words, the value of an additional hectare to a large wetland is of lower value than an additional hectare to a small wetland. The positive effect of the income variable (GDP per capita) indicates that wetland ecosystem services have higher values in countries with higher incomes, i.e., wetland ecosystem services are normal goods for which demand increases with income.

The positive effect of the area of lakes and rivers in the vicinity of a wetland indicates that lakes and rivers are complements to wetland ecosystem services, i.e., that the combination of surface water bodies results in higher valued ecosystem services. The negative effect of the size of other wetland areas in the vicinity of a wetland indicates substitution effects between wetlands. The ESSs from a specific wetland will be of higher value if there are fewer other wetlands in the vicinity.

The positive effect of population on the value of wetland ecosystem services relates to market size or demand for ESSs. A larger population in the vicinity of a wetland means that more people benefit from the ESSs that it provides. The negative effect of HANPP indicates the effect of ecosystem degradation on the value of services provided by wetlands. More intensive use and appropriation of environmental resources has a negative effect on the value of wetland services.

Table 28 Dependent and explanatory variables for wetlands.

Variable label	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VAL_HA	US$/ha/annum 2007 prices	4773.57	250.19	15027.98	0.00	140218.75
VAL_LN	Natural log of US$/ha/annum	5.61	5.53	2.77	0.00	11.85
AREA_LN	Natural log of the study site area (ha)	9.35	9.24	3.24	0.18	16.45
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	9.38	10.20	1.48	6.40	10.64
LAK50_LN	Natural log of area of lakes and rivers within 50km radius of study site	4.42	4.69	2.02	0.00	8.60
WET50_LN	Natural log of area of wetlands within 50km radius of study site	4.99	6.01	3.28	0.00	8.96
POP50_LN	Natural log of population within 50km radius of study site	12.90	13.11	1.69	6.25	16.23
HAN50_LN	Natural log of human appropriation of NPP within 50km radius of study site	14.17	14.46	1.49	0.00	15.78

Table 29 Wetlands value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		1.708	1.978	0.725
AREA_LN	Natural log of the study site area (ha)	-0.209	0.049	0.000
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	0.610	0.106	0.000
LAK50_LN	Natural log of area of lakes and rivers within 50km radius of study site	0.159	0.081	0.050
WET50_LN	Natural log of area of wetlands within 50km radius of study site	-0.175	0.048	0.000
POP50_LN	Natural log of population within 50km radius of study site	0.426	0.106	0.000
HAN50_LN	Natural log of human appropriation of NPP within 50km radius of study site	-0.201	0.118	0.091
N		247		
Adjusted R^2		0.32		

8.6 Mangroves

Table 30 presents the dependent and explanatory variables included in the mangrove value function together with the definition and descriptive statistics for each variable. The average mangrove value is US$803/ha/annum and the median is US$220/ha/annum.

The explanatory variables included in the value function are as follows: the area of the mangrove study site; the GDP per capita of the country in which the study site is located; the population within a 50km radius; the length of roads within a 50km radius of the study site; the area of mangroves within a 50km radius of the study site; the area of urban land use within a 50km radius; and the area of wetland within a 50km radius of the study site.

The value function is presented in Table 31. The estimated coefficients on the explanatory variables mostly have the expected signs and are all statistically significant at the 5% level, except for the length of roads variable which is significant at the 10% level. The negative coefficient on the area of the mangrove indicates diminishing returns to scale. The positive effect of the income variable (GDP per capita) indicates that mangrove ESSs have higher values in countries with higher incomes. The positive effect of population on the value of mangrove ESSs relates to market size or demand for ESSs. A larger population in the vicinity of a mangrove means that more people benefit from the ecosystem services that it provides.

The positive effect of the area of other mangroves on the value of a mangrove study site indicates that mangrove patches within a region are complements. This suggests that isolated patches of mangrove are of lower value than more intact contiguous mangrove systems. The negative effect of the area of urban land uses in the vicinity of a mangrove reflects the associated effect of degradation on the value of ecosystem services. Similarly the negative effect of roads on mangrove ecosystem services reflects the detrimental effects of fragmentation. The negative coefficient on wetland area in the vicinity of a mangrove indicates substitution effects between wetlands and mangroves. The estimated value function is a relatively good fit of the data, with an adjusted R^2 of 0.41 showing that 41% of variation in mangrove values is explained by the model. This still means that 59% of variation in values is not explained by the variables included in the regression model.

Table 30 Dependent and explanatory variables for mangroves.

Variable label	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VAL_HA	US$/ha/annum 2007 prices	803.19	220.91	1556.39	0.00	8207.21
VAL_LN	Natural log of US$/ha/annum	4.92	5.40	2.25	0.00	9.01
AREA_LN	Natural log of the study site area (ha)	9.37	9.39	2.48	5.30	15.75
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	8.17	8.12	1.06	6.40	10.58
POP50_LN	Natural log of population within 50km radius of study site	13.43	13.71	1.48	9.05	15.96
RDS50_LN	Natural log of length of roads within 50km radius of study site	8.76	9.02	1.23	0.00	9.75
MAN50_LN	Natural log of area of mangrove within 50km radius of study site	3.76	4.63	2.35	0.00	8.53
URB50_LN	Natural log of urban area within 50km radius of study site	3.59	3.85	1.92	0.00	6.74
WET50_LN	Natural log of area of inland wetland within 50km radius of study site	4.03	4.99	2.87	0.00	8.49

Table 31 Mangrove value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		-8.239	3.157	0.010
AREA_LN	Natural log of the study site area	-0.311	0.069	0.000
GDPPC_LN	Natural log of country level GDP per capita (PPP US$ 2007)	1.499	0.218	0.000
POP50_LN	Natural log of population within 50km radius of study site	0.572	0.194	0.004
MAN50_LN	Natural log of area of mangrove within 50km radius of study site	0.208	0.083	0.014
URB50_LN	Natural log of urban area within 50km radius of study site	-0.382	0.173	0.029
RDS50_LN	Natural log of length of roads within 50km radius of study site	-0.317	0.182	0.084
WET50_LN	Natural log of area of inland wetland within 50km radius of study site	-0.158	0.064	0.016
N		111		
Adjusted R^2		0.41		

8.7 Coral reefs

Table 32 presents the dependent and explanatory variables included in the coral reef value function together with the definition and descriptive statistics for each variable. The average coral reef value is US$4,422/ha/annum and the median is US$772/ha/annum.

The explanatory variables included in the value function are as follows: the area of coral cover study site; the GDP per capita of the country in which the study site is located; the population within a 50km radius; the length of roads within a 50km radius of the study site; the human appropriation of net primary product within a 50km radius; the net primary product within a 50km radius; and the area of coral cover within a 50km radius of the study site.

Table 32 Dependent and explanatory variables for coral reefs.

Variable label	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VAL_HA	US$/ha/annum 2007 prices	4421.56	771.82	8226.64	0.03	40821.03
VAL_LN	Natural log of US$/ha/annum	6.51	6.65	2.41	0.03	10.62
AREA_LN	Natural log of the study site area (ha)	9.30	9.21	2.86	0.69	15.45
GCP50_LN	Natural log of Gross Cell Product within 50km radius	4.71	5.97	3.03	0.00	8.62
POP50_LN	Natural log of population within 50km radius of study site	11.74	12.55	2.75	0.00	14.85
RDS50_LN	Natural log of length of roads within 50km radius of study site	6.96	8.29	3.09	0.00	9.36
HAN50_LN	Natural log of human appropriation of NPP within 50km radius of study site	11.22	12.80	4.20	0.00	14.76
NPP50_LN	Natural log of NPP within 50km radius of study site	13.78	14.32	1.27	8.90	15.09
COR50_LN	Natural log of area of coral reef within 50km radius of study site	5.44	5.49	1.00	0.00	7.01

The value function is presented in Table 33. The estimated coefficients on the explanatory variables all have the expected signs but only the area of cover coral cover at the study site is statistically significant. The negative coefficient on the area of coral cover indicates diminishing returns to scale. The positive effect of the income variable (GDP per capita) indicates that coral reef ecosystem services have higher values in countries with higher incomes, i.e. coral reef ecosystem services are normal goods for which demand increases with income. This variable, however, is difficult to define and interpret clearly since the beneficiaries of coral reef ecosystem services are often not from the country in which the reef is located. This is the case for most tourism and recreational services. The positive effect of population on the value of coral reef ecosystem services relates to market size or demand for ecosystem services. A larger population in the vicinity of a coral reef means that more people benefit from the ecosystem services that it provides. The negative effect of the length of roads in the vicinity of a coral reef reflects the associated effect of fragmentation and degradation on shore. Similarly, the negative effect of HANPP indicates the extent of human exploitation of natural resources in the region. Often conversion of natural land uses or cultivation of crops results in increased sedimentation in coastal waters, which can negatively affect reefs. The negative coefficient on the area of coral reefs in the vicinity of a specific reef indicates substitution effects between patches of coral reef. In areas where coral reefs are abundant, the value of a specific patch of reef will be lower.

The adjusted R^2 is relatively low (0.18) indicating that the estimated model only explains 18% of variation in coral reef values. There are clearly a number of important factors that influence the value of coral reefs that are not captured by our set of explanatory variables. The direction and magnitude of estimated effects of our set of explanatory variables do, nevertheless, follow theoretical expectations. We therefore cautiously use the estimated value function to transfer values to changes in the extent of coral cover in Section 10 in the analysis of policy inaction.

Table 33 Coral reef value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		16.093	3.707	0.000
AREA_LN	Natural log of the study site area	-0.293	0.066	0.000
GCP50_LN	Natural log of Gross Cell Product within 50km radius	0.039	0.099	0.695
POP50_LN	Natural log of population within 50km radius of study site	0.238	0.154	0.125
RDS50_LN	Natural log of length of roads within 50km radius of study site	-0.035	0.107	0.743
HAN50_LN	Natural log of human appropriation of NPP within 50km radius of study site	-0.076	0.054	0.161
NPP50_LN	Natural log of NPP within 50km radius of study site	-0.379	0.287	0.189
COR50_LN	Natural log of area of coral reef within 50km radius of study site	-0.207	0.231	0.372
N		163		
Adjusted R^2		0.18		

8.8 Lakes and rivers

As stated above, value data has been standardised to a common unit as willingness to pay (WTP) per household per year for a change in water quality in US$ 2007 price levels. This is the dependent variable in the meta-analysis and estimated value function. Values that are originally reported as WTP per person have been multiplied by the average household size for the country in which the study is located. Values that are originally reported as WTP per other unit of time (e.g., day, month, lump sum) have been converted to annual values using appropriate multipliers or by calculating annualised values using time horizons and discount rates reported in the underlying study when available. We also included value estimate of willingness to accept (WTA) compensation for negative changes in water quality.

The explanatory variables used in the estimation of the value function include two measures of water quality. These are the initial level of water quality and the change in water quality described in the contingent valuation scenario. Following Van Houtven *et al.* (2007) we standardise the descriptions of water quality in the selected literature to a 10 point index. This index is based on the water quality ladder developed by Vaughan (1986), which translates descriptions of water quality in terms of associated suitability for a number of activities into a 10 point index. For example, water quality that is considered 'boatable' has a water quality index value of 2.5, 'fishable' water has a value of 5.1, and 'swimmable' water has an index value of 7.0.

The explanatory variables also include two binary variables indicating whether the valued water body is a lake or a river. The omitted category variable in this case represents other types of surface water bodies including wetlands and sea. Other explanatory variables are the gross cell product per capita within a 50km radius of the study site; the area of urban land use (km^2) within 20km radius of the study site; the area of lakes and rivers (km^2) within a 10km radius of the study site; and the accessibility index.

The dependent and explanatory variables are listed in Table 34 together with the definition and descriptive statistics for each variable.

Table 34 Dependent and explanatory variables.

Variable label	Variable definition	Mean	Median	Std. Deviation	Minimum	Maximum
VALUE	WTP per household per year (PPP US$ 2007)	130.09	78.33	160.51	1.33	985.32
VAL_LN	Natural log of WTP per household per year (PPP US$ 2007)	4.34	4.37	1.06	0.85	6.89
WQI_CHANGE	Change in 10 point water quality index	1.98	2.00	1.71	-5.00	6.25
WQI_BASE	Base water quality 10 point index value	3.55	3.00	1.52	0.75	8.00
WATS_RIVER	Dummy variable: 1 = river, 0 = other	0.43	0.00	0.50	0.00	1.00
WATS_LAKE	Dummy variable: 1 = lake, 0 = other	0.18	0.00	0.38	0.00	1.00
GCP50_LN	Natural log of Gross Cell Product within 50km radius	8.13	8.67	2.31	0.00	11.45
URB20_LN	Natural log of urban area within 20km radius	3.20	3.00	2.44	0.00	6.88
LAK10_LN	Natural log of area of lakes and rivers within 10km radius	1.50	1.10	1.54	0.00	5.33
ACCESS_INDX	Accessibility index	0.48	0.36	0.43	0.00	1.00

The value function is presented in Table 35. The estimated coefficients on the explanatory variables mostly have the expected signs and are all statistically significant at the 5% level, except for the base water quality variable. The negative signs on the binary variables indicating lakes and rivers show that WTP for water quality improvement is lower for these types of water bodies. The positive sign on the income variable (gross cell product per capita) indicates that WTP increases with income. The negative sign on the urban area variable indicates that WTP is lower when there is more urban land use in the vicinity of a water body. This may reflect preferences for natural areas and therefore a greater willingness to pay for water quality improvements in water bodies in non-urbanised areas. A similar effect is found with the accessibility index. The negative sign on the lakes and rivers area variable indicates that WTP is lower when there are more substitute water bodies in the vicinity of a specific water body. In other words, the public are willing to pay more to improve water quality in a water body if there are few other water bodies in the surrounding area.

Table 35 Lakes and rivers value function.

Variable name	Variable definition	Beta	Std. Error	Sig.
Constant		4.898	0.314	0.000
WQI_CHANGE	Change in 10 point water quality index	0.081	0.032	0.011
WQI_BASE	Base water quality 10 point index value	-0.046	0.041	0.262
WATS_RIVER	Dummy variable: 1 = river, 0 = other	-0.472	0.127	0.000
WATS_LAKE	Dummy variable: 1 = lake, 0 = other	-0.563	0.209	0.007
GCP50_LN	Natural log of Gross Cell Product within 50km radius	0.103	0.036	0.005
URB20_LN	Natural log of urban area within 20km radius	-0.106	0.030	0.000
LAK10_LN	Natural log of area of lakes and rivers within 10km radius	-0.099	0.037	0.008
ACCESS_INDX	Accessibility index	-0.416	0.204	0.042
N		388		
Adjusted R^2		0.17		

8.9 GIS analysis: upscaling values

As mentioned above in the introduction to Section 8, there is a fourth substantive GIS step: upscaling of spatial relationships resulting from the meta-regression analysis between ecosystem values and explanatory spatial variables to a global (or regional) scale. The outputs of IMAGE-GLOBIO are changes in the distribution of land cover within grid cells. The pertinent methodological question is as follows: if a patch changes in extent, what is the value of that change given the local spatial characteristics? There are five sub-steps:

1. Preparation and mapping of seven different non-overlapping biomes represented at patch level.
2. Construction of global datasets with selected variables, covering the spatial extent of all considered biomes.
3. Integration and analysis of IMAGE-GLOBIO modelling data resulting in change factors for all grid-cells concerning land-use change, infrastructure change, economic change and water quality change. Spatial transfer to full spatial extent of selected biomes.
4. Combination for each biome of all relevant spatial variables into one raster map.

5. Export to tables of all relevant variables and change factors per biome for statistical processing of value functions (outside GIS environment, using SPSS 16.0)

The results for the value of land-use change are set out in Section 12. Further details concerning the GIS processes and data sources are set out in Appendix II, and a descriptive 'script' for the value transfer exercise provided in Appendix III.

9 Cost estimates for change scenarios

9.1 Introduction

As noted earlier, the change scenarios modelled in PBL (2010) which form the basis for the analysis in Part III are not policies *per se*. Although some map well to 'policy levers', others do not. Notwithstanding this, evaluating the benefits arising from change scenarios is arguably interesting even without a policy lever (and thus without a cost estimate), in the same way that global assessments of Ecological Footprints can be informative and stimulate discussion[51]. We have however reviewed the extant literature to provide cost estimates for each change scenario in so far as this is possible. Each of the change scenarios is treated in turn below.

9.2 Agricultural productivity

This change scenario is concerned with closing the agricultural yield gap between the developed and developing world through investment in Agricultural Knowledge, Science and Technology (AKST), leading to an assumed 40% increase in crop productivity and a 20% increase in livestock productivity relative to the baseline.

Agricultural yields depend *inter alia* on access to production inputs, management of the natural environment and the adoption of techniques and technologies; all of these factors depend on market and institutional constraints (Neumann *et al*, 2009; Dreyfus *et al*, 2009) such as agricultural subsidies, institutional incentives, property rights, and land distribution (Morton *et al*, 2006; Bello, 2010). The extent to which different factors affect overall productivity has been analysed empirically (e.g. Alvarez and Grigera, 2005; Morris *et al*, 1997) but there is no consensus in this regard. This is turn implies that the efficacy of AKST policy interventions in terms of increasing agricultural productivity will likely vary on the basis of these constraints.

The change scenario for agricultural productivity is explicitly based upon an influential study assessing the future of agriculture, entitled Agriculture at a Crossroads (IAASTD, 2009). This study combines partial equilibrium (IMPACT) and computable general equilibrium (CGE) models (GTEM) to analyse alternative scenarios and their impact on agricultural yield to 2050. The study considers five factors as catalysts of a growth in agricultural yield: (1) investment in education in rural areas, particularly focusing on women; (2) investment in rural roads; (3) irrigation management; (4) policies propagating access to clean water; and (5) agricultural R&D.

The 'AKST high 2' scenario in IAASTD (2009) is estimated by the authors to cost circa US$30 billion per annum for total cumulative costs. This 'AKST high 2' scenario is used in PBL (2010). The question that is pertinent to our study is whether this cost estimate is realistic and defensible.

There is limited evidence in this regard. Schmidhuber *et al* (2009) provide an estimate of capital requirements needed for agriculture up to 2050 if developing countries are to meet FAO baseline projections (FAO, 2006). It is not possible to draw a like-for-like comparison

[51] See for instance the WWF Living Planet Report:
http://wwf.panda.org/about_our_earth/all_publications/living_planet_report/

between IAASTD (2009) and Schmidhuber *et al* (2009) as the outcomes for which costs are estimated differ. Notwithstanding this important caveat, the overall total estimate in the latter study is around US$5.2 trillion, a figure considerably higher than the IAASTD estimate.

The IAASTD (2009) figures might under-estimate costs owing to assumptions vis-à-vis policy implementation; there is evidence (e.g. Easterly, 2002; Rist, 2001) that 'big pushes' in terms of development aid has often not fulfilled the investment requirements of developing countries. This point links with the discussion concerning market and institutional constraints.

A further issue from a cost perspective with the intensification that often goes hand-in-hand with AKST is losses in agro-biodiversity as a consequence of pollution spillovers (Harris, 2006; Matson and Vitousek, 2006; UNEP, 2009). In this respect, the environmental impacts following the green revolution in Asia are particularly illustrative (Matson *et al*, 1997). Such losses are difficult to fully capture in bio-physical modelling. Some research findings indicate that considerable yield improvements can be made in some agricultural systems at little or even a positive environmental impact (IAASTD, 2009; Keating *et al*., 2010; Brussaard *et al*., 2010). The precise nature of productivity improvements (e.g. education; access to credit; increased inputs and intensification) and their interactions with agricultural systems and local social and environmental conditions will drive the extent of these impacts.

9.2.1 Summary

Notwithstanding the caveats discussed above, we use the cost estimates provided in IAASTD (2009). The figure of US$ 30 billion per annum is not used as we consider net costs, i.e. additional investment requirements rather than total cumulative investments. We assume that the profile of these investments is flat. As such the figure used for costs is US$ 14.5 billion per annum from 2000 to 2050. It is not possible to generate a cost estimate range *per se* without further primary research. Overall results for this option are presented in Table 36.

Table 36 Cumulated costs of closing the yield gap (Billion US$ 2007).

	DR=0%	DR=1%	DR=4%
Cost estimate for change scenario 1	725	568.3	311.5

Source: Based on Rosegrant *et al*. (2009) in IAASTD (2009)

9.3 Reducing post-harvest losses in the food chain

No quantitative monetary estimations for addressing supply-chain losses were found. We therefore only provide a brief generic analysis of potential costs. In developing countries, the following determinants of supply chain losses should be assessed: inadequate marketing systems; limited transportation facilities; lack of availability of required equipment; lack of information, education and training of farmers; and finally lack of facilities maintenance (Kader, 2005). A framework is provided in FAO (2003), set out in Figure 12. Note that this conceptual scheme does not necessarily match the biophysical modelling used in this report.

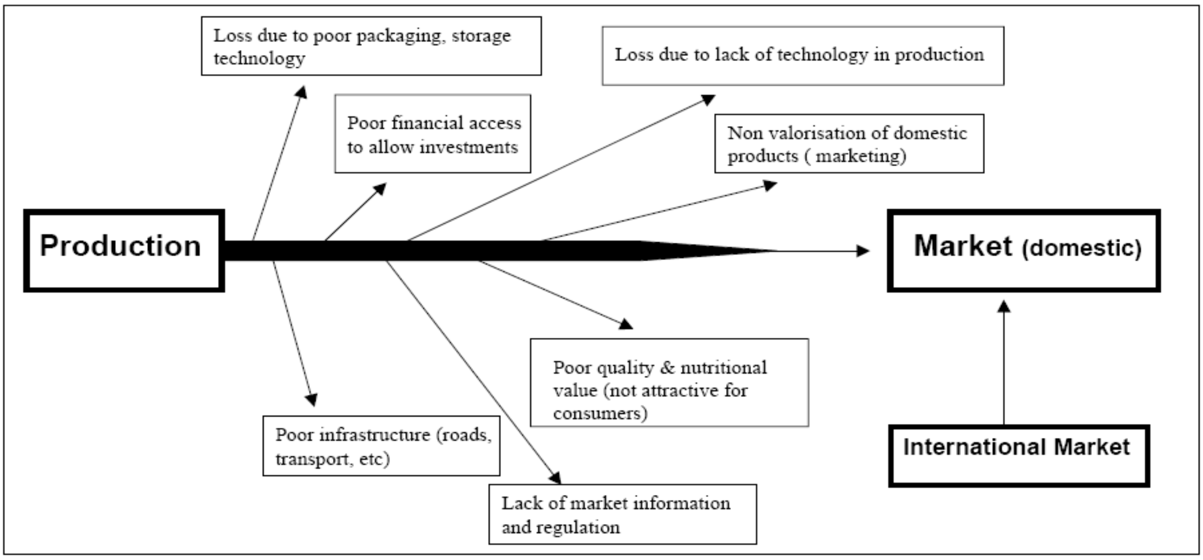

Figure 12 Determinants of post-harvest losses in developing countries.

Source: FAO, 2003

The extent to which specific policies can go about influencing these factors has not been assessed to our knowledge, therein not allowing costs to be estimated.

9.4 Forest management

This scenario option entails two elements: first, a transition from conventional logging (CL) practices to reduced impact logging (RIL)[52]; and second, an expansion of global plantation forestry in order to compensate for, and eventually surpass, the reduction in the supply of timber products that arises from the transition to RIL. No quantitative analysis or data were found to assess the costs of a wide-scale expansion of plantation forestry. Thereafter we focus solely on costs of the transition from conventional logging to a RIL regime. As such our cost analysis is partial.

The analysis of conventional logging versus RIL in this option scenario option also requires a discussion of illegal logging, which has been estimated to constitute around 15% of the global trade in timber products (Contreras-Hermosilla, 2002). Adinegoro (2001) goes as far as suggesting that comparing RIL to CL is flawed as illegal logging is so widespread that it represents the appropriate counter-factual to RIL in terms of cost-benefit assessment. However, estimating the costs of establishing and enforcing property rights structures to enact a shift from illegal logging is extremely difficult and complex, and the evidence base is very limited. As such we retain the focus on CL as opposed to illegal logging.

The constituent elements of the cost analysis in comparing CL with RIL are as follows: (1) investment costs; (2) financial returns from timber sales; and (3) transactions costs pertaining to regulatory enforcement. Each is treated in turn below.

(1) Several empirical studies have dealt with one-off and incremental investment costs of RIL compared to conventional logging (e.g. Applegate, 2002; Putz *et al*., 2000; Putz *et al*., 2005; Durst and Enters, 2001). Perhaps the most comprehensive analysis is Killmann *et al* (2001).

[52] PBL (2010) treat RIL and sustainable timber management (STM) as being equivalent, and we follow this approach in this study. However RIL and STM differ markedly from sustainable forest management (SFM).

Through an extensive literature review, Killmann *et al* (2001) find an overall (median) 43% higher cost for RIL compared to CL for this cost category. In general the evidence from the literature suggests that the initial investment requirements and incremental costs for RIL are invariably found to be higher than CL, although the magnitude of additional investment requirements depends on the characteristics of specific forest sites and on the economic context (Enters *et al*., 2001; Applegate *et al*., 2004; Siry *et al*., 2005).

(2) The evidence on financial returns from timber sales is less conclusive. The revenue stream from timber sales is likely to be lower for RIL in the short run (Pearce *et al*., 1999). On the other hand, unsustainable logging practices can be financially unprofitable in the medium or long term: first, continuous forest degradation can lead to a reduction in timber available for harvesting, particularly from second and third cut extractions (FAO, 1997; FAO, 2007); second, RIL practices can contribute significantly to increased efficiency in the harvesting process, for instance by reducing operational losses (Applegate *et al*., 2004; FAO, 2004; Samad *et al*., 2009). The optimal choice between CL and RIL depends on a number of factors *inter alia* the logger's private rate of time preference (i.e. discount rate) and the status of the forest vis-à-vis property rights (Contreras-Hermosilla, 1999; Pearce *et al*., 1999; Ostrom, 1990). A summary of the review of the literature on financial returns[53] is provided in Table 37.

Table 37 Summary of literature review on financial returns to conventional logging versus sustainable timber management (STM) and reduced impact logging RIL)

Study	Country	Change in profits: conventional logging compared to STM/RIL
1. Bann (1997)	Cambodia	+316%
2. Samad *et al*. (2009)	Malaysia	+88%
3. Kumari (1996)	Malaysia	+77%
4.Boscolo and Mendelsohn (1998)	Malaysia	+65%
5. Tay *et al*. (2001)	Malaysia	+56%
6. Kishor and Constantino (1993)	Costa Rica	+51%
7. Howard *et al*. (1996)	Bolivia	+27% to +67%
8. Dagang *et al*. (2001)	Malaysia	-9% to +30%
9. Holmes *et al*. (2002)	Brazil	-18%
10. Johns *et al*. (1996)	Brazil	- 29% to -34%
11. Boltz *et al*. (2001)	Brazil	-34%
12. Barreto *et al*. (1997)	Brazil	-49%

Of the 12 empirical studies covering five tropical countries, seven cases studies presented higher financial returns from CL compared to RIL, while four of these show higher financial returns for RIL management. (The range for Dagang *et al*., 2001 falls both below and above 0%, i.e. is equivocal.) The reason that some studies show lower costs for CL over RIL and *vice versa* can be explained by several factors. First, the time series used to assess financial returns vary, ranging from single-year returns to 40-year projections; shorter time series are likely to favour conventional logging. Second, the discount rates vary from 4% to 20%; a

[53] Note that the percentage difference in profits pertain only to financial costs and do not include other economic cost elements, such as the positive benefits from ecosystem conservation associated with RIL. This is to avoid double-counting as the latter are in part measured in the benefit estimation in our study.

higher discount rate would favour conventional logging. Finally, some analyses (e.g. (Enters *et al*., 2001; Applegate *et al*., 2004; Smith and Applegate, 2001) point to the fact that the financial returns of RIL are highly dependent on geographical locations: limited accessibility and/or particular geo-morphological and ecological conditions can impose a cost penalty on RIL implementation.

(3) There are other elements to the assessment of the costs to adopting RIL. Analyzing barriers to the adoption of RIL, Durst and Enters (2001) identify the logging industry's imperfect and incomplete information vis-à-vis RIL benefits, lack of tenure security, and ineffective regulation enforcements in tropical countries. These represent economically-relevant transactions costs but are difficult to estimate.

9.4.1 Global assessment of costs

The only global cost estimation found in our review is Robledo and Blaser (2007). Their findings, based on Whiteman (2006) and ITTO (1995), suggest that the costs of implementing sustainable forest management for all tropical forests in Kyoto non-Annex 1 tropical countries would be 8.3 billion US$ (2007). For non-Annex 1 countries with temperate climates, an additional 1 billion US$ (2007) is estimated. These figures are based on the postulate that exploited forest cover remains identical up to 2030. Yet, the nature of these costs are not specified, and is therefore (1) difficult to determine whether figures represent annual or aggregate costs (in the case of their study: for the period 2009-2030) and (2) whether opportunity costs are included. Finally, they represent figures for SFM and not RIL.

An alternative approach that could be adopted is to consider the Whiteman (2006) estimate, coupled with the 12 country-level studies cited above (Table 37):

(1) Whiteman (2006) suggests a mean figure of 1200 US$ per hectare per year and a range of 500 to 3000 US$ per hectare per year; these figures are expressed as 2006 US$.

(2) In 2007 US$, adjusted using PPP inflator, this is equivalent to a mean value of 1285 US$ per hectare per year, with a range of 514 to 3085 US$.

(3) Data on global hectares currently logged (excluding plantation forestry) were found in Robledo and Blaser (2007), based on and adjusted from ITTO (1995), FAO (2003) and MEA (2006). Robledo and Blaser (2007) estimate total logged tropical forests to represent about 350 million hectares.

(4) Given the data on per hectare mean values (and the associated estimate range) expressed in 2007 US$ and the data on global hectares, the mean value of global CL can be estimated at 437 billion 2007 US$, with a value range 175 billion US$ to 1 trillion 2007 US$.

(5) The value for Bann (1997) appears to be an outlier vis-à-vis the sample. If we remove this observation from the others, CL is on average 23.8% more profitable than STM/RIL.

(6) Using the value ranges estimated in (4) above, we can generate an indicative estimate of profit losses from RIL versus CL. In 2007 US$, this profit loss is estimated at around 23.8% of 437 billion 2007 US$, i.e. around 103.9 billion, with an uncertainty range from 41.6 billion to 237.7 billion 2007 US$.

Note that several caveats should be applied to this assessment. First, it relies entirely on one study (Whiteman, 2006) as this is the only global study that was found. Second, we implicitly assume that the 12 studies are representative globally. We accept that this is a heroic assumption, particularly as there are only five countries assessed across the 12 studies in the sample. Further, all four studies from Brazil provide a cost ratio of less than 1, i.e. RIL is more profitable than CL; these four studies are the only studies which unequivocally present this outcome.

Note that the assessment above pertains to profitability and profitability alone. It has not been possible to derive a global estimate for the other two elements of cost (one-off investment set-up costs and regulatory costs). There is also very limited evidence as to how significant these other two cost elements are relative to financial returns.

9.4.2 Summary

There is some evidence that there are net financial benefits (in terms of profits to the logger) from RIL as compared to CL. However the balance of evidence suggests that there are net financial costs. The per annum cost figures used in our study are 103.9 billion 2007 US$ (lower bound estimate 41.6 billion, upper bound 237.7 billion 2007 US$). A summary of total estimated costs at 0%, 1% and 4% discount rates is provided in Table 38, applying these per annum estimates from 2000 to 2050. As noted above our cost estimates are partial as we have no data on the costs of plantation establishment.

Table 38 Cumulated costs of RIL implementation (2000-2050) in billions 2007 US$

Discount rate	Best Estimate	Lower bound	Upper bound
0%	5,195	2,080	11,885
1%	4,073	1,631	9,317
4%	2,232	894	5,106

Source: Authors' calculations (see text)

9.5 Protected areas

Despite their numerous benefits, most notably in terms of positive externalities (Naidoo *et al*., 2007), the establishment of protected areas entails considerable costs; these costs are considered to be the source of both ecological under-representation and poor management (Bruner *et al*., 2004; Balmford *et al*., 2004; Galindo *et al*., 2005; Ruiz, 2005). Of these, opportunity costs in terms of foregone alternative use of land are the most critical (Faith and Walker, 1996; Ferraro, 2002). Global level estimations of costs are widely divergent and even contradictory (Pearce, 2007); they require considerable assumptions to be made and extensive modelling analysis (TEEB, 2009).

This option scenario refers to a precise percentage increase of 20% and 50% *for the earth's 83 eco-regions* rather than biomes; this is an important distinction as the extant cost literature does not in general estimate costs on an eco-region basis. The distribution of global eco-regions per biome is presented in Table 39 while the distribution of high priority Global-2000 eco-regions is presented in Table 40.

A typology of costs that is common to an estimate for expansion by eco-region or by biome is set out in Table 41.

We found seven studies on the global costs of conserving PAs with some variability with regards the types of costs estimated (Table 42). Values from these studies are converted (where possible) to 2007 US$ per hectare per year for the results to be comparable. Bruner *et al* (2004) estimate total management financial requirements at 13.8 billion per year (2007 US$ equivalent) for adequately managing and expanding current protected areas network to cover key unprotected species.

Table 39 Total area of global eco-regions per biome

Eco-regions	Total area (million km^2)
Tropical and sub-tropical moist broadleaf forest	19.78
Tropical and sub-tropical dry broadleaf forest	3.01
Tropical and sub-tropical coniferous forest	0.71
Temperate broadleaf and mixed forests	12.83
Temperate Conifer forests	4.09
Boreal forests/Taiga	15.13
Tropical and sub-tropical grasslands, savannas, and shrublands	20.18
Temperate grasslands, savannas and shrublands	10.10
Flooded grasslands and savannas	1.09
Montane grasslands and shrublands	5.19
Tundra	8.35
Mediterranean forests, woodlands and scrub	3.22
Deserts and Xeric shrublands	27.89
Mangroves	0.35

Source: Adapted from Jenkins and Joppa (2009)

Table 40 Distribution of Global-2000 Eco-regions

Geographical Realm	Continents/regions	Area ('000 km^2)
Australasia	Australia, New Zealand	5,194
Afro-tropic	Sub-Saharan Africa	9,904
Indo-Malaya	India, Southeast Asia	4,643
Neoarctic	North America, Greenland	6,735
Neotropic	South and Central America	11,931
Palearctic	Europe, Russia, China, Japan	13,369

Source: Naidoo and Iwamura (2007)

Table 41 A typology of costs for Protected Areas

Cost	Characteristics
Acquisition costs	Transfers of property rights, land rental, conservation easements
Management costs	Establishing and maintaining the network of areas
Transaction costs	Negotiations with landholders/stakeholders, transfers of property rights
Damage costs	Damage to crops/livestock in adjacent areas (provoked by conservation of wild animals, etc)
Opportunity costs	Highest-value alternative use of land of a given area.

Source: adapted from Naidoo *et al*. (2007)

Table 42 Global assessments of the costs of Protected Areas

Authors/study	Scope of study	Type of costs	Type of assessment	Mean cost estimation (2007 US Dollars)
Balmford *et al*. (2004)	Global	Management Costs	Costs for adequately financing current PA system	8.2 billion/year
Bruner *et al*. (2004)	Global	Management Costs	Costs for adequately financing current PA system, and its expansion to high priority sites	13.8 billion/year
James *et al*. (1999)	Global	Management and opportunity costs	Requirements for protecting and expanding PA system to represent 20% of global biomes	19.1/ha/year
James *et al*. (2001)	Global	Opportunity, management and acquisition costs	Requirements for protecting and expanding PA system to represent 20% of global biomes	20.2 –21.4/ha/year (management costs 3.8-5/ha/year)
Lewandrowski *et al*. (1999)	Global	Opportunity Costs	Setting aside 5%, 10% and 15% of global terrestrial area	98/ha/year
Naidoo and Iwamura (2007)	Global	Opportunity costs	Estimation of cost effectiveness of protection juxtaposing species richness and agricultural returns	58/ha/year
World Bank (2002)	Developing countries	Management and opportunity costs	Additional protection of 800 million hectares of land in developing countries	93/ha/year (2000) (management costs 10/ha/year)

Source: Pearce (2007) and authors' synthesis

James *et al*. (1999 and 2001) estimate values on a per hectare basis. As such, adequately managing and expanding the protected area network to represent 20% of ecological biomes would cost approximately 18 to 27.5 billion US$ per year (or 23 to 34 2007 US$). Their estimation includes management costs obtained through surveys of currently protected areas, acquisition costs for new land areas and finally compensation for foregone land use. The latter is nonetheless based on an estimation of the value of land (in terms of rent) rather than as a flow of foregone benefits from alternative land-use.

More methodologically consistent for estimating opportunity costs are the analyses of Lewandrowski *et al*. (1999) and Naidoo and Iwamura (2007). Lewandrowski *et al*. (1999) combine GIS modelling with Computable General Equilibrium modelling, the former for identifying potential protected area status, the latter in order to estimate global returns from agricultural production in an input-output fashion. In this way the authors estimate potential foregone revenues for protected areas covering 5%, 10% and 15% of world biomes. Their approach is however more focused on human geography territorial criteria (rather than ecological). Naidoo and Iwamura (2007) consider global agricultural returns using a GIS model with a view to identifying cost-effective conservation strategies. Their model notably allows the identification of any overlap between relatively low agricultural returns and high biodiversity levels. While not dealing with protected areas *per se* their figures are relevant to our analysis, since the levels of agricultural returns per bio-geographic realm can be considered as opportunity costs of conservation.

Comparing opportunity cost results across the studies, it appears that there are significant differences in mean estimates, ranging from more than 98US$ per hectare per year for management costs alone to around 21US$ for opportunity and management costs *combined*. However, there are reasons for this variation in estimates:

(1) James *et al*. (1999, 2001) deal mostly with opportunity costs in developing countries, considering that only these are relevant for compensation of foregone land uses. This explains the low mean estimate, since the higher land returns in Lewandrowski *et al* (1999) and Naidoo and Iwamura (2007) estimations are located in developed countries – with the exception of parts of the developing world in Asia.

(2) Moreover, the divergence in the estimates in Lewandrowski *et al*. (1999) and Naidoo and Iwamura (2007) can partly be explained by the fact that the former does not include parts of the world that have low agricultural returns (such as Greenland or Siberia) whereas the latter does.

9.5.1 Using the data to estimate a composite cost

Previous studies calculating protected areas expansion costs have done so by assuming an area protected per biome or geographic assemblies, not per ecological region; the scenario option requires a cost estimate per eco-region, as defined by the CBD (Coad *et al*, 2009) which in turn entails inevitable complexities. We carry out two sets of calculations: first, we estimate costs for protecting 20% and 50% of the earth's 834 eco-regions. Second, we estimate costs for reaching 20% and 50% protection coverage of the Global-200 Eco-regions, defined as 'priority sites' by the WWF (Coad *et al*, 2009). Data on the total coverage (km^2) of eco-regions were obtained from Coad *et al* (2009) and Naidoo and Iwamura (2007) (see Table 40 above). Percentages of current protection levels are only available for so-called 'priority eco-regions' as defined by the WWF (2009). While the World Database on Protected Areas (WDPA[54]) provides specific regional data, and maps protected areas, we did not find a specific percentage for eco-regions. The change scenario developed in PBL (2010) assumes current protection to be 10% of all eco-regions, *viz*. not just priority eco-regions, and this figure is used as the counter-factual.

We therefore estimate the total costs of achieving 20% and 50% coverage. We assume that cost increments are linear, i.e. the costs of protecting 20% are double the costs of protecting 10%. This assumption of linearity is applied in the absence of clear evidence to support an alternative formulation: (1) management and opportunity costs are ongoing irrespective of any incremental protection; (2) opportunity costs can rise when additional area is added to

[54] See: http://www.wdpa.org/Statistics.aspx

the protected network whereas there is some evidence of countervailing scale economies for management costs (James *et al*., 2001; Balmford *et al*., 2004).

In our estimation, the total hectares of eco-regions are scaled up per bio-geographic realm (see Table 40). For both estimations we provide three different cost estimation scenarios based on the availability of data, different cost estimations on a per hectare basis (or per km^2), and on different assumptions applied. Table 43 summarises the outcomes across the three scenarios for an expansion across all global eco-regions while Table 44 summarizes our findings for an expansion of protected areas in Global-200 eco-regions. A summary of the constituent elements of Scenarios 1-3 are given below:

Scenario 1: Data from James *et al*. (1999 and 2001) is used which estimates combined management and opportunity costs but excludes land acquisition costs.

Scenario 2: Data from Naidoo and Inamura (2007) and Lewandrowski *et al*. (1999) are used to estimate opportunity costs in foregone land returns; management costs estimates inputted are based on James *et al* (2001) and Balmford *et al* (2004) for this scenario.

Scenario 3: As per Scenario 2 but opportunity costs for developed world countries are excluded from the calculations.

As is the case for most global cost estimations, for an expansion to 20% results show a wide range from 30.9 billion 2007 US$ per annum for Scenario 1 to 132.9 billion 2007 US$ for Scenario 2 for total costs, and 15.4 to 66.4 billion 2007 US$ as marginal costs (see Table 43).

Table 43 Estimated costs of expanding protection across global eco-regions (billions 2007 US$).

	Scenario 1		Scenario 2		Scenario 3	
	20%	50%	20%	50%	20%	50%
Costs/annum	30.9	77.1	132.9	332.4	71.3	178.2
Marginal annual costs (assuming 10% current protection)	15.4	61.7	66.4	265.9	35.6	106.9

Source: Authors calculations (see text)

In terms of expanding the protected area network to a 20% coverage of Global-200 priority eco-regions, results are presented in Table 44, showing a range from 17.1 billion to 69.0 billion 2007 US$ per year for 20%.

Table 44 Estimated costs of expanding protection across Global 200 eco-regions (billions 2007 US$).

	Scenario 1		Scenario 2		Scenario 3	
	20%	50%	20%	50%	20%	50%
Costs/annum	17.1	42.6	69.0	172.5	22.0	55.0
Marginal annual costs (assuming 10% current protection)	8.5	34.1	34.5	138.0	11.0	44.0

Source: Authors' calculations (see text)

9.5.2 Methodological approaches, limits and developing a ‘best-guess’ estimate

We consider Scenario 1 to be our ‘best guess estimate’ for the reason set out in this section.

The comparison between Scenarios 2 and 3 requires the determination of whether or not to include opportunity costs in developed world nations. From a conceptual point of view, the estimates for global returns from agricultural land use in Naidoo and Inamura (2007) and Lewandrowski *et al*. (1999) do not consider shadow values, in this case agricultural subsidies. As such, the net opportunity cost of conservation should be annual returns from alternative agricultural land use minus potential subsidies (Wesley and Peterson, 2009). This is particularly (but not exclusively) an issue in the developed world. Payments for protected area establishment vis-à-vis opportunity costs are also likely to be restricted in many cases to land-owners in developing world countries; in this respect, the EC estimates management costs alone when assessing financing of the EU Natura-2000 network (EC, 2006). For these reasons we would support estimates from Scenario 3 over those derived from Scenario 2. However we retain Scenario 2 as an ‘upper bound’ estimate.

Our mid-level (Scenario 3) and upper bound (Scenario 2) estimates are largely based on the FARM (combined CGE and GIS) model results used by Lewandrowski *et al* (1999) to compute opportunity costs of conservation, and a similar application in Naidoo and Iwamura (2007). In these studies, opportunity costs are considered to be highest potential feasible per year returns from agricultural land for a given geographic area. As Lewandrowski *et al.* (1999) note, one critical assumption made using the FARM model is that all global land can potentially be of economic use. The authors observe that this assumption is a necessary modelling simplification: clearly, protected areas are often located in areas where there is low (or even non-existent) human productive exploitation. As such, opportunity costs are systematically over-estimated in their model, and thus in our mid and upper bound estimates. The approach combining CGE and GIS is clearly defensible (and arguably potentially the most robust) but of course has its own methodological limitations.

Our rationale for choosing the lower bound (Scenario 1) estimate as our ‘best guess’ is partially based on these methodological concerns, but is also determined by a stream of potential benefits that is missing, *viz*. eco-tourism returns (Gossling, 1999; Naidoo *et al*., 2007; Carret and Lover, 2003) that are not accounted for in the cost estimates. Further, other influential global cost estimates such as Bruner *et al.* (2002) and Balmford *et al.* (2004)*,* often cited in conservation literature (Bruner *et al.*, 2008) as the costs of maintaining and expanding the PA system, focus on *acquisition* and management costs. James *et al.* (1999 and 2001) is used as the basis for the Scenario 1 estimate; the authors consider compensation mechanisms in the developing world derived from land prices, a proxy for acquisition costs. Thus Scenario 1 is consistent with some influential extant estimates (e.g. IUCN, 2003; TEEB, 2010).

9.5.3 Summary

The most defensible estimate for costs for expansion to 20% and 50% from the 10% baseline is 15.4 billion and 61.7 billion 2007 US$ per annum respectively. The upper bound estimate for costs for expansion to 20% and 50% from the 10% baseline is 66.4 billion and 265.5 billion 2007 US$ per annum respectively. The latter assumes compensation payments to land-owners in the developed world, and compensation that includes market distortions in the form of agricultural subsidies (Scenario 2).

These values represent (1) one-off acquisition or on-going opportunity costs and (2) on-going management costs implied by the expansion of the protected area network. In summary, we assume one-off acquisition costs at a conservative level of 2.2 billion US$ 2007 occurring during the first implementation year; and ongoing management and opportunity costs of 15.4 (61.7) and 66.4 (265.9) billion US$ 2007 per year respectively for scenarios 1 and 2 (our 'best guess' and 'upper bound' estimates) from 2000 to 2030 for a 20% (50%) coverage of global eco-regions. Cumulative results discounted at 0%, 1% and 4% are presented in Table 45.

Table 45 Cumulative costs of Protected Areas expansion (2000-2030) in billions 2007 US$.

Discount rate (DR)	'Best-guess' (Scenario 1)		Upper limit (Scenario 2)	
	20%	50%	20%	50%
DR=0%	465	1854	1996	7973
DR=1%	400	1595	1717	6859
DR=4%	269	1069	1151	4597

9.6 Reduced deforestation

As per the 'forest management' option sceanrio, there are three principal costs of implementing the 'reduced deforestation' option scenario, a variant of REDD (Pagiola and Bosquet, 2009): (1) opportunity costs; (2) management and monitoring costs; and (3) transaction costs. The most critical cost category is likely to be opportunity costs in terms of foregone revenues either from forest conversion to agricultural land or possible returns from forest exploitation through logging (Chomitz, 2006). Accurately estimating foregone revenues on a global scale can be complex since it implies a generalization of possible alternative land returns to vast geographical areas (Barreto *et al*., 1998), and the need to predict changes over time. Policy implementation of REDD are highly dependent on the institutional factors prevailing in developing world countries, as well as on equity in management of payment transfers (Gomez, 2009). These elements might well be determinants of the additionality and efficiency of the scheme (Karousakis, 2009).

Adapting from Boucher (2008), methodologies to address costs might be categorized as follows:

(1) Empirical studies that have a local, regional or national scope, which are based on specific bottom-up calculations of REDD implementation or simulation of REDD implementation.

(2) 'Hybrid studies' which partly use bottom-up calculations is order to model global costs notably by using, explicitly or implicitly, benefit transfer methodology, and applying specific assumptions to variables[55]. The estimations used by the Stern review, for example, are in this category (Stern, 2006; Grieg-Gran, 2006).

[55] Assumptions are related to the extrapolation of carbon density levels (per hectare) to other sites (from local to regional extrapolation): see Boucher (2008) pps. 15-17.

(3) Global modelling through partial equilibrium dynamic models. While they may have global coverage, their scope is almost uniquely the assessment of opportunity costs with additional costs generally added to estimated opportunity costs by using the results of empirical studies. The Eliash review (2008) can notably be classified in this category. A synopsis of studies that have attempted to estimate costs for REDD are provided in Table 46.

Table 46 Studies that have estimated global costs for REDD.

Study	Study type	Costs assessed	Cost/year (bn US$)	Time horizon	Percentage reduction in GHGs
Eliash (2008)	Global model	OC+MC+TC	17 to 33	2030	46%
Kinderman *et al*. (2008)	Global model	OC+MC+TC	17 to 28	2030	46%
Grieg-Gran/ Stern (2006)	Hybrid approach	OC+MC	5 + 0.5	2030	50%
Grieg-Gran, (2008)	Hybrid approach	OC+MC	6.8 to 8 +0.5	2030	50%
Strassburg *et al*. (2008)	Empirically derived	OC	29.6	n/a	100%
Boucher *et al*. (2008)	Global (partial equilibrium)	OC+MC+TC	14 to 48.5	2030	20% to 80%
Blaser *et al*. (2007)	Global	OC	12.2 (min)	2030	100%

OC - Opportunity costs; MC - Management costs; TC - Transaction costs

Results range by an order of magnitude, but there are various factors that might explain this variability. First, the studies do not estimate the same percentage reduction in greenhouse gas emissions from deforestation and forest degradation. Second, different categories of cost are assessed. Third, different assumptions are made vis-à-vis returns to the land use alternatives and therefore to opportunity costs. Grieg-Gran (2006) for instance only considers returns from conversion of forests to agriculture: other forms of forestry activity are not considered, and she states that using higher (and plausible) returns for agriculture could increase opportunity costs to 26 billion per year using her data (Grieg-Gran, 2006; 2008). Kinderman *et al*. (2008), a background paper for the Eliash (2008) review, consider not only land returns from agriculture but also from timber activities, and include foregone flows of revenues and also foregone rents to land. In short, the assumptions applied and methodologies used are divergent across the studies in Table 46.

One reason to present the study type (empirically derived/hybrid/global) is that analysis by Boucher (2008) suggests that this is influential in determining cost estimates (see Table 47).

Table 47 Opportunity costs differentials depending on methodological approach.

Approach	Mean opportunity cost (US$/t$CO_2$eq)
Empirical / Regional	2.51 (Range: 0.84 to 4.18)
Hybrid	5.52 (Range: 2.76 to 8.28)
Global partial equilibrium models	11.26 (Range: 6.77 to 17.86)

Source: adapted from Boucher (2008)

Although the sample size of studies is small, it is noteworthy that high-point in the estimate range for empirical studies (4.18) is less than the lower bound estimate for global partial equilibrium models (6.77).

9.6.1 Summary

All the estimates presented in Table 46 have been peer reviewed. There is no definitive ‘correct’ approach vis-à-vis methodology, assumptions, or data sources. In this sense choosing an estimate from any single, any combination or all the studies cited in Table 46 would be defensible. Perhaps the most significant estimate with regards the policy perspective is Grieg-Gran/Stern (2006). However, since Grieg-Gran (2008) is an update we would recommend using this estimate; it would be a lower bound estimate (taking the mid-point in the range 6.8 to 8 billion US$ plus 0.5 billion US$). The choice of higher-bound estimate is somewhat arbitrary, but Kinderman *et al* (2008) might be picked for the following reasons: (1) it is linked with the Eliash (2008) review and produces very similar estimates; (2) the percentage reduction is similar to the lower bound estimate from Grieg-Gran (i.e. 46% versus 50%); and (3) it covers all three cost categories.

The lower bound estimate is 7.9 billion US$ per annum and an upper bound estimate is 22.5 billion US$ per annum. (Both values are mid-points in the respective ranges.) Both estimates take into account the evolution of opportunity costs up to 2030, following possible increases in returns from alternative land use. Hence, non-linearity can be considered as endogenous to these estimations. Two additional non-linear elements were taken into account when calculating aggregate costs:

(1) Grieg-Gran (2008) provides incremental management costs as additional area is included in the network. Thus management costs represent 50 million US$ in the first year of implementation, and increase up to 500 million US$ when full implementation takes place. This cost trajectory has been taken into account when using the Grieg-Gran (2008) figure.

(2) Eliash (2008) proposes additional one-off implementation costs which occur during the first four implementation years (representing globally an additional 4 billion US$ over 5 years). This element is also taken into account.

In summary, to assess non-linearities in costs, we analytically separated (1) our lower bound estimate based on Grieg-Gran (2008) with (2) our higher bound estimate based on Eliash (2009) for the sake or remaining faithful to the cost quantification of the respective analyses.

(1) Following Grieg-Gran (2008) for our lower bound estimate, initial management and transaction costs represent 50 million US$ 2007 in the first year (2001), 100 million US$ 2007 in the second year, and reach 500 million US$ 2007 in year 10 (2010), after which the 500 million US$ 2007 figure is ongoing from 2010 to 2030. Opportunity costs are assessed as ongoing by Grieg-Gran (2008), and we thus assume costs of 7.9 billion US$ per annum from 2000 to 2030.

(2) Following Eliash, initial implementation costs were assumed to represent 4 billion 2007 US$ for the period 2000-2005 (evenly distributed in the first five years i.e, 800 million 2007 US$ per year). Ongoing costs are assumed to be of 22.5 billion US$ per annum for the period 2000-2030.

The costs discounted at 0%, 1% and 4% are presented in Table 48. These estimates pertain to the total costs and do not account for distributional impacts, i.e. who bears the costs.

Table 48 Cumulative costs of REDD (2000-2030) in billions 2007 US$..

Discount rate (DR)	Lower bound	Upper bound
DR=0%	248.3	679.0
DR=1%	213.7	584.6
DR=4%	142.2	392.6

Source: authors' calculations (see text)

9.7 Mitigating climate change with bio-energy

The change scenario is mainly based upon the prospective commercialisation of 'second-generation' bio-energy (although it also models sugarcane ethanol which is a 'first generation' bio-fuel), the costs of which cannot be estimated reliably since they remain at a developmental stage (IEA, 2010; OECD, 2008; Peters and Thielmann, 2007). It is expected they will be commercially available on a large scale after 2020 although there is considerable uncertainty in this respect (Doornbosch and Steenblick, 2008). We consider the direct and indirect costs of first-generation bio-energy policy and assess the validity of transferring these estimates.

Both first and second generation bio-energy sources are currently uncompetitive in comparison to fossil fuels (see Peters and Thielmann, 2007 for price differentials). There are several indirect costs associated with bio-fuel expansion. First, there are economic costs leading to changes in employment: the balance of evidence suggests a neutral impact for first generation bio-fuels (De Santi *et al*., 2008; OECD, 2008). Second, current policies have raised concerns related to food security. While additional income earned by farmers, notably in the developing world, can constitute a benefit, exposure to rising food prices (owing to the substitution of land previously used for food production to bio-fuel production) can be classified as a welfare cost, particularly for food importing and/or less developed countries (Msangui *et al*., 2010). Third, there can be detrimental environmental effects on habitats. We do not explore these latter two types of indirect cost so as to avoid double-counting, as land-use changes and (to an extent) habitat degradation are captured in the IMAGE-GLOBIO assessment.

We do consider direct costs in this assessment, which are the direct subsidies paid to support bio-fuel production given its current lack of price competitiveness vis-à-vis fossil fuels. Three broad types of supporting policies have been applied to are directly or indirectly subsidize bio-energy: (1) direct subsidies to production; (2) fuel tax exemptions; and (3) blending quotas (Peters and Thielmann, 2007; Steenblick, 2008).

Using data from the International Institute for Sustainable Development (IISD) (IISD, 2008; Steenblick, 2008; Jung *et al*, 2010; Koplow and Track, 2007; Laan *et al*, 2009; Pio-Lopez *et al*, 2008; Quirke *et al*, 2008; Steenblick *et al*, 2008; Singh *et al*, 2008) shows that the total *recorded* costs of current bio-fuel policies could represent 16.7 billion US$ 2007 per year, with 16.4 billion US$ representing only OECD (Table 49). (Note that the estimates in Table 49 are not cumulative; they refer to different regions and different time periods.)

Table 49 Cost of bio-fuel support policies for selected countries.

Selected countries	Support (US$/year)	Expected support (US$/year)
China	118 m	2020: 1.2 bn
Australia	289 m	n/a
EU	2.8 bn	n/a
Indonesia	200 m	n/a
Malaysia	16 m	2012: 122 m
Canada	300 m	n/a
United States of America	13 bn	2014: 16 bn
Switzerland	11 m	2017: 100 m
Total OECD	16.4 bn	2015-2017: 27 to 31 bn
Total	**16.7 bn**	**28.7 to 32.7 bn**

Sources: Adapted from IISD (2008); Steenblick (2008); Jung *et al*. (2010); Koplow and Track (2007); Laan *et al*. (2009); Pio-Lopez *et al*. (2008); Quirke *et al*. (2008); Steenblick *et al*. (2008); Singh *et al*. (2008).

It is estimated that this figure will reach 27 to 31 billion US$ in 2015 (OECD, 2008; Steeblick, 2008). A study conducted on behalf of the EC (De Santi *et al*., 2008) estimated the total costs (across the duration of the policy) of the EU bio-fuel target (4.5% of total fuel) at 56.7 billion.

9.7.1 Summary

As a lower bound estimate, the sum of direct support payments set out in Table 49 is used in our study, i.e. US$16.7 billion per annum. Given the time-frame of our study (to 2050) the direct support-estimate derived from OECD (2008) and Steeblick (2008) is also used as an upper bound estimate, i.e. 28.7 to 32.7 billion US$ with a mid-point value of 30.7 billion US$ per annum.

It is worth mentioning that this figure represents solely the direct costs of bio-fuel support policies, i.e. it does not take into account possible indirect costs. Further the costs of bio-fuel policies for key countries, such as Brazil, are not assessed due to lack of data. Hence, as already mentioned, these represent solely global recorded costs. Finally, additional direct costs, such as public R&D spending, are not assessed due to limited data – albeit the fact these could be significant (OECD, 2008; OECD, 2009).

In order to estimate cumulative costs, it is appropriate to use the 16.7 billion figure up to 2017, after which Steenblick (2008) and OECD (2008) consider that expected policy support is likely to reach approximately 30.7 billion US$ per year (minimum value 28.7 billion and maximum value 32.7 billion). Due to lack of additional data, we thus postulate that costs remain stable from 2017 onwards – which could clearly underestimate actual costs. Using these figures, cumulative discounted costs (at 0%, 1% and 4%) are presented in Table 50.

Table 50 Cumulative costs of bio-energy expansion in billions 2007 US$.

Discount rate (DR)	Cumulative costs (2000-2050)
DR=0%	1297.5
DR=1%	985.9
DR=4%	489.5

9.8 Global dietary patterns

The livestock sector currently represents around 40% of total agricultural turnover and around 1.5% of global GDP (FAO, 2006). A transformation of production patterns (partial or complete) from the livestock sector to agricultural production of crops and vegetables would certainly imply transitional costs. In turn, the magnitude of these costs depends in part on institutional capacities. The authoritative analysis of FAO (2006) entitled *Livestock's Long Shadow* notably estimates that 70% of the world's extremely poor (income of less than US$1 per day) depend on livestock for their livelihoods. Hence livestock is also a significant capital stock for the world's poor. A failure to finance this kind of transition could also have adverse health impacts associated with reduced (and insufficient) levels of protein and minerals intake (FAO, 2006).

Public health education has attempted for years to modify dietary patterns for the purposes of improving human health outcomes. Given the desirability of these improved health outcomes many alternative policies have been considered and applied but none can be reliably categorised as ‘likely’ to achieve such a dietary transformation. Thus no cost estimate is available for this option scenario.

9.9 Global agricultural trade

There is a lack of consensus as to whether trade liberalization has net costs or net benefits as a development strategy (Krugman, 1996). On the one hand, the findings of computable general equilibrium (CGE) models endorsed by some multilateral institutions (see for instance Stiglitz, 2003) imply that there would be virtually zero net costs in a global agricultural liberalization (Anderson *et al*, 2006). On the other hand, there have been various criticisms of these models vis-à-vis their empirical validity (e.g. Ackerman and Gallagher, 2008; Taylor *et al*, 2006; Stiglitz and Charlton, 2004). Other CGE models using different assumptions derive different conclusions (Polaski, 2006; Bouet *et al*., 2005; Francois *et al*., 2006). Moving away from CGE modelling, the institutional economics literature assesses possible dynamic development costs and gains, i.e. those costs and benefits incurred in the transition period moving from one state to another (see for instance Rodrick, 2002 and Taylor *et al*., 2006). We discuss the various positions below so as to determine what the possible and plausible outcomes of agricultural trade liberalization might be in terms of costs.

9.9.1 ‘Conventional’ computable general equilibrium (CGE) models

Two of the most prominent CGE models are the GTAP and LINKAGE models. These are recurrently used by multilateral institutions such as the World Bank and the WTO in order to make the case for a reduction of trade barriers in agricultural commodities. The overall results using these models and assessing the potential impacts of complete liberalization by 2015 are presented in Table 51.

Table 51 Positive economic impacts of complete multilateral trade liberalization using GTAP and LINKAGE models.

Scenario	GTAP,2005 (US$ bn)	LINKAGE, 2006 (US$ bn)
Total net welfare impacts	85	297
Net welfare impacts for developing countries	22	90
Net welfare impacts from agricultural trade liberalization	56	182
Net welfare impacts from agricultural liberalization for developing countries	12	54

Source: adapted from Hertel and Keney (2005) and Anderson *et al*. (2006)

The results from the GTAP and LINKAGE models are that trade liberalization has *net benefits* (as opposed to net costs). Although benefits are unevenly distributed (the majority of them accruing to developed countries as a consequence of consumer welfare effects) it should be noted that both developed world *and* developing world countries benefit from trade liberalization under the two models.

The different magnitude of net benefits between the two studies is essentially explained by the fact the assumptions applied: Anderson *et al.* (2006) in the LINKAGE model introduce an exogenous technological gain assumption which propagates benefits, although applying this assumption has been criticized (see Daudin, 2003). Irrespective of whether the higher LINKAGE or the lower GTAP estimate is used, the modelled welfare benefits are relatively small vis-à-vis percentage of global GDP or expressed on a per capita basis. This has led to the assertion that there are 'shrinking gains' from agricultural liberalization, particularly for developing countries (Ackerman and Gallagher, 2008; RIS, 2005). But *they remain as net gains nonetheless*, and this is of significance to the analysis in our study.

Whether the zero-net cost perspective provided by these CGE models is accepted depends on the validity of the LINKAGE and GTAP models. There have been various criticisms in this regard:

(1) CGE models describe a situation of automatically passing from one 'steady state' condition of the economy to another. They do not incorporate transition and adjustment costs associated with moving from one equilibrium point to another (Stiglitz and Charlton, 2004)[56]. Two potential transition costs are particularly critical. First, the employment effects of liberalization are of major concern, especially for developing countries (Ackerman and Gallagher, 2008). The models assume fixed full employment, perfect labour mobility, and perfect substitution between factors of production[57]. If these conditions do not apply – and it is clear for instance that unemployment does persist – then trade liberalization may simply move workers from low productivity sectors into unemployment rather than into higher productivity sectors. Second, the loss of fiscal revenue (foregone tax revenue) from the elimination of tariffs is categorized as a transition cost. The capacity to replace this foregone revenue stream is lower in the developing world compared to the developed world (Taylor, 2006) as the infrastructure to collect taxes is generally better developed. In a context where public infrastructural investment is, among others, critical for development policy (Chang,

[56] There may be costs but also benefits from this transition; see Sachs and Warner (1995) for a discussion.
[57] Although these assumptions do not apply for all versions of GTAP, they do apply for the studies assessing global agricultural liberalisation mentioned above, as does Anderson *et al*. (2006).

2002) this effect could be categorized as an opportunity cost (Rodrick, 2002; RIS, 2005; Sapir, 2007).

Neither the employment costs nor the opportunity costs of fiscal revenues are calculated as costs in the LINKAGE and GTAP cost estimates.

(2) GTAP and LINKAGE models are based on conventional trade theory; specialization in any sector, say primary, is virtually identical in terms of development and welfare potential to specialization in another sector, say manufacturing or services. This is highly contestable, first in the light of economic history (Chang, 2002) and empirical evidence (Wise, 2008), and second with regards developments in trade theory (Krugman, 1987; Krugman, 1991; Ackerman and Gallagher, 2008).

(3) Stiglitz and Charlton (2004) analyze what they term 'exposure cost to external volatility'. In CGE models, agricultural trade liberalization is generally forecast to spur an increase of agricultural commodities prices in the short and medium term (Arnold, 2005). While this is not necessarily problematic for countries projected to be net exporters, it is for developing countries projected to be net importers, such as most of sub-Saharan African economies (Arnold, 2005; Koning and Andersen, 2007).

This could raise issues of food security and represent a significant cost in terms of development and poverty reduction, but a cost that is not measured in the GTAP and LINKAGE models.

9.9.2 Alternative quantification of the costs of trade liberalization

Four studies present different methodologies and conclusions than Hertel and Keney (2005) and Anderson *et al* (2006): Polaski (2006); Brown *et al*. (2002); Bouet *et al*. (2005); and Francois *et al*. (2005). However in each of these four cases the models assess the impacts of a 'Doha' (partial liberalization) scenario rather than full liberalization, so they cannot be compared like-for-like with the GTAP and LINKAGE models. While assuming an instantaneous, complete liberalization of agricultural commodities provides an interesting outcome, this assumption makes little sense from a policy perspective. In light of past GATT and WTO agreements it is difficult to foresee complete agricultural liberalization in one step, without the intermediate step of partial liberalisation.

Polaski (2006) uses the Carnegie CGE model and presents potential costs of 63 million US$ incurred by developing world countries. These are notably linked to employment effects of liberalization. Brown *et al* (2002) develop the BDS model using new trade theory assumptions with increasing returns to scale for secondary and tertiary sectors. They find that global costs of a 33% reduction of agricultural protection are circa 8 billion US$. Bouet *et al*. (2005) and Francois *et al*. (2005) use the CEPII model, which also presents imperfect competition assumptions, and assess possible effects arising from the erosion of preferential trade agreements (Bureau *et al*, 2005). Bouet *et al* (2005) notably find a welfare loss of 2 billion US$ for developing world countries, although global welfare change is found to be positive.

Overall, these studies show that agricultural liberalization impacts are uneven among developing countries, presenting net welfare costs for some countries. Further, the BDS model used by Brown *et al*. (2005) is unique in assessing net global welfare costs for agricultural liberalization, although the methodology applied has been criticized (see Ackerman and Gallagher, 2008). Brown *et al*. (2005) is the only analysis across the four studies cited above that produces a net global welfare loss – *the other three show net benefits*.

9.9.3 Summary

We use the value of $0 net costs associated with trade liberalization as the balance of evidence implies that, on a global basis, this is appropriate. However, considering evidence from models based on partial liberalization as well as some of the limitations of the GTAP and LINKAGE models, caution should be applied in using this $0 cost value. Net costs *are* likely to be incurred by developing world nations. These include (1) the costs of under-employment, (2) opportunity costs arising from decreasing fiscal revenue from the elimination of tariffs, (3) the long run effects of potentially locking developing world countries into producing primary commodities, and (4) increased risks in terms of food security as food prices rise in the short and medium term (for a synthesis see Cordoba and Laird, 2006). These costs should be juxtaposed against the net benefit estimates to 2050 produced by GTAP ($85 billion) and LINKAGE ($297 billion). However, quantified estimates are not available for these four cost elements.

9.10 Summary of change scenario cost estimates

Table 52 provides an overall synopsis of the cost analysis; the estimates are used in the benefit-cost ratios presented (where applicable) in the results section.

Table 52 Overall summary of cost estimates for change scenarios (all figures billions 2007 US$).

Sector/trend	Change scenarios	Cost Estimate (2007 bn US$ per annum)	Cost Estimate (Cumulative at 0% Dr)	Cost Estimate (Cumulative at 1% Dr)	Cost Estimate (Cumulative at 4% Dr)
1. Agricultural productivity	Investment in Agricultural knowledge Science and Technology (AST) leads to 40% additional crop productivity and 20% additional livestock productivity compared to the baseline.	14.5 - this is treated as a minimum point in the cost estimate, although it is not possible to generate a cost estimate range *per se* without further primary research.	725	568.3	311.5
2. Reducing post-harvest losses in the food chain	Supply chain losses (i.e. post-harvest losses) are reduced by half, to 15% of total food supplies. Change scenario assumes full agricultural liberalization.	No cost estimate is available.			
3. Forest Management	Replacement of conventional selective logging practices by reduced impact logging (RIL) with a compensating increase in forest plantations.	103.9 Lower bound estimate 41.6; Upper bound 237.7	5195 Lower bound 2080 Upper bound 11885	4072.5 Lower bound 630.6 Upper bound 9316.9	2232 Lower bound 893.7 Upper bound 5106.3
4. Protected Areas	(1) Expansion of protected areas to 20% of each ecological region. (2) Expansion of protected areas to 50% of each ecological region.	(1) 'Best-guess' 15.4 Upper bound 66.4 (2) 'Best-guess' 61.72 Upper bound 265.95	(1) 'Best-guess' 465.1 Upper bound 1995.7 (2) 'Best-guess' 1860 Upper bound 7982.8	(1) 'Best-guess' 400.4 Upper bound 1717.1 (2) 'Best-guess' 1601.6 Upper bound 6868.5	(1) 'Best-guess' 268.9 Upper bound 1151.1 (2) 'Best-guess' 1075.7 Upper bound 4604.6
5. Reduced deforestation	Protection of all forests and woodlands from agricultural expansion.	Lower bound estimate is 7.9 Upper bound estimate is 22.5	Lower bound 248.3 Upper bound 679	Lower bound 213.7 Upper bound 584.6	Lower bound 142.18 Upper bound 392.63
6. Bio-energy	1) GHG concentration limited to 445 ppm CO_2-equivalent by including an expansion in bio-energy. Bio-energy land requirement of 4 million square km by 2050.	Lower bound estimate is 16.7 Upper bound estimate is 30.7	The estimate is 1297.5 using 16.7 bn per year up to 2017 and 30.7bn per year from 2017 to 2050.	The estimate is 985.9 using 16.7 bn per year up to 2017 and 30.7bn per year from 2017 to 2050.	The estimate is 489.5 using 16.7 per year up to 2017 and 30.7 per year from 2017 to2050.
7. Global dietary patterns	1) Complete substitution of all meat by plant-based protein consumption. 2) 'Willett diet'; '.	No cost estimate is available.			
8. Global agricultural trade	Tariff, non-tariff barriers and subsidies are gradually removed between 2010 and 2020 resulting in full liberalization of trade in agricultural commodities.	0 net costs associated with trade liberalization as the balance of evidence implies that, on a global basis, this is appropriate.			

10 Valuation of changes in wetlands, mangroves, coral reefs, and lakes and rivers

10.1 Introduction

This section provides a complementary analysis for changes in biomes for which the IMAGE-GLOBIO model does not produce output. *The results in Section 10 do not pertain to the eight option scenarios*. The IMAGE-GLOBIO model focuses on terrestrial land-use change and uses a classification of land use that reflects this. The Global Land Cover classification relates primarily to forest and grassland biomes. The other biomes that we assess in this section (Section 10) are wetlands, mangroves, coral reefs, and lakes and rivers.

The valuation of changes in ecosystem services derived from these biomes follows a similar methodology as for forests and grasslands. Meta-analyses of the economic valuation literature related to each biome are used to estimate biome specific value functions that relate differences in value to differences in the characteristics of patches of each biome. These value functions are then used to estimate the value of patch level changes in biome extent. The biome-level discussion of databases and value functions is provided in Section 8 and not repeated here.

Projected changes in the extent of each biome over time are not, however, explicitly modelled in a bio-physical model. Instead, *in the absence of new bio-physical modelling*, changes in the extent of each biome are assumed to follow similar patterns to the IMAGE-GLOBIO modelled changes for forests and grasslands. Since there is no available bio-physical model for the biomes assessed in this section, we only evaluate changes for the baseline (no policy action) scenario for the period 2000 to 2050.

The structure of this section is as follows, first the method for deriving assumed changes in the extent of biomes is outlined and a summary of regional changes is presented; secondly, for each biome in turn, estimated changes ecosystem service values are presented.

10.2 Bio-physical changes in biomes

The approach used for deriving bio-physical changes differs between wetlands, mangroves and coral reefs on the one hand and lakes and rivers on the other. Changes in wetlands, mangroves and coral reefs are assessed in terms of quantity (area) whereas changes in lakes and rivers are assessed in terms of water quality. The analysis for the latter biome is therefore described separately.

10.2.1 Changes in wetlands, mangroves and coral reefs

Due to the unavailability of a global bio-physical model for wetlands, mangroves and coral reefs, we assume that changes in these biomes follow similar spatial patterns to the IMAGE-GLOBIO modelled changes for forests and grasslands. The reasoning behind this assumption is that the population, development and land use pressures that drive changes in the extent of forests and grasslands will also tend to drive degradation and conversion of wetlands, mangroves and coral reefs. We recognise, however, that there are differences in the way in which development pressures affect different biomes and different regions, and indeed that some biomes face unique pressures and are more sensitive than others.

In the IMAGE model underlying the GLOBIO model, land use change is largely a result of food demand, trade and land intensity assumptions. Depending on the eco-region, land that is unutilised for agriculture is allocated to the climax vegetation (forest in most of the world, grass in dryer areas). Pressure due to agricultural demand on grassland and forest does not necessarily translate well to pressure on mangroves, coral reefs and wetlands. For wetlands there may be a relation to some extent given that agricultural demand tends to result in wetland conversion. The process resulting in degradation and conversion of mangroves, however, is largely related to shrimp aquaculture production. Coral reefs arguably face a more complex mix of pressures including sedimentation, pollution, over fishing, invasive species, excessive tourism, ocean acidification and climate change. For this reason it is highly doubtful that increasing grassland and forest area translates to increasing wetland, mangrove and coral area.

With the above in mind, it is considered appropriate to select and transfer the lowest change factor for forest or grassland to other biomes within the same 50km grid cell. In the case that there are no data available for either forest or grassland within a specific grid cell, data are taken from the nearest available cell. If changes in grassland and forest are both positive, we assume 'no change' in the other biomes. In other words, we take the pessimistic view that wetlands, mangroves and coral reefs can only decrease or remain constant in area.

Using spatially differentiated change factors derived in this way, we calculate the change in area of each patch of wetland, mangrove and coral reef. The aggregated change in area for each biome by region is presented in Table 53. The largest losses in wetland area are projected to occur in Western Africa, Southern Africa, Russia, India and Indonesia. The largest losses in mangrove area are expected in again Western Africa and Indonesia. For coral reefs, the largest losses in area or coral cover are expected in Indonesia.

10.2.2 Changes in water quality in lakes and rivers

Changes in water quality are based directly on output from the IMAGE-GLOBIO model. The IMAGE-GLOBIO model at present focuses on terrestrial land use change, but also calculates the effects of these changes on freshwater systems (wetlands, lakes and rivers) in terms of water quality. Data on surface water concentration levels of total nitrogen and phosphorous at the 50km grid cell level are available for the baseline scenario 2000-2050. This data was translated first to units of total nitrogen equivalent using a conversion factor of 10 units phosphorous to 1 unit nitrogen equivalent. This measure of water quality was then converted to the 10 point water quality index used in the value function. This step uses conversion factors described in Brouwer *et al.* (2009). Descriptive statistics for changes in water quality in lakes and rivers by region are presented in Table 54. All regions are projected to experience overall decreases in water quality (except for Western Europe and Japan) but the spatial variation in changes within regions is wide. Within all regions there are both very high negative as well as positive changes in water quality. At the regional level, Southern Africa, Eastern Africa and India are expected to experience the highest levels of degradation in water quality. It is important to note that changes in water quality addressed in this analysis are only those related to land use change and specifically only to nitrogen and phosphorous. Other sources of water pollution and other pollutants are not covered by this analysis. Given the range of other important sources of water pollution, our analysis should be treated as partial assessment of the impact of change in water quality on ecosystem services.

Table 53 Losses in wetland, mangrove and coral reef area 2000-2050 by region (hectares; thousands).

Region	Wetlands	Mangroves	Coral reefs
Canada	4,998		
USA	1,761	1	26
Mexico	60	25	3
Rest Central America	153	63	146
Brazil	2,420	42	9
Rest South America	8,020	195	13
Northern Africa	22		
Western Africa	11,548	1,046	9
Eastern Africa	3,482	196	222
Southern Africa	11,024	398	243
Western Europe	542		
Central Europe	2		
Turkey	321		
Ukraine +	1,844		
Asia-Stan	3,516		
Russia +	10,093		
Middle East	24		1
South Asia	10,216	418	979
Korea	6		
China +	3,377		6
Southeastern Asia	4,879	334	269
Indonesia +	10,092	1,844	2,155
Japan	57		1
Oceania	1,908	355	271
Total	**90,365**	**4,916**	**4,354**

Table 54 Changes in water quality in lakes and rivers 2000-2050 by region (10 point water quality index, positive changes indicating water quality improvement).

Region	Mean	Median	Std. Deviation	Minimum	Maximum
Canada	-1.26	-0.99	2.43	-9.24	9.25
USA	-0.25	-0.19	2.19	-8.99	9.25
Mexico	-0.47	-0.74	2.34	-7.29	7.00
Rest Central America	-1.26	-1.04	2.85	-8.99	8.00
Brazil	-0.82	-0.99	2.07	-7.99	8.00
Rest South America	-0.93	-0.99	2.11	-8.99	8.00
Northern Africa	-1.68	-0.99	2.27	-7.99	1.00
Western Africa	-1.92	-1.91	1.60	-7.99	5.00
Eastern Africa	-2.30	-1.99	2.31	-8.99	8.00
Southern Africa	-2.80	-2.93	1.85	-8.97	1.82
Western Europe	0.91	1.00	1.79	-7.99	9.00
Central Europe	-0.22	-0.24	1.98	-5.65	7.00
Turkey	-2.51	-1.49	2.67	-7.99	5.00
Ukraine +	-1.79	-1.99	1.82	-7.05	6.00
Asia-Stan	-1.87	-1.90	1.88	-8.99	7.00
Russia +	-1.24	-0.99	1.67	-7.99	8.00
Middle East	-0.65	-0.24	1.74	-9.24	3.00
South Asia	-2.25	-0.99	2.57	-9.24	4.30
Korea	-0.01	0.24	2.31	-6.99	7.00
China +	-1.22	-0.99	2.08	-8.99	7.00
Southeastern Asia	-1.32	-0.99	2.75	-8.99	9.00
Indonesia +	-1.99	-0.99	2.79	-8.99	8.00
Japan	1.30	1.00	1.72	-2.99	9.00
Oceania	-1.92	-0.99	3.35	-8.99	7.33

10.3 Wetlands

At the level of individual patches of wetland, patch specific parameter values are substituted into the value function to estimate the marginal value (US$/ha/annum) of each patch. This is then used to calculate the value of the projected change in area of each patch. Changes in wetland values aggregated to the regional level are presented in Table 55. Comparing the 2000 stock of wetlands to the projected 2050 stock (a global loss of 90 million hectares) the global value of lost ecosystem services from wetlands is estimated to be approximately 187 billion US$ per year (2007 prices). Assuming a linear time profile of these losses between 2000 and 2050, the present value of the stream of lost ecosystem services is 4,767 billion US$ using a 0% discount rate, 3,435 billion US$ using a 1% discount rate and 1,431 billion using a 4% discount rate.[58]

[58] The use of an assumed linear trajectory from 2000 to 2050 is discussed further in Sections 12 and 13.

Table 55 Change in value of ecosystem service provision from wetlands 2000-2050 (billions US$, 2007 prices)

Region	Annual value in 2050	Present value (0%)	Present value (1%)	Present value (4%)
Canada	-3.71	-94.64	-68.18	-28.40
USA	-5.45	-138.89	-100.07	-41.68
Mexico	-0.48	-12.34	-8.89	-3.70
Rest Central America	-0.23	-5.85	-4.22	-1.76
Brazil	-2.07	-52.83	-38.06	-15.85
Rest South America	-8.28	-211.05	-152.06	-63.34
Northern Africa	-0.21	-5.44	-3.92	-1.63
Western Africa	-15.03	-383.14	-276.05	-114.99
Eastern Africa	-4.84	-123.30	-88.84	-37.00
Southern Africa	-16.21	-413.26	-297.75	-124.03
Western Europe	-2.80	-71.35	-51.40	-21.41
Central Europe	-0.03	-0.73	-0.52	-0.22
Turkey	-1.83	-46.62	-33.59	-13.99
Ukraine +	-5.76	-146.80	-105.77	-44.06
Asia-Stan	-12.18	-310.48	-223.69	-93.18
Russia +	-13.16	-335.69	-241.86	-100.74
Middle East	-0.63	-16.08	-11.58	-4.83
South Asia	-56.41	-1,438.33	-1036.29	-431.66
Korea	-0.02	-0.59	-0.42	-0.18
China +	-17.44	-444.62	-320.34	-133.44
Southeastern Asia	-7.40	-188.59	-135.87	-56.60
Indonesia +	-9.39	-239.48	-172.54	-71.87
Japan	-1.26	-32.21	-23.21	-9.67
Oceania	-2.16	-55.14	-39.73	-16.55
World	**-187**	**-4,767**	**-3,435**	**-1,431**

10.4 Mangroves

At the level of individual patches of mangrove, patch specific parameter values are substituted into the value function to estimate the marginal value (US$/ha/annum) of each patch. This is then used to calculate the value of the projected change in area of each patch. Changes in mangrove values aggregated to the regional level are presented in Table 56. Comparing the 2000 stock of mangroves to the projected 2050 stock, the global value of lost ecosystem services from mangroves is estimated to be approximately 4 billion US$ per year (2007 prices). Assuming a linear time profile of these losses between 2000 and 2050, the present value of the stream of lost ecosystem services is 102 billion US$ using a 0% discount rate, 74 billion US$ using a 1% discount rate and 31 billion using a 4% discount rate.

Table 56 Change in value of ecosystem service provision from mangrove 2000-2050 (billions US$, 2007 prices)

Region	Annual value in 2050	Present value (0%)	Present value (1%)	Present value (4%)
USA	-0.02	-0.40	-0.29	-0.12
Mexico	-0.18	-4.69	-3.38	-1.41
Rest Central America	-0.07	-1.77	-1.27	-0.53
Brazil	-0.12	-3.03	-2.18	-0.91
Rest South America	-0.31	-7.85	-5.65	-2.36
Western Africa	-0.36	-9.13	-6.58	-2.74
Eastern Africa	-0.04	-1.02	-0.74	-0.31
Southern Africa	-0.10	-2.65	-1.91	-0.80
South Asia	-0.22	-5.51	-3.97	-1.65
China +	0.00	-0.02	-0.01	-0.01
Southeastern Asia	-0.38	-9.69	-6.98	-2.91
Indonesia +	-1.59	-40.57	-29.23	-12.18
Oceania	-0.62	-15.70	-11.31	-4.71
World	**-4**	**-102**	**-74**	**-31**

10.5 Coral reefs

At the level of individual patches of coral reef, patch specific parameter values are substituted into the value function to estimate the marginal value (US$/ha/annum) of each patch. This is then used to calculate the value of the projected change in area of each patch. Changes in coral reef values aggregated to the regional level are presented in Table 57. Comparing the 2000 stock of coral reefs to the projected 2050 stock, the global value of lost ecosystem services from coral reefs is estimated to be approximately 82 billion US$ per year (2007 prices). Assuming a linear time profile of these losses between 2000 and 2050, the present value of the stream of lost ecosystem services is 2,088 billion US$ using a 0% discount rate, 1,504 billion US$ using a 1% discount rate and 627 billion using a 4% discount rate.

Table 57 Change in value of ecosystem service provision from coral reefs 2000-2050 (billions US$, 2007 prices)

Region	Annual value in 2050	Present value (0%)	Present value (1%)	Present value (4%)
USA	-0.27	-6.88	-4.96	-2.07
Mexico	-0.07	-1.73	-1.24	-0.52
Rest Central America	-2.40	-61.29	-44.16	-18.39
Brazil	-0.27	-6.94	-5.00	-2.08
Rest South America	-0.25	-6.31	-4.55	-1.90
Northern Africa	0.00	-0.04	-0.03	-0.01
Western Africa	-0.15	-3.76	-2.71	-1.13
Eastern Africa	-4.96	-126.55	-91.18	-37.98
Southern Africa	-4.03	-102.72	-74.01	-30.83
Middle East	-0.05	-1.37	-0.99	-0.41
South Asia	-16.00	-407.89	-293.88	-122.41
China +	-0.18	-4.61	-3.32	-1.38
Southeastern Asia	-4.62	-117.73	-84.82	-35.33
Indonesia +	-43.37	-1105.97	-796.83	-331.92
Japan	-0.01	-0.30	-0.22	-0.09
Oceania	-5.25	-133.92	-96.49	-40.19
Total	**-82**	**-2,088**	**-1,504**	**-627**

10.6 Lakes and rivers

To estimate the mean household WTP for water quality changes in lakes and rivers in each IMAGE-GLOBIO grid cell, we substitute grid cell specific parameter values into the estimated value function. The total value of water quality change in each grid cell is then calculated by multiplying mean household WTP by the number of households in each grid cell. The number of households is computed as the population divided by mean household size. The value of changes in water quality aggregated to the regional level is presented in Table 58. Comparing the 2000 water quality to the projected 2050 water quality, the global value of this change is estimated to be approximately -69 billion US$ per year (2007 prices). India is expected to suffer the highest loss in value of ecosystem services from lakes and rivers due to deteriorating water quality. Western Europe and Japan are projected to experience small increases in the value of lakes and rivers due to improving water quality. Assuming a linear time profile of these changes between 2000 and 2050, the present value of the changes in ecosystem services is -1,768 billion US$ for a 0% discount rate, -1,274 billion US$ for a 1% discount rate and -531 billion US$ for a 4% discount rate.

There are three reasons why we consider these estimates to be conservative. First, the available value information on water quality change, and therefore the scaled up values, are focussed on recreational and non-use values. For this reason our estimates are expected to underestimate the value of provisioning services related to water, which are of greater importance in developing countries. This underestimate is compounded by the fact that it is in developing countries where the greatest degradation of water quality is expected to occur. In other words, the largest losses in ecosystem services are forecasted in developing countries but the services most relevant to developing countries may be underestimated. Second, our analysis is limited to examining water quality changes resulting only from land-use change and specifically from the pollutants nitrogen and phosphorous. Other sources and pollutants are not included in the assessment. Third, the reported regional and global values are *net* values. We assess both positive and negative changes in water quality. Within all regions there are areas that experience improvements in water quality and other areas that experience degradation. In most regions, the reduction in the value of ecosystem services from lakes and rivers due to reduced water quality outweighs the benefits of localised improvements.

Table 58 Value of change in water quality in lakes and rivers 2000-2050 (billions US$, 2007 prices)

Region	Annual value in 2050	Present value (0%)	Present value (1%)	Present value (4%)
Canada	-0.46	-11.65	-8.39	-3.50
USA	-0.56	-14.20	-10.23	-4.26
Mexico	-1.21	-30.95	-22.30	-9.29
Rest Central America	-0.96	-24.41	-17.59	-7.33
Brazil	-2.11	-53.68	-38.68	-16.11
Rest South America	-2.43	-61.86	-44.57	-18.57
Northern Africa	-1.90	-48.42	-34.89	-14.53
Western Africa	-8.65	-220.69	-159.01	-66.23
Eastern Africa	-3.93	-100.20	-72.20	-30.07
Southern Africa	-3.75	-95.52	-68.82	-28.67
Western Europe	2.17	55.34	39.87	16.61
Central Europe	-0.21	-5.28	-3.80	-1.58
Turkey	-0.96	-24.47	-17.63	-7.34
Ukraine +	-1.07	-27.41	-19.75	-8.22
Asia-Stan	-0.62	-15.89	-11.45	-4.77
Russia +	-1.32	-33.75	-24.32	-10.13
Middle East	-0.91	-23.32	-16.80	-7.00
South Asia	-25.23	-643.41	-463.56	-193.10
Korea	-0.22	-5.60	-4.03	-1.68
China +	-8.02	-204.39	-147.26	-61.34
Southeastern Asia	-4.55	-116.04	-83.60	-34.82
Indonesia +	-3.52	-89.83	-64.72	-26.96
Japan	1.60	40.74	29.36	12.23
Oceania	-0.53	-13.61	-9.80	-4.08
World	**-69**	**-1,768**	**-1,274**	**-531**

10.7 Conclusions

In Section 10 we have made an attempt to examine changes in four biomes that fall outside the scope of the IMAGE-GLOBIO model, namely wetlands, mangroves, coral reefs, and lakes and rivers. We assume that changes in these biomes follow the same spatial distribution as changes in forests and grasslands, since to a large extent they are also driven by the same underlying pressures. The economic value of projected changes in the provision of ecosystem services from these biomes is substantial. The combined value of the loss in ecosystem services derived from them between 2000 and 2050 is US$8,725 billion (0% discount rate). The value of lost ecosystem services from wetlands dominates this finding with an estimated value of US$4,767 billion. Mangroves and coral reefs naturally cover much smaller areas and the value of reductions in their ecosystem services is lower. In terms of value per unit of area, however, coral reefs are shown to have by far the highest value: US$220,000ha/year compared with US$17,000ha/year for wetlands and US$4,500ha/year for mangroves. The value of changes in water quality in lakes and rivers over the period 2000-2050 is estimated to be US$1,768 billion (0% discount rate). However, we consider this to be an underestimate since the valuation literature under-represents values for provisioning services related to clean water. Further research is required to provide more information on the value of changes in water quality, particularly in developing countries.

11 Complimentary analysis of marine ecosystems

11.1 Introduction

Marine ecosystems provide a range of ESSs that have been estimated to constitute over half of global ESSs provision (see Hussain *et al.*, 2010; Costanza *et al.*, 1997). Not only is the level of ESS provision significant but it is also under threat (e.g. Halpern *et al.*, 2008). Section 11 considers marine ecosystems under a specific change scenario, *viz*. temporary reductions in fishing catch and effort in order to regenerate wild fish stocks, with a compensating increase in aquaculture so as to provide substitute animal protein intake.

This scenario option is developed by PBL (2010) in collaboration with University of British Columbia (UBC) (2010); the bio-physical assumptions and modelling results are set out in Appendix IV, with a synopsis provided in Section 11.3. IMAGE-GLOBIO is restricted in scope to terrestrial ecosystems. The authors of PBL (2010) modelled the effects on terrestrial biomes of increased aquaculture, i.e. the effects of switching agricultural production systems to provide feedstuffs for aquaculture. However, the authors found these effects to be negligible, with a global area loss of circa 0.2 million km^2 and therefore the IMAGE-GLOBIO output is limited for this change scenario.

The analysis is thus partial for two reasons: UBC (2010) modelling is restricted to fish catch, i.e. one specific provisioning ESS; there is relatively limited specificity in UBC (2010) vis-à-vis the location of aquaculture developments and the species expected to be harvested. The approach that we adopt in this current study is as follows: (1) to use secondary data on spatially-referenced fish price projections to determine the economic efficiency (or otherwise) of scenario options for reduced fish catch modelled in UBC (2010); (2) to provide a background review of the literature on the potential costs of aquaculture development.

11.2 Background to marine fisheries analysis

A number of studies (e.g. FAO, 1997; FAO, 1998a; FAO, 1998b; Ludwig *et al.*, 1993; Safina, 1995; Mullon *et al.*, 2005) have established that the catching capability of participants in fisheries world-wide has substantially increased, and that capacity needs to be reduced in virtually all fisheries to achieve a sustainable balance. In most cases, this requires the development of long term recovery programmes such as effort reduction to achieve the sustainable exploitation of fish stocks (Murawski *et al.*, 1999). However, policy makers continue to grapple to bring fishing effort under control (Rosenberg 2003) and many other problems still remain.

The objective of the sections pertaining to marine capture fisheries (Sections 11.2-11.4) is to estimate the benefits arising from effort reductions in a simulated baseline scenario (or the current *status quo*) and three policy change scenarios for 15 of the 19 FAO marine fishing regions. The three scenarios to restore global fish stocks, borrowing from UBC (2010), are (1) High Ambition. (2) High Ambition with Ramp-down and (3) Medium Ambition. While fishing effort reduction confers numerous benefits which may be broader in scope, the objective is to estimate expected benefits and costs arising from catch projections (and catch projections alone) up to 2049 from the aforementioned scenarios.

11.3 Description of bio-physical baseline and policy change scenarios

Projection in the baseline and policy change scenarios were implemented in UBC (2010) using a slightly modified version of the global EcoOcean model detailed in Alder *et al.* (2007). The most notable modification in the new models is the application of a 'technology creep' of 3% per year to capture the effect of modernisation of fishing technology since 1950. The model was built using 43 functional groups (composed of a single species or of a group of species) that are common to the world's oceans. The groups were selected with special consideration for exploited fish species, but are intended to jointly include all major groups in the oceans. The fish groups are based on both size categories and feeding and habitat characteristics. Fishing effort is the most important driver for the ecosystem model simulations. Five major fleet categories (demersal, distant-water, baitfish tuna/purse seine, tuna longline and small pelagic) were used to distinguish different fishing effort based on historical information.

The business-as-usual scenario assumes that: (1) world demand for fish continues to increase, (2) landings from captured fisheries level off and (3) over-exploitation puts many stocks at risk of depletion. The business-as-usual scenario uses projections based on estimated 2002 effort levels for each fleet into the future. Further details are provided in UBC (2010).

Three scenarios were formulated, each addressing the problem of declining global captured fish stocks and each applying regulatory interventions to increase the productivity in fish stocks through the effort reduction strategies for 15 FAO marine regions (Figure 13)[59]. The change scenarios are based on the assumption that production from marine captured fisheries has peaked and fishing effort will have to be reduced so as to optimise long-term catches. In the first scenario (High Ambition) this entails an immediate reduction in fishing effort such that the fisheries are restored so as to produce the maximum sustainable yield (MSY)[60]. To achieve MSY in this scenario would require sharp cuts in fishing efforts in many FAO fishing regions. In the second scenario (High Ambition with Ramp-down), there is gradual effort reduction over the period of 10 years. The third scenario (Medium Ambition) involves fishing effort reduction but not to the same degree of the other two policy change scenarios. The biophysical results of catch projections for business–as-usual and the 3 policy change scenarios are shown in Figure 14 to Figure 17. Note that the modelling in UBC (2010) assumes that changes occur from 2004, which is why there is variation between the models for the period 2004 to the current day.

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[59] 'FAO Major Fishing Areas for Statistical Purposes are arbitrary areas, the boundaries of which were determined in consultation with international fishery agencies on various considerations, including (i) the boundary of natural regions and the natural divisions of oceans and seas; (ii) the boundaries of adjacent statistical fisheries bodies already established in inter-governmental conventions and treaties; (iii) existing national practices; (iv) national boundaries; (v) the longitude and latitude grid system; (vi) the distribution of the aquatic fauna; and (vii) the distribution of the resources and the environmental conditions within an area (FAO).' FAO (2007)

[60] MSY is not generally considered economically efficient but is still often used as an objective nonetheless, as is the case in UBC (2010).

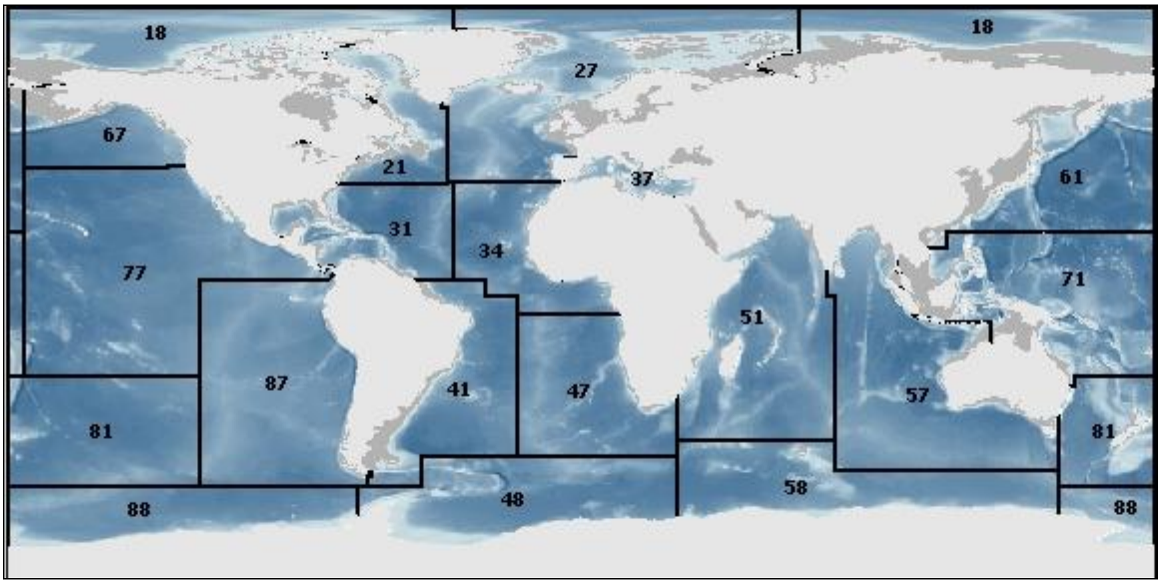

Figure 13 FAO Major Fishing Areas

Source: (FAO, http://www.fao.org/fishery/area/search/en)

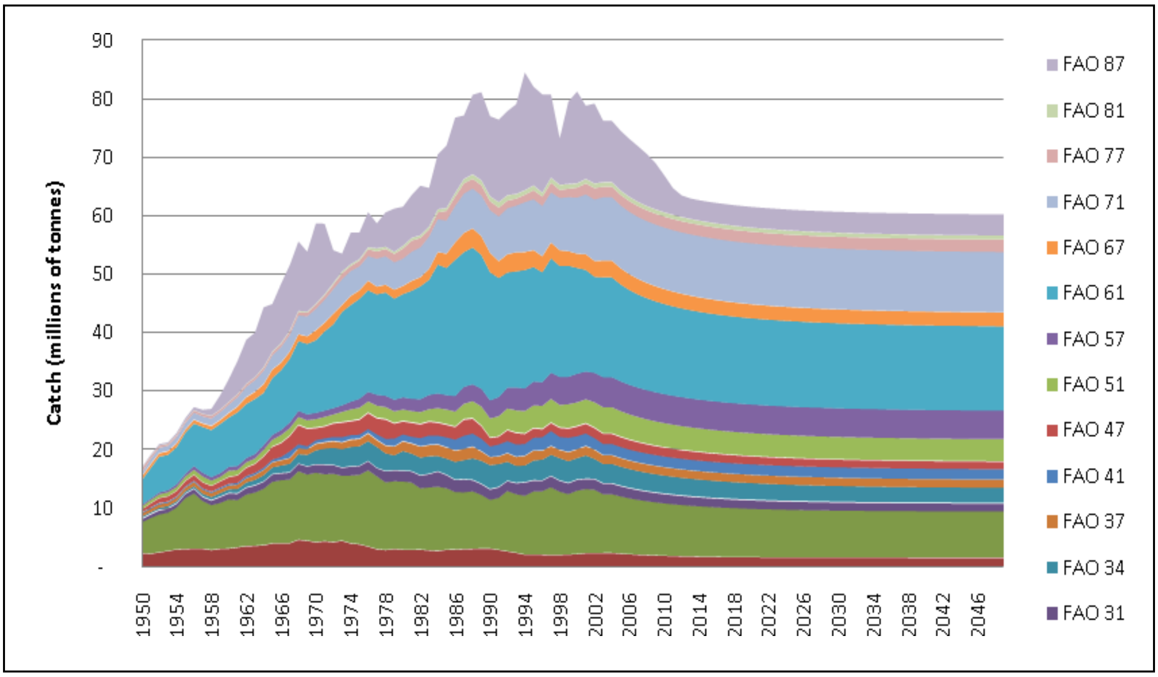

Figure 14 Marine bio-physical modelling: Business as usual

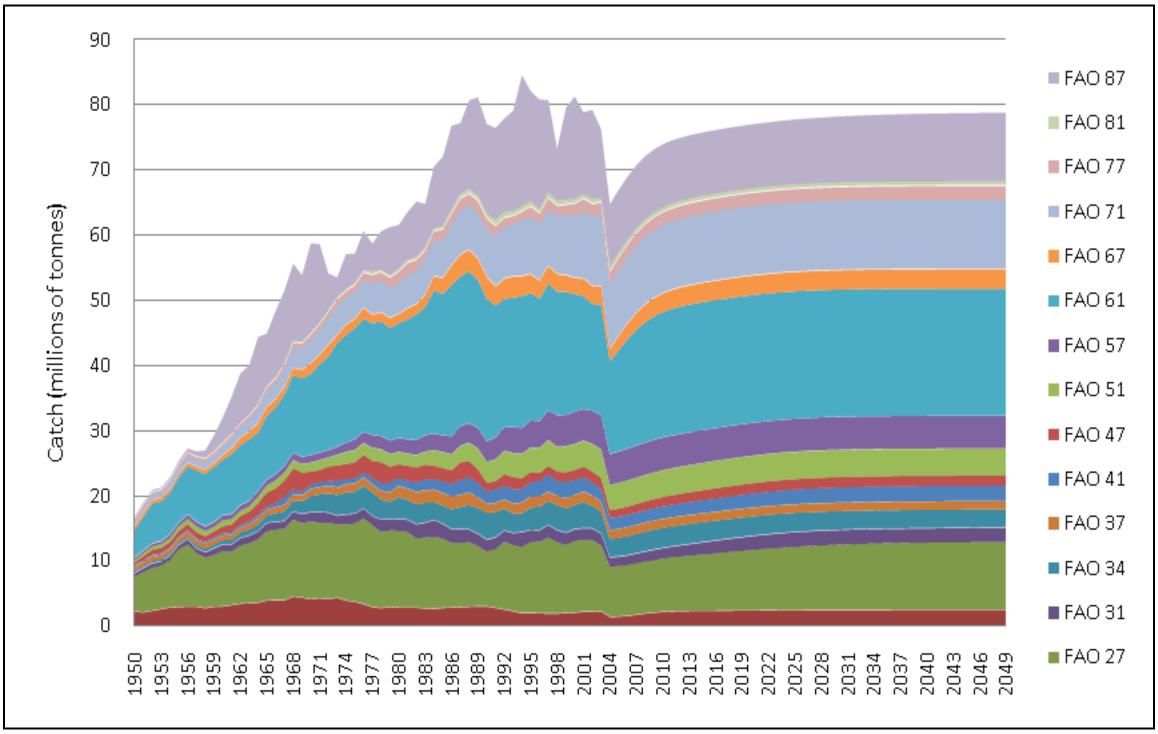

Figure 15 Marine bio-physical modelling: High Ambition

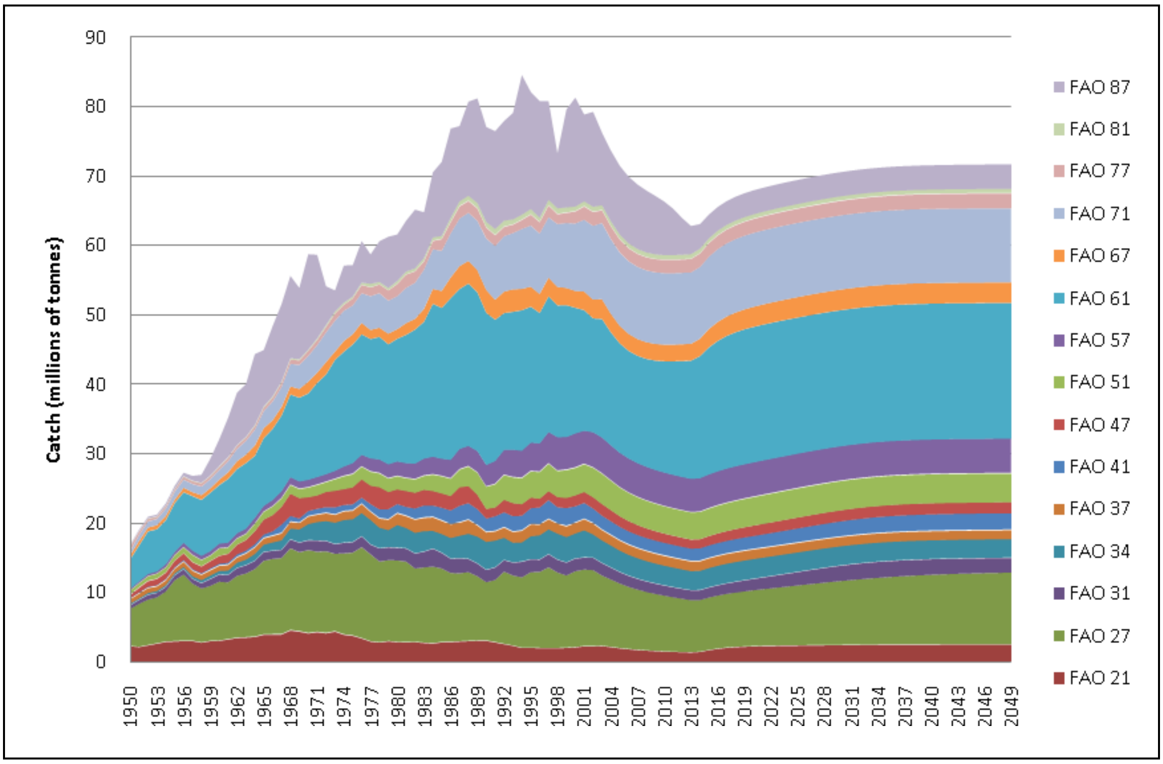

Figure 16 Marine bio-physical modelling: High Ambition with Ramp

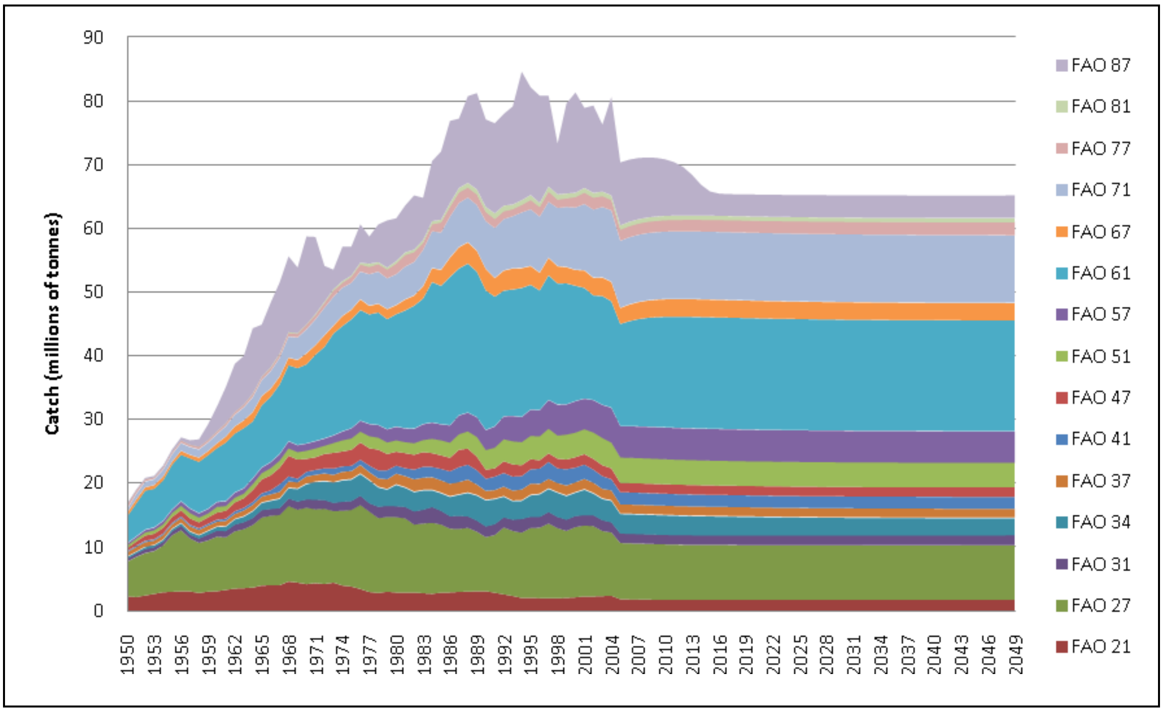

Figure 17 Marine bio-physical modelling: Moderate Ambition

Applying the 2002 effort level in the in the business-as-usual scenario (Figure 14) shows that total catch decreases to around 60 million tonnes from the 2002 levels of slightly under 80 million tonnes. The High Ambition scenario which involves an immediate reduction in effort to optimal levels showed an instant decline in catch of just over 10 million tons (Figure 15). The shortfall is regained over the proceeding 20-30 years with catch returning to just under 80 million tons. In the High Ambition with Ramp-down which involves the gradual effort reduction over the period of 10 years, the results show a moderate decline in marine caches to just over 70 million tons (Figure 16). The final scenario (Moderate Ambition) involves effort levels intermediate between 2002 levels and optimal levels, the results show landings to be around 65 million tons (Figure 17). Detailed results of the projection are present in Appendix V.

11.4 Economic appraisal of marine catch options

In this analysis, economic costs can be viewed as benefits foregone, i.e. the opportunity cost of regulating to reduce fish catch is the benefit foregone from the sale of fish. To estimate benefits and opportunity costs of the catch forecast under the four scenarios requires information pertaining to fish prices of catch projections up to 2049. Fish catch projections are estimated in the simulation model (UBC, 2010) as an assemblage of 43 *functional groups* of fish species common to the world ocean for each year. This makes benefits estimate by individual species inapplicable. Thus an average ex-vessel price[61] per tonne of the assemblage of fish caught for each FAO fishing region are used. The ex-vessel prices were derived by value – quantity transformation from the Sea Around Us project database[62]. It must be noted that the application of an average price can be an under-estimate of benefits for some species or an overestimate for others. Autoregressive integrated moving

[61] The ex-vessel price is the price received by fishermen for fish, shellfish etc. landed at the dock.
[62] http://www.seaaroundus.org/

average (ARIMA) or autoregressive moving average (ARMA) models were used to forecast ex-vessels prices of fish per tonne in each FAO fishing regions over the period 2006 to 2049. ARIMA and ARMA models were used because of their robustness and the less onerous demands on data availability as compared with structural multivariate models[63].

The flows of benefits associated with effort limitation take place over time. Discounting is needed to express future benefits or costs in present value terms. In most countries, governments provide guidelines for discount rates and these vary significantly from between 3% and 15% (Gustavson, 2000; Pearce and Ulph, 1999; Pendleton, 1995) from one country to the other, and the rate tend to be higher for developing countries than for industrialised countries. For example, FAO (2004) cite the following discount rates used in selected countries: United States (3%), Iceland (8%), Namibia (10%), Norway (3.5%), and United Kingdom (4%); we present results for three different discount rates, *viz*. 3%, 5% and 10%[64]. The results applying a 3% discount rate are presented Table 59. Note that the use of ex-vessel prices (which is fairly common in fisheries analysis) over-states total value as these prices do not include fishing costs.

Table 59 Discounted Benefits 2004-2049 in 2000 US$ (billions) prices at 3% discount rate.

	Business as Usual	High Ambition	High Ambition with Ramp	Moderate Ambition
FAO21: Atlantic, Northwest	9.55	13.44	12.23	10.17
FAO27: Atlantic, Northeast	3.38	3.59	3.45	3.38
FAO31: Atlantic, Western Central	7.02	9.44	8.80	7.75
FAO34: Atlantic, Eastern Central	13.46	13.60	13.53	13.61
FAO37: Mediterranean and Black Sea	47.78	44.15	44.99	46.48
FAO41: Atlantic, Southwest	28.89	33.60	31.90	29.10
FAO47: Atlantic, Southeast	4.47	4.93	4.80	4.74
FAO51: Indian Ocean, Western	69.65	74.38	73.23	71.68
FAO57: Indian Ocean, Eastern	38.78	38.69	38.60	38.83
FAO61: Pacific, Northwest	81.82	103.73	99.33	94.35
FAO67: Pacific, Northeast	3.74	4.34	4.21	4.14
FAO71: Pacific, Western Central	113.95	114.38	113.93	114.34
FAO77: Pacific, Eastern Central	31.55	31.61	31.61	31.67
FAO81: Pacific, Southwest	15.56	15.34	15.34	15.38
FAO87: Pacific, Southeast	21.27	44.11	21.71	24.02
Total	**490.86**	**549.33**	**517.64**	**509.63**

The present value of aggregate benefit ranged from US$ 491 billion to US$ 549 billion (2000 US$) across the four scenarios; business-as-usual scenario generates the lowest aggregate benefits whereas the highest benefit would be generated in the High-Ambition scenario. However, the proportional changes in net benefits relative to business-as-usual are relatively small, even at a relatively low discount rate: the proportional changes are 12%, 5% and 4% for High-Ambition, High-Ambition-with-Ramp and Moderate-Ambition scenarios respectively. It is interesting to note that in all scenarios the highest benefits will be extracted from regions FAO61 (Northwest Pacific) and FAO71 (Western Central Pacific). The high benefits from

[63] See Appendix V for detailed descriptions of ARIMA and ARMA forecasting technique used
[64] We consider different discount rates in this section for marine ecosystems as compared to the analysis elsewhere in Part III as the analysis is exclusively concerned with fish productivity, i.e. a provisioning ecosystem service.

these regions maybe due to the dominance of tuna catch (skipjack, yellowfin, bigeye and albacore) which are of higher commercial values. Table 60 and Table 61 present results of aggregate benefits estimates at discount rates 5% and 10% respectively.

As is the case for the analysis using the 3% discount rate, at both 5 % and 10% discount rates business-as-usual scenario would generate the lowest aggregate benefits and the highest aggregate benefit would be generated in the High Ambition scenarios. However, the proportional differences between the four options are again relatively low.

Table 60 Discounted Benefits 2004-2049 in 2000 US$ (billions) prices at 5% discount rate

	Business as Usual	High Ambition	High Ambition with Ramp	Moderate Ambition
FAO21: Atlantic, Northwest	7.48	9.97	8.97	7.77
FAO27: Atlantic, Northeast	2.66	2.71	2.63	2.62
FAO31: Atlantic, Western Central	5.42	6.92	6.42	5.84
FAO34: Atlantic, Eastern Central	10.34	10.34	10.30	10.39
FAO37: Mediterranean and Black Sea	36.97	34.34	35.04	35.94
FAO41: Atlantic, Southwest	21.92	24.83	23.55	21.93
FAO47: Atlantic, Southeast	3.54	3.78	3.69	3.68
FAO51: Indian Ocean, Western	53.19	56.15	55.18	54.24
FAO57: Indian Ocean, Eastern	29.78	29.58	29.52	29.72
FAO61: Pacific, Northwest	64.08	79.75	75.71	72.86
FAO67: Pacific, Northeast	2.92	3.31	3.19	3.19
FAO71: Pacific, Western Central	90.06	89.92	89.60	89.87
FAO77: Pacific, Eastern Central	24.24	24.28	24.28	24.23
FAO81: Pacific, Southwest	11.83	11.66	11.66	11.68
FAO87: Pacific, Southeast	17.58	33.39	18.03	20.09
Total	**381.99**	**420.94**	**397.77**	**394.03**

Table 61 Discounted Benefits 2004-2049 in 2000 US$ (billions) prices at 10% discount rate

	Business as Usual	High Ambition	High Ambition with Ramp	Moderate Ambition
FAO21: Atlantic, Northwest	4.39	5.16	4.61	4.36
FAO27: Atlantic, Northeast	1.57	1.47	1.48	1.51
FAO31: Atlantic, Western Central	3.09	3.51	3.28	3.18
FAO34: Atlantic, Eastern Central	5.80	5.65	5.67	5.77
FAO37: Mediterranean and Black Sea	21.07	19.76	20.20	20.57
FAO41: Atlantic, Southwest	12.06	12.91	12.34	11.93
FAO47: Atlantic, Southeast	2.12	2.10	2.08	2.13
FAO51: Indian Ocean, Western	29.67	30.46	29.94	29.78
FAO57: Indian Ocean, Eastern	16.74	16.46	16.47	16.68
FAO61: Pacific, Northwest	37.30	44.38	41.50	41.30
FAO67: Pacific, Northeast	1.68	1.80	1.74	1.79
FAO71: Pacific, Western Central	53.32	52.58	52.55	52.97
FAO77: Pacific, Eastern Central	13.59	13.62	13.62	13.56
FAO81: Pacific, Southwest	6.53	6.44	6.44	6.45
FAO87: Pacific, Southeast	11.64	18.28	11.98	13.68
Total	**220.58**	**234.57**	**223.89**	**225.66**

What this analysis therefore tells us is that (1) the preference ordering across the options is not significantly sensitive to the discount rate applied, and (2) results indicate that the marginal benefit of any of the three policy options over the business-as-usual baseline is low. The latter point is important as *the level of confidence that we can have in the ranking of options is low*; there is uncertainty vis-à-vis both the bio-physical modelling and the economic forecasting. However, it must be noted that this outcome does not allow for co-benefits in terms of increases in the provision of other ESSs arising from both (1) reduced fish catch in earlier years and (2) more stable fish populations post-recovery.

11.5 Aquaculture expansion as a scenario sub-option

Between 1970 and 2004 the expansion in global aquaculture was significant with an annual global growth of around 8.8% per year (Stickney, 2009), but it does not currently match capture fisheries production. In combination with the fact that fish protein constitutes a significant, albeit variable, percentage of animal protein around the world, this means that many communities globally currently have a significant and increasingly imperilled reliance on marine capture fisheries. Consequently, as human populations continue to increase, the expansion of aquaculture production will become increasingly important if the world is to maintain the same per capita fish protein availability in the future, as exists now (Bardach, 1997b; Donaldson, 1997; Stickney, 2009). Accordingly, some of the policy scenarios included in this modelling effort considered addressing the problem of declining capture fisheries by enforcing a reduction in allowable fishing effort in most of the FAO regions (see section 11.3), and compensating this reduction in capture fisheries production with an increase in global aquaculture production. The High Ambition scenario involves an immediate and severe decrease in catch effort, and therefore implies an almost immediate increase in aquaculture production. The High Ambition scenario with a built in Ramp involves reducing catch effort over the course of ten years, which would imply a systematic increase in aquaculture production. The intermediate effort scenario involves decreasing catch effort, but not to the extent of the previous two scenarios, and so implies a less dramatic increase in aquaculture production.

We are not able to estimate the costs of increased aquaculture production in this report: (1) increases in production are non-marginal; (2) aquaculture production efforts have the potential to incur a wide variety of production-related and environment-related costs, all of which are very situation and location specific (Pillay and Kutty, 2005).

The discussion will therefore accomplish the following in lieu of the quantification of the costs of increased aquaculture production: a qualitative discussion of the types of costs that could be incurred by such a hypothetical increase in aquaculture, and a discussion of the potential value of explicitly modelling specific aquaculture scenarios in future research efforts. This is presented in Appendix VI.

11.6 Conclusions

The marine/aquaculture scenario option is inconclusive. Although the business-as-usual scenario for marine capture fisheries has the lowest net benefits across the four options modelled, the extent to which competing options are superior is low relative to the uncertainties in the bio-physical modelling and economic valuation. This is from the perspective of fish catch; there may well be significant co-benefits from allowing stocks to replenish but these have not been modelled.

The change scenario includes a significant development in aquaculture to compensate for the reduction in wild fish. It has not been possible to place a monetary value on such a non-marginal (transformational) change. We found there to be a paucity of data and literature in this regard.

12 Results for the economic analysis of change scenarios

12.1 Introduction

In Section 12 we present the results of the value transfer exercise and valuation of the change scenarios. The value changes for each of the options scenarios are based on the three terrestrial biomes (temperate forests, tropical forests and grasslands) which were modelled in IMAGE-GLOBIO[65]. Thus the results in this section do *not* consider the other biomes modelled separately in the Sections 10 and 1 (i.e. wetlands, mangroves, coral reefs, lakes and rivers, and open oceans) as changes in these biomes are not modelled in the IMAGE-GLOBIO analysis. Our results are presented at the level of the regions used PBL (2010) when presenting land-use change within the IMAGE-GLOBIO analysis, as illustrated in Figure 18.

Table 62 presents the area of the terrestrial biomes in each region; discussions of land-use change under the scenario options below should be considered in the context of these baseline values. The biome sites used for the value transfer were derived from GLC2000 data; forest biomes were classified in our study as either temperate or tropical on the basis of latitude. Our classification of grassland differs from that used by PBL (2010) in that we include patches classified as grassland in cultivated areas; we use this classification because our grassland value function includes values for such pasture sites. The consequence of our grassland classification is that we have a larger area of grassland, but the relative change factors for that biome are lower than those used in the PBL (2010) analysis.

[65] GLOBIO presents results for seven distinct biomes, of these we combine 'boreal forest' and 'temperate forest', and 'grassland and steppe' and 'scrubland and savannah' into two single biomes: 'temperate forest' and 'grassland'. Further we do not have value functions for 'ice and tundra' or 'desert' so do not undertake analyses for these biomes.

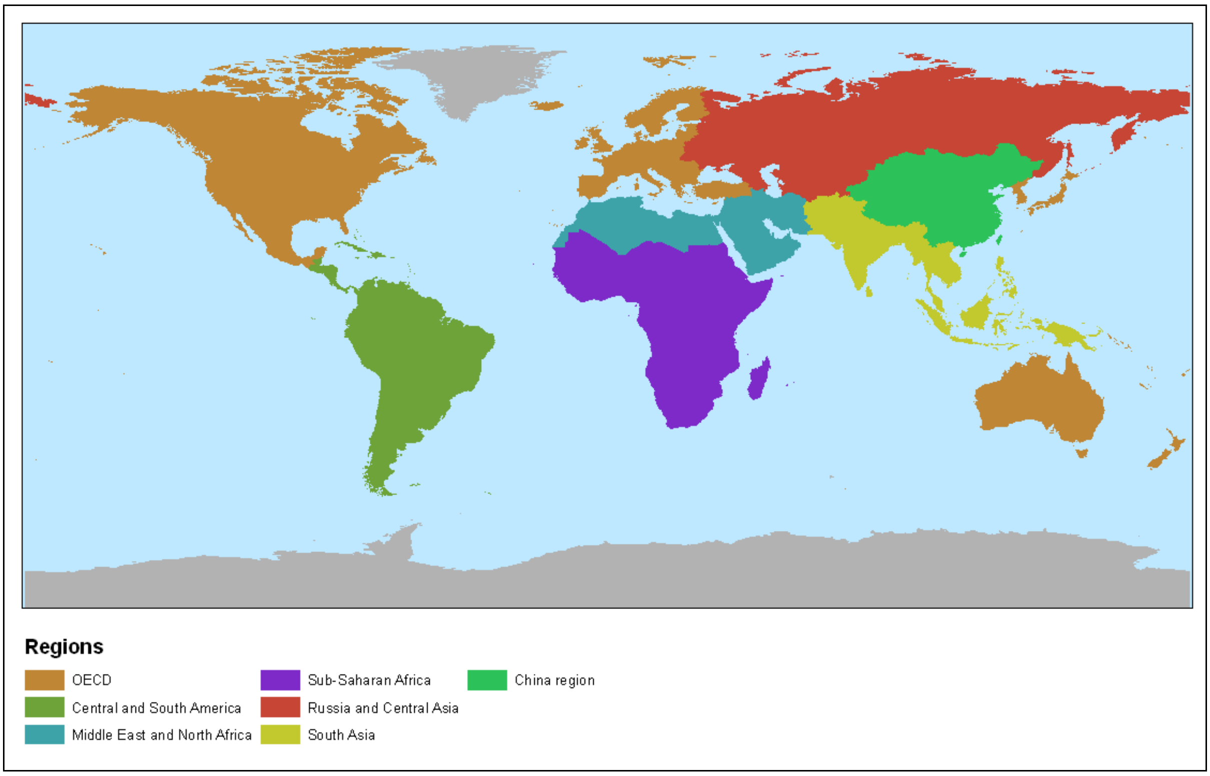

Figure 18 Map of regions

Table 62 Baseline area of terrestrial biomes considered in analysis ('000 km^2)

	Grassland	Temperate Forest	Tropical Forest
OECD	14197.4	9663.4	872.7
Central and South America	4255.2	789.3	6963.9
Middle East and North Africa	1464.3	71.6	10.0
Sub-Saharan Africa	7692.1	296.9	6401.4
Russia and Central Asia	5952.6	8356.8	0.0
South Asia	1723.9	509.8	2525.9
China Region	3983.7	1940.7	84.7
Total	**39269.1**	**21628.4**	**16858.6**

There are three constituent segments to the analysis of value changes that *cumulatively* describe whether or not an change scenario is economically efficient: (1) the value of changes in carbon storage arising with the change scenario versus the baseline to 2030/2050; (2) the value in 2030/2050[66] of land cover change that is forecasted to occur with the change scenario versus the 2030/2050 baseline; and (3) the costs of implementing the change scenario.

An important point to be made with regards the estimation of any value changes arising from each change scenario is that our analysis partial for several reasons:

(1) In this study we focus on valuing changes in land-use, i.e. the *quantity* of land-use under a particular categorization (i.e. GLC2000) as opposed to the *quality* of the ecosystem. We do attempt to capture some aspects of changes in quality by testing various spatial variables which affect habitat quality in the derivation of the value functions (see Section 8), e.g.

[66] The REDD and protected areas scenarios are modelled under GLOBIO to 2030, whereas other options are modelled to 2050.

'human appropriation of net primary product' (HANPP) as a proxy for intensity of land-use and 'roads' as a proxy for habitat fragmentation. But such variables are likely to only partially capture changes in habitat quality. As discussed in Section 1, the only alternative is to infer changes in quality from MSA changes, but this requires (1) mapping changes in MSA to changes in ESS provision and (2) valuing changes in ESS. We argue that the evidence base from the scientific literature to support these inferences is limited. But the outcome of this methodological choice is that the approach in our study is likely to systematically under-value changes in habitat quality.

(2) Aside from the results for carbon storage (which are treated as preliminary and indicative, and are not available for all change scenarios) we do not attempt value transfer across ESS categories. We use valuation estimates from primary studies once screened for methodological integrity, specificity of study area etc. But most data points in the valuation database are for study sites where only some subset of ESSs has been valued (see Section 1 for further analysis). Since these site-level values are thus only partial (but are the ones used in the valuation database) this implies a systematic under-valuing of benefits. This second issue of omitted values for ESS is generic to environmental valuation studies and to site-level benefits transfer (as opposed to ESS-level transfer).

(3) Our value estimations for the change scenarios are based on changes to only three terrestrial biomes (temperate forest and woodland; tropical forest; and grasslands). It is very likely that there are significant value changes to other biomes.

In summary, these three factors contribute to a systematic under-representation of benefit estimates (although it is conceivable that value changes for land-use are positive for the three biomes tested and negative for those omitted). Thus we contend that if benefits exceed costs then this is a *sufficient* condition to infer economic efficiency, but it costs exceed benefits we should be careful not to over-state the confidence we have in such outcomes.

Section 12.2 considers the results for carbon storage.

12.2 Change scenarios: changes in net carbon storage

We treat net carbon storage[67] as a separate ESS as benefits are truly global in nature and do not rely on an appraisal of local socio-ecological conditions to the same extent as other ESSs (*c.f.* Barbier *et al.*, 2008). Further, there is a well-established literature on the valuation of changes in carbon storage to 2050. There are various databases of values for carbon to 2030/2050. In our cost benefit analysis we use alternative measures of the value of carbon to provide some sensitivity. These are the Social Cost of Carbon (SCC) estimated by Defra (2007), the POLES and RICE models[68]

The IMAGE-GLOBIO models consider changes in three main sources: (1) deforestation, (2) re-growth of vegetation, and (3) increased carbon sequestration by existing forests (fertilisation).The change scenarios generally impact upon all three. For instance, a reduction

[67] The modelled results are for net carbon flux from biosphere to atmosphere; we use the negative of these values as a measure of net additional carbon storage within the biosphere.

[68] For details of the Poles model see: http://webu2.upmf-grenoble.fr/iepe/textes/POLES8p_01.pdf. For a further discussion of the RICE model see: Nordhaus, W.D. and Z. Yang (1996), 'A Regional Dynamic General-Equilibrium Model of Alternative Climate Change Strategies'. *American Economic Review*, 886, p.741-765.

in agricultural land reduces deforestation, increases the re-growth of vegetation and affects sequestration by increasing the area of natural forest, but also by reducing atmospheric CO_2 concentrations. These are summarised in Table 63.

The analysis of net carbon storage changes that are modelled by the IMAGE-GLOBIO team for a selection of change scenarios[69] might be treated as adjunct and preliminary values or co-benefits; the results are presented at an aggregated level (global) in our study[70]. The bio-physical results are presented in Table 64. Table 65 provides values for the bio-physical changes based on the SCC.

Table 63 Alternative carbon values for emissions in years out to 2050 (US$2007 per tonne CO_2e)

	POLES	SCC	RICE High	RICE Low
2000		29		
2005	0	32	10	5
2010	8	34	13	7
2015	11	38	17	9
2020	37	42	22	10
2025	59	47	28	12
2030	107	53	35	13
2035	182	63	41	14
2040	256	74	48	16
2045	331	93	55	18
2050	406	112	61	21

[69] Results at global scale, at five year intervals, were provided for change scenarios 1 (agricultural productivity), 2 (post-harvest food production and distribution), 3 (forest management), 4 (protected areas), 5 (Reduced Deforestation), 7 (global dietary patterns). It is our understanding that no carbon storage analysis was carried out for the climate change with/without bio-energy options, and results for trade liberalisation are not available.

[70] The IMAGE/GLOBIO results for carbon storage were only available close to the report submission deadline. Given time scales, we have not been able to review the full model outputs with regards carbon storage changes and thus we treat these results as indicative and preliminary. It is our understanding that the global results are derived from IMAGE/GLOBIO projections at grid-cell level and thus we propose further valuation analysis in any follow-up to the QA study as our results are presented at global level.

Table 64 Modelled projections of changes in net carbon storage relative to the baseline (billion tonnes CO_2-equivalent)

	Agricultural Productivity (high AKST)	Agricultural Productivity (no AKST)	Forest Management	Protected Areas (20%)	Protected Areas (50%)	REDD	Healthy Diet (no meat)	Healthy Diet (Willett diet)
2000	0.00	0.00	0.56	-0.29	-0.22	0.71	0.00	0.29
2005	-0.57	-3.19	-0.03	-0.04	0.20	6.34	0.00	-0.12
2010	0.04	-3.96	0.25	-0.02	0.14	6.71	0.08	-0.03
2015	1.39	-6.64	0.16	0.66	2.90	9.16	2.80	-0.21
2020	0.96	-7.64	0.89	0.49	3.33	8.65	2.88	0.12
2025	2.49	-7.69	1.32	0.31	4.24	8.05	5.50	1.47
2030	2.41	-8.91	1.55	0.76	4.27	6.97	8.80	4.40
2035	3.05	-5.50	1.42				9.89	6.20
2040	2.85	-9.63	1.21				10.99	7.34
2045	3.70	-8.10	1.52				8.42	5.95
2050	3.74	-9.28	1.08				5.06	3.63
Total (all years)	**90.88**	**-330.41**	**45.60**	**8.18**	**64.29**	**216.53**	**260.81**	**135.81**

Table 65 Values of changes in net carbon storage relative to the baseline (billion 2007 US$ based on Defra SCC)

	Agricultural Productivity (high AKST)	Agricultural Productivity (no AKST)	Forest Management	Protected Areas (20%)	Protected Areas (50%)	REDD	Healthy Diet (no meat)	Healthy Diet (Willett diet)
2000	0.00	0.00	16.19	-8.46	-6.39	20.58	8.48	0.00
2005	-17.92	-100.56	-1.00	-1.38	6.43	199.84	-3.81	0.00
2010	1.25	-135.24	8.64	-0.64	4.80	229.14	-1.00	2.63
2015	52.94	-252.95	5.99	25.28	110.39	348.81	-7.82	106.57
2020	40.23	-321.03	37.62	20.54	139.81	363.39	5.24	121.14
2025	117.56	-363.59	62.46	14.56	200.73	380.69	69.35	260.25
2030	126.38	-468.14	81.37	39.69	224.38	366.22	230.99	462.36
2035	192.57	-346.54	89.34				390.93	623.49
2040	209.57	-707.99	89.27				540.23	808.59
2045	343.97	-752.71	140.97				552.95	783.05
2050	420.75	-1043.01	121.74				408.39	568.69
Total (all years)	**6342.82**	**-19788.46**	**2912.84**	**367.10**	**2843.97**	**8655.67**	**9922.40**	**17292.95**

12.3 Overall assessment: structure

We treat each change scenario in turn in the sections that follow. Whereas the results for carbon storage (change scenario versus baseline) are available as values of carbon stored per annum at five year intervals, the results presented in this study for the analysis of changes in land-use pertain to one (and only one) interval, *viz*. 2000 to 2030 or 2000 to 2050 depending on the change scenario, with no intermediate points estimated. In the absence of any defensible alternative, we assume a linear trajectory of benefits from 2000 to 2030 or 2050. This is clearly a simplification.

The trajectory is of course important with regards to discounting. We present aggregated results at 0%, 1% and 4% discount rates (DRs). If cost estimates are available, the benefit-cost ratio is obtained by applying the same DR to costs as to benefits. The choice of positive DRs (1% and 4%) is relatively arbitrary[71]; although these rates were applied in the COPI study (Braat and ten Brink, 2008).

We present results for changes in land-use values in the sections that follow in the following form:

1. Results for change scenarios from the bio-physical analysis: percentage change in extent of the three biomes being valued (grassland, temperate forest and tropical forest) comparing the 2030 or 2050 BAU baseline and the 2030 or 2050 option scenario.

   We are valuing the *marginal* benefits of each option-scenario. So for example +10% for the grasslands biome for the OECD region implies that there is estimated to be 10% more grassland in this region in 2030/2050 *with* the option scenario as compared to without, i.e. the BAU. Whether grasslands in the OECD increase or decrease between 2000 and 2030/2050 under BAU is not shown in the figures that follow, although this does form part of the discussion vis-à-vis marginality and the applicability of our results.

2. A ‘value map’ of changes in land-use (change scenario at 2030 or 2050 versus the baseline at 2030 or 2050) presented in terms of present value of changes discounted at 1%. (Note that the value ranges in the legends pertaining to each value map differ between change scenarios.)

3. Results for value changes disaggregated by biome and IMAGE region.

4. A table showing the specific values for changes in land use for each IMAGE region, with 0%, 1% and 4% discount rates.

We then provide a synopsis of the overall results (values for land-use change; carbon storage change where available and costs where available) and a discussion in each case. An overall discussion of results and accompanying sensitivity analysis then follows in Section 13.

12.4 Agricultural productivity

Based on Rosegrant *et al.* (2009), Van Vuuren *et al.* (2009) and FAO (2006), the baseline assumes that the current levelling-off of agricultural productivity growth persists: productivity

[71] See Markandya *et al*. (2001) for a discussion of appropriate discount rates for long term policy assessment.

growth is projected to be 0.64% growth per year, or cumulated growth in productivity of 25.6% to 2050 relative to productivity in 2000.

12.4.1 Agricultural productivity – high AKST

Under the change scenario, productivity growth is spurred by investment in Agricultural Knowledge, Science and Technology (AKST), increasing productivity growth by 40% and 20% for crop and livestock respectively, relative to the baseline.

Figure 19 presents the percentage changes in land-use for each biome under the high AKST scenario relative to the baseline. There are increases in the area of each biome in each region with the exception of small reductions in temperate forest in the 'Middle East and North Africa' and in grasslands in 'Russia and Central Asia', the latter being due to an expansion of arable cropping in Central Asia into previously unsuitable areas (PBL, 2010). Most of these changes are below 10% and so might be described as marginal. The approximately 40% increase in grassland area in 'South Asia' counteracts a 25% decline in that biome under the baseline relative to the 2000 base year; the increase is thus a more modest 5% when compared to the 2000 situation.

A value map by region is presented for the high AKST scenario in Figure 20. These results are also presented by region and by biome (Table 66); with the overall aggregated results from 2000 to 2050 at three discount rates for the high AKST scenario (Table 67).

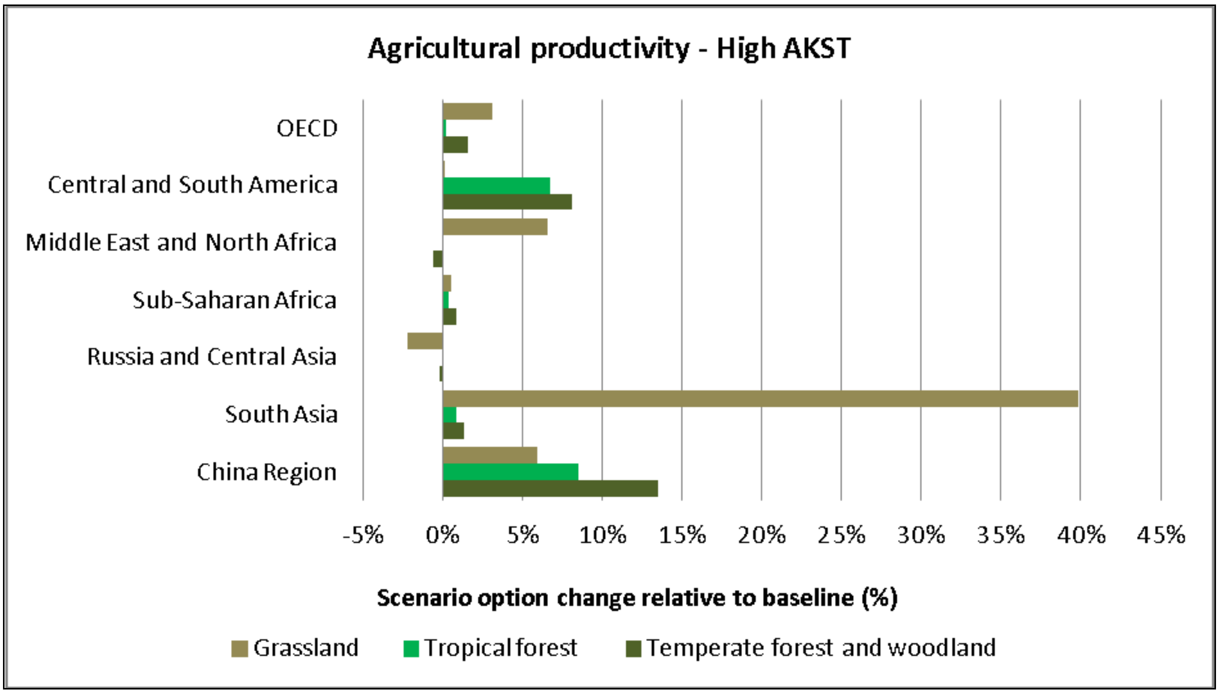

Figure 19 Agricultural productivity: change in area of biomes for high AKST scenario option relative to the baseline

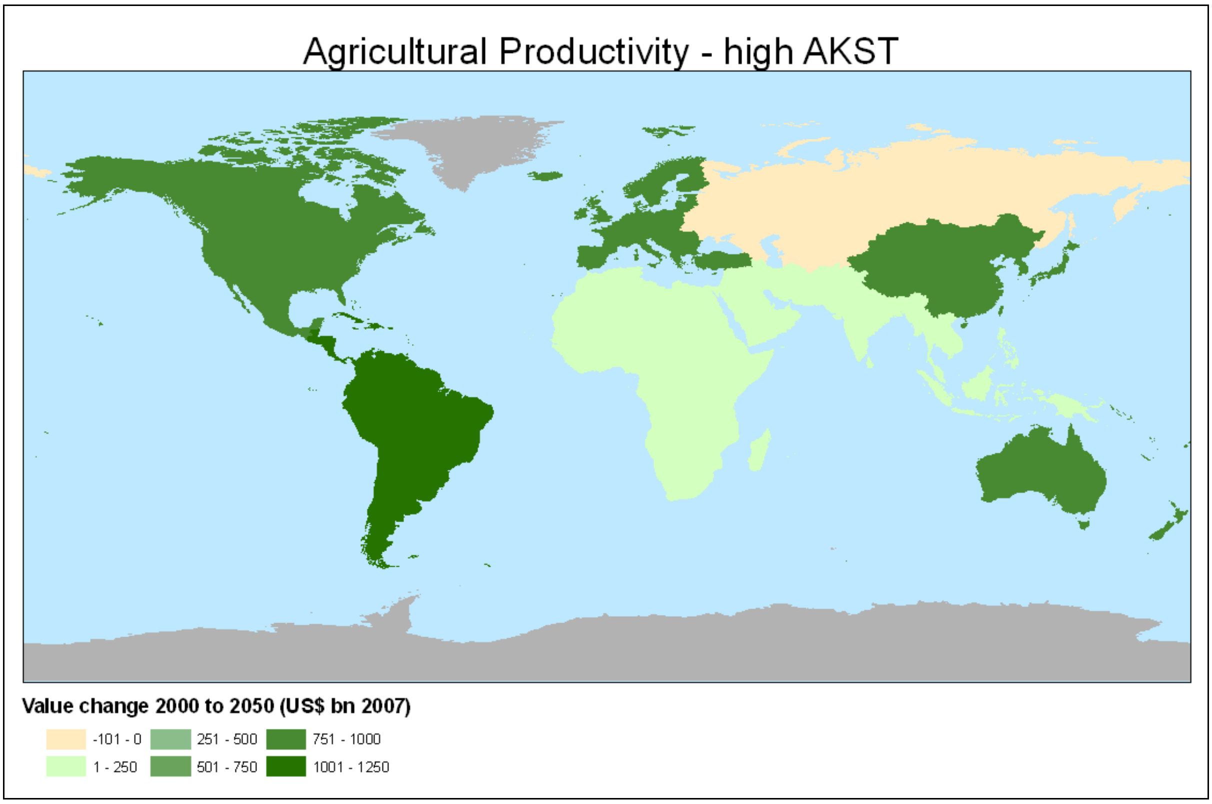

Figure 20 Agricultural productivity (high AKST): map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Note that Table 66 shows the breakdown on a biome-by-biome basis. Thus the sum of the value changes across the three biomes (27.8 billion 2007 US$ for grasslands; 81.7 billion 2007 US$ for temperate forest; 52.6 billion 2007 US$ for tropical forests) are summed in the summary table (Table 67, second column, i.e. 162.1 billion 2007 US$). This presentational style is repeated for all the change scenarios below.

Further, the changes in the extent of the three biomes do not 'cancel each other out' in Table 66. This is because terrestrial ecosystems under IMAGE-GLOBIO consist of more than just these three biomes; see Section 5 for further discussion.

A further point to note is that the annual values for each region cannot be calculated directly from the changes in area and the mean per ha values. This is because the value functions include patch size as an explanatory variable, hence the value per ha varies across patches[72]. The patch size coefficient for each biome is negative indicating that larger patches have lower per ha values.

[72] As an example, assume we have a region with three patches of biome X that are initially 100, 200 and 500 ha in size and per ha values are $400, $300 and $200 respectively. Then if each patch increases by 10% the sum of the individual patch values is (10 x 400) + (20 x 300) + (50 x 200) = $20,000. If we use the total change in patch area and mean per ha values the estimated value would be (10 + 20 + 50) x 300 = $24,000.

Table 66 Agricultural productivity (high AKST): value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	423.7	645.3	20.6
Central and South America	4.7	252.9	0.1
Middle East and North Africa	91.2	326.4	2.4
Sub-Saharan Africa	35.3	63.6	0.3
Russia and Central Asia	-128.2	351.2	-3.6
South Asia	511.8	146.2	4.8
China Region	229.4	232.3	3.2
Total	**1167.8**		**27.8**
Temperate Forest			
OECD	155.7	21054.7	26.1
Central and South America	57.0	17673.2	19.1
Middle East and North Africa	-0.4	16464.7	-0.1
Sub-Saharan Africa	2.4	8135.5	0.2
Russia and Central Asia	-18.0	18170.2	-1.9
South Asia	6.6	9787.0	1.4
China Region	253.6	15765.8	37.0
Total	**456.8**		**81.7**
Tropical Forest			
OECD	1.7	9958.5	0.6
Central and South America	420.9	8308.7	45.9
Middle East and North Africa			
Sub-Saharan Africa	21.1	4015.4	0.8
Russia and Central Asia			
South Asia	20.8	7593.5	3.4
China Region	7.1	8502.5	1.7
Total	**471.7**		**52.6**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 67 Annual and discounted aggregated regional benefits (billions 2007 US$) of agricultural productivity increase (high AKST) versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	47.3	1207.1	869.7	362.3
Central and South America	65.1	1660.6	1196.4	498.4
Middle East and North Africa	2.3	58.3	42.0	17.5
Sub-Saharan Africa	1.3	33.1	23.9	9.9
Russia and Central Asia	-5.5	-140.3	-101.1	-42.1
South Asia	9.6	244.9	176.4	73.5
China Region	41.9	1069.6	770.6	321.0
Total	**162.1**	**4133.2**	**2977.9**	**1240.4**

12.4.2 Agricultural productivity – no AKST

The results above pertain to the baseline above (i.e. 0.64% growth per year). PBL (2010) also model an alternative baseline for this change scenario wherein there is a decline in productivity growth (as opposed to the levelling-off of productivity growth at current levels). For completeness we present the results for this alternative counter-factual, i.e. decline versus the BAU. We do not discuss these results further as this appears to be an extreme counter-factual.

Figure 21 presents the percentage changes in area for each biome under the more extreme scenario of no investment in AKST. Here we observe large changes in biome extent particularly in 'Sub-Saharan Africa' and 'South Asia'. A value map by region is presented for the high AKST scenario in Figure 22. These results are also presented by region and by biome (Table 68); with the overall aggregated results from 2000 to 2050 at three discount rates for the high AKST scenario (Table 69).

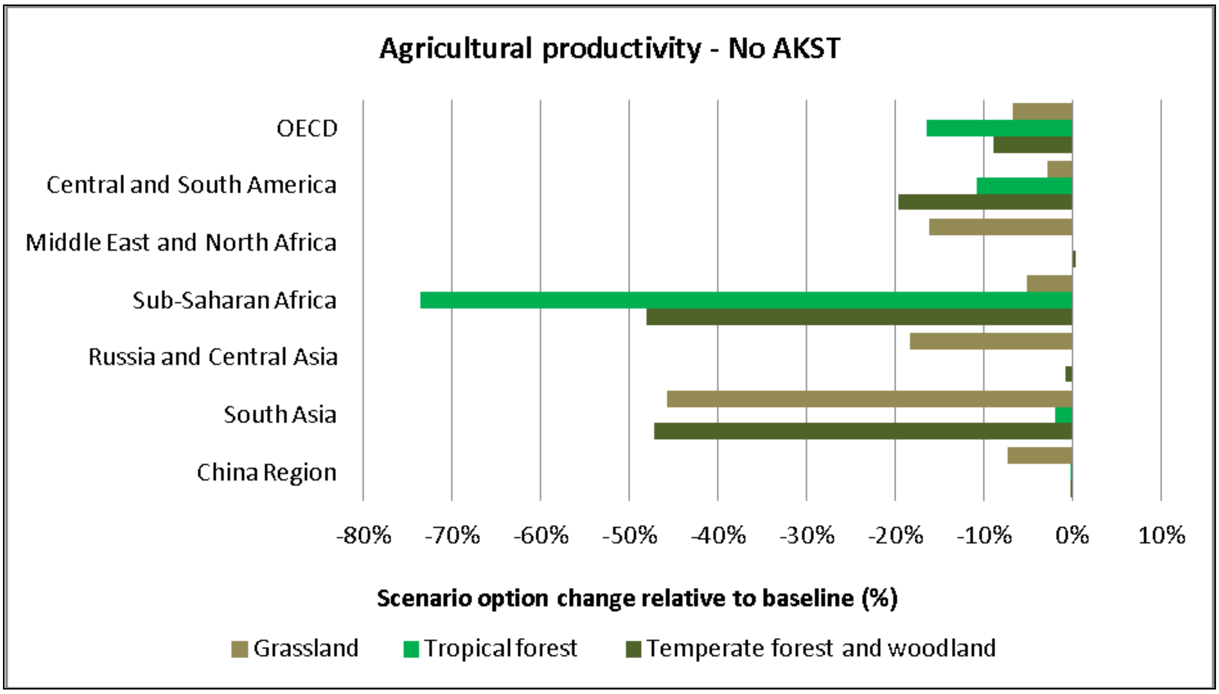

Figure 21 Agricultural productivity: change in area of biomes for no AKST scenario option relative to the baseline

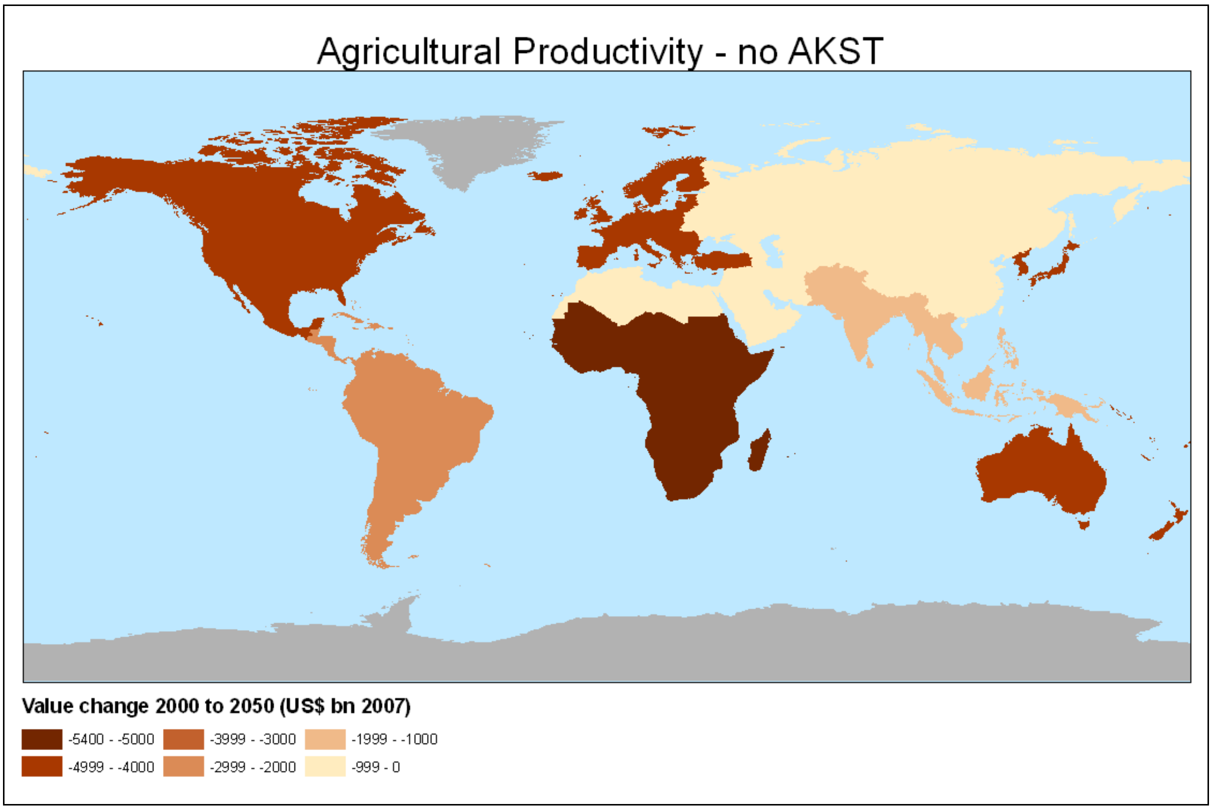

Figure 22 Agricultural productivity (no AKST): map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 68 Agricultural productivity (no AKST): value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-926.5	646.3	-45.1
Central and South America	-125.0	253.0	-2.2
Middle East and North Africa	-223.6	327.6	-6.0
Sub-Saharan Africa	-340.4	63.6	-2.5
Russia and Central Asia	-1046.3	352.1	-29.6
South Asia	-587.3	148.2	-5.6
China Region	-279.6	232.8	-3.9
Total	**-3528.7**		**-95.0**
Temperate Forest			
OECD	-850.5	21539.1	-145.9
Central and South America	-137.8	18812.0	-49.2
Middle East and North Africa	0.3	16428.4	0.1
Sub-Saharan Africa	-138.1	9427.1	-12.4
Russia and Central Asia	-65.9	18192.1	-6.9
South Asia	-234.0	11300.5	-55.9
China Region	-2.3	16185.5	-0.3
Total	**-1428.3**		**-270.4**
Tropical Forest			
OECD	-144.7	10054.3	-54.8
Central and South America	-675.0	8472.6	-76.4
Middle East and North Africa			
Sub-Saharan Africa	-4674.3	5351.0	-279.0
Russia and Central Asia			
South Asia	-49.5	7626.3	-8.2
China Region	0.0	8605.2	0.0
Total	**-5543.5**		**-418.4**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 69 Annual and discounted aggregated regional benefits (billions 2007 US$) of agricultural productivity (no AKST) versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	-245.8	-6267.0	-4515.3	-1880.8
Central and South America	-127.8	-3258.4	-2347.6	-977.9
Middle East and North Africa	-5.9	-149.5	-107.7	-44.9
Sub-Saharan Africa	-293.9	-7495.6	-5400.4	-2249.5
Russia and Central Asia	-36.5	-931.7	-671.3	-279.6
South Asia	-69.7	-1776.9	-1280.3	-533.3
China Region	-4.3	-108.9	-78.5	-32.7
Total	**-783.8**	**-19988.0**	**-14401.0**	**-5998.7**

12.4.3 Discussion

Globally, the land-use value change is significantly positive, i.e. around 2977.9 billion 2007 US$ at the 1% discount rate (see Table 67). Note that these gains are against the 'moderate' counter-factual of 0.64% growth in productivity per year, i.e. *not* against the more severe no-AKST option. Using the no-AKST counter-factual would produce much more significant results. The results show that there are significant welfare gains associated with the high AKST scenario across the three biomes; however there are some distributional issues across regions. Specifically the 'Russia and Central Asia' region sees a loss in welfare of 5.5 billion 2007 US$ per annum in 2050; this arises due to an expansion of agricultural production and improved growing conditions in that region (PBL, 2010). This welfare loss reflects a decrease in extent of uncultivated area relative to the baseline, whereas welfare gains in other regions reflect a greater extent of uncultivated areas when compared to the baseline.

Given the development-focused nature of the change scenario, the IMAGE regions that show the largest benefits from land-use change include 'Central and South America' 'OECD' and 'China region'. These benefits arise largely from increased forest area relative to the baseline across these regions; although there are also substantial benefits from increased grassland area in the 'OECD' region.

Alongside the benefits from land-use change, the estimated net benefit (relative to the baseline) from additional carbon sequestration is valued at is 6342.8 billion 2007 US$ (see Table 65). In Section 9, costs are estimated to be 568.3 billion 2007 US$ at a 1% discount rate. The benefit-cost ratios across a range of discount rates and carbon values are set out in Table 70.

Table 70 Overall benefit-cost ratios for agricultural productivity (high AKST)

		Discount rate		
		0%	1%	4%
Benefits (bn US$2007)				
Change in biome areas		4133.2	2977.9	1240.4
Carbon values (bn US$2007)				
	POLES	18414.3	18414.3	18414.3
	SCC	6342.8	6342.8	6342.8
	RICE high	3843.6	3843.6	3843.6
	RICE low	1384.6	1384.6	1384.6
Costs (bn US$2007)		725.0	568.3	311.5
Benefit/cost ratios				
No carbon value		5.7	5.2	4.0
Social Cost of Carbon		14.4	16.4	24.3
High carbon value (POLES model)		31.1	37.6	63.1
Low carbon value (RICE model low)		7.6	7.7	8.4

Even without adding the additional carbon storage estimated to occur with the change scenario, the benefit/cost ratio is significantly positive, i.e. 4.0 with a higher 4% discount rate. The majority of the benefits from land-use change come from the forest biomes: of the 162.1

billion 2007 US$ undiscounted annual benefit (see Table 67) 27.8 billion is attributed to the grasslands biome (see Table 66), i.e. 17%. This is significant: although one of the changes in the grasslands biome was arguably non-marginal (e.g. ~40% ‘South Asia’), the average change in the forest biomes is 3%, i.e. clearly marginal. Even removing the grasslands benefits (510.3 bn US$) and the carbon benefits leaves a high benefit-cost ratio (~4.3 at 1% discount rate).

We can say with very high confidence that this change scenario is economically efficient on the basis of land-use change alone.

12.5 Reducing post-harvest losses in the food chain

The baseline scenario assumes no change to current practices. The change scenario sets out a reduction of food supply chain wastes and losses by 15% of total supplies, which would roughly correspond to halving the estimated current losses. The changes in biome extent relative to the baseline are illustrated in Figure 23. The changes under the scenario do not exceed 3%, and are thus clearly marginal. Figure 24 presents a value map for changes; Table 71 presents value results by region and by biome; and Table 72 presents annual and discounted aggregated regional benefits.

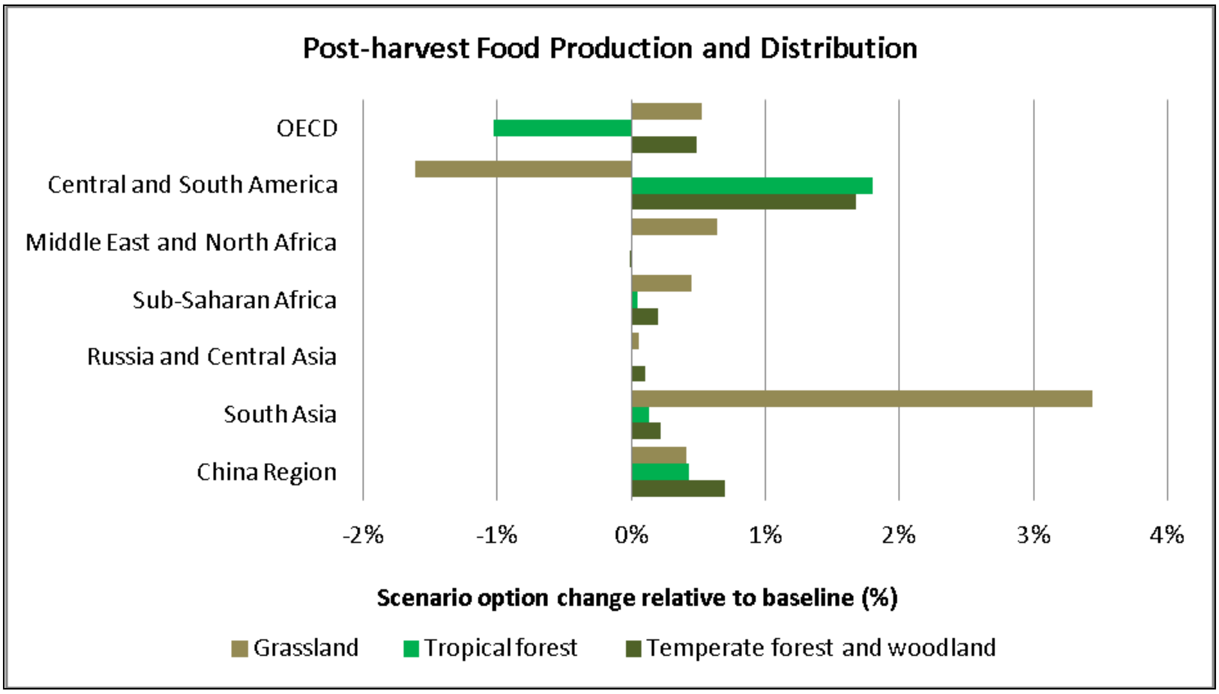

Figure 23 Post-harvest food production and distribution: change in area of biomes for scenario option relative to the baseline

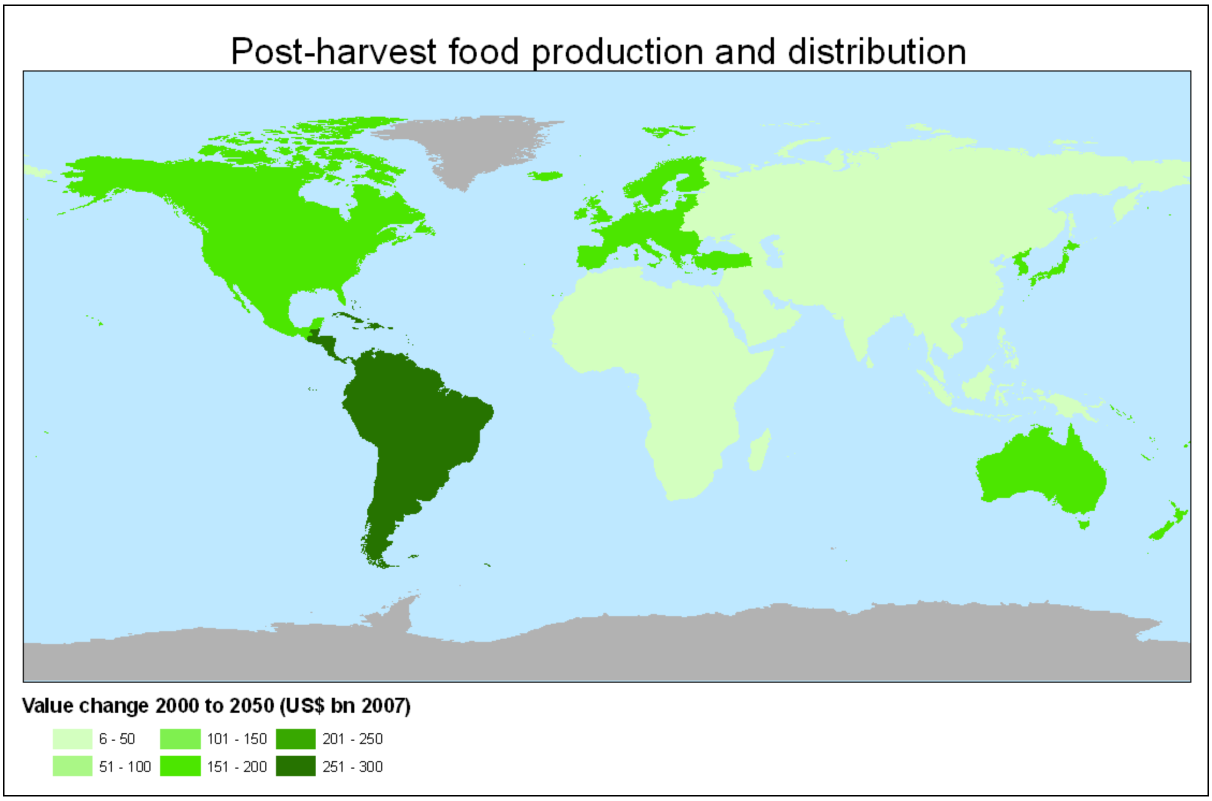

Figure 24 Post-harvest losses: map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 71 Post-harvest losses: value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	73.4	645.2	3.6
Central and South America	-65.5	253.5	-1.2
Middle East and North Africa	9.0	326.6	0.2
Sub-Saharan Africa	31.0	63.5	0.2
Russia and Central Asia	3.4	350.6	0.1
South Asia	50.0	146.3	0.5
China Region	16.0	232.4	0.2
Total	**117.2**		**3.7**
Temperate Forest			
OECD	47.8	20950.4	8.0
Central and South America	12.5	17466.1	4.1
Middle East and North Africa	0.0	16610.5	0.0
Sub-Saharan Africa	0.6	8069.1	0.0
Russia and Central Asia	8.6	18168.2	0.9
South Asia	1.1	9776.2	0.2
China Region	13.4	16083.6	2.0
Total	**84.0**		**15.3**
Tropical Forest			
OECD	-9.0	9962.7	-3.3
Central and South America	121.4	8259.3	13.1
Middle East and North Africa			
Sub-Saharan Africa	3.3	4015.9	0.1
Russia and Central Asia			
South Asia	3.3	7611.5	0.5
China Region	0.4	8586.3	0.1
Total	**119.4**		**10.6**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 72 Annual and discounted aggregated regional benefits of reducing post-harvest losses in the food chain versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	8.2	272.2	196.1	81.7
Central and South America	16.1	412.3	297.1	123.8
Middle East and North Africa	0.2	8.7	6.2	2.6
Sub-Saharan Africa	0.4	16.4	11.8	4.9
Russia and Central Asia	1.0	29.8	21.5	8.9
South Asia	1.3	21.5	15.5	6.5
China Region	2.3	55.9	40.3	16.8
Total	**29.5**	**816.8**	**588.5**	**245.1**

12.5.1 Discussion

There are net benefits arising from this change scenario across all regions, with the largest benefits being observed in the 'OECD' and 'Central and South America' regions. These benefits are largely due to increased forest area relative to the baseline. The effects of this change scenario on carbon flux were not estimated so carbon values for the land-use change are not available.

The change scenario is beneficial but there is no cost estimate available, and as such no benefit/cost ratio.

12.6 Forest management

The baseline scenario assumes that forests continue to be exploited in an unsustainable way through conventional logging practices. In the change scenario conventional selective logging practices are replaced by reduced impact logging (RIL) with a compensating increase in forest plantations. Figure 25 presents the bio-physical changes in grasslands and forests relative to the baseline. These changes are clearly marginal and do not exceed the 1.7% reduction in temperate forest and woodlands observed in 'South Asia'. Figure 26 maps the changes in values by IMAGE region at 1%; Table 73 provides value results by region and by biome; and Table 74 provides results at 0%, 1% and 4%. An important point is that this change scenario is geared towards environmental quality, i.e. improving biodiversity; consequently there is less land cover change for which we can determine welfare changes.

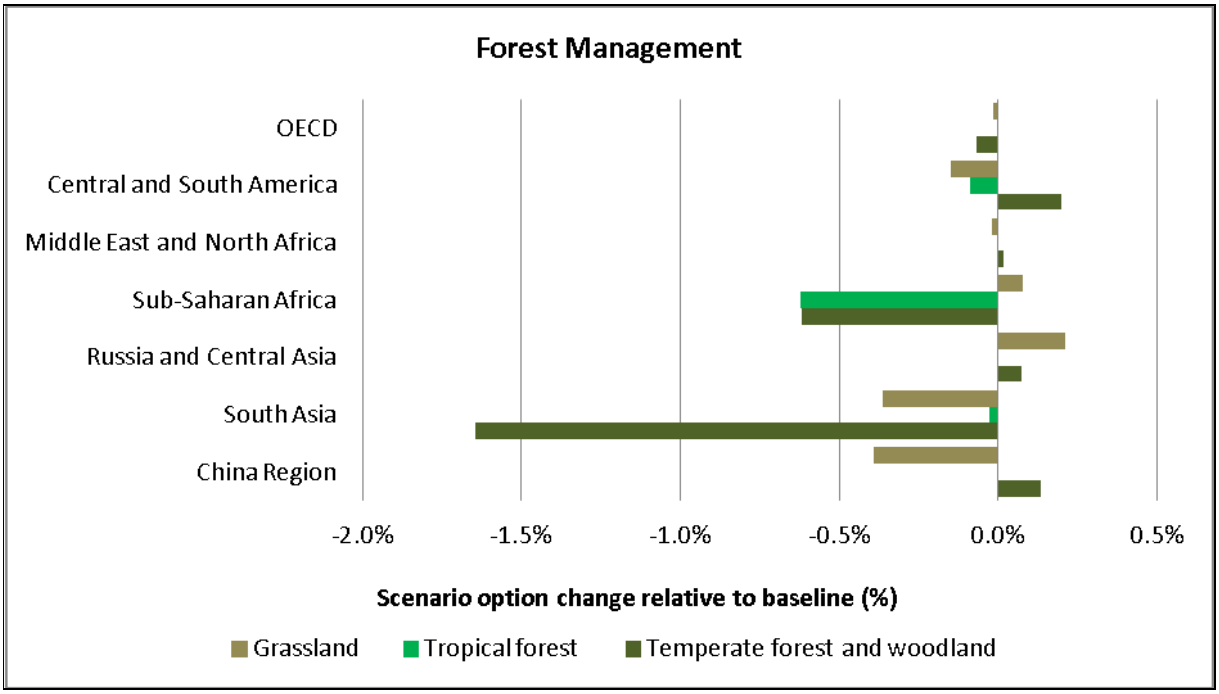

Figure 25 Forest management: change in area of biomes for scenario option relative to the baseline

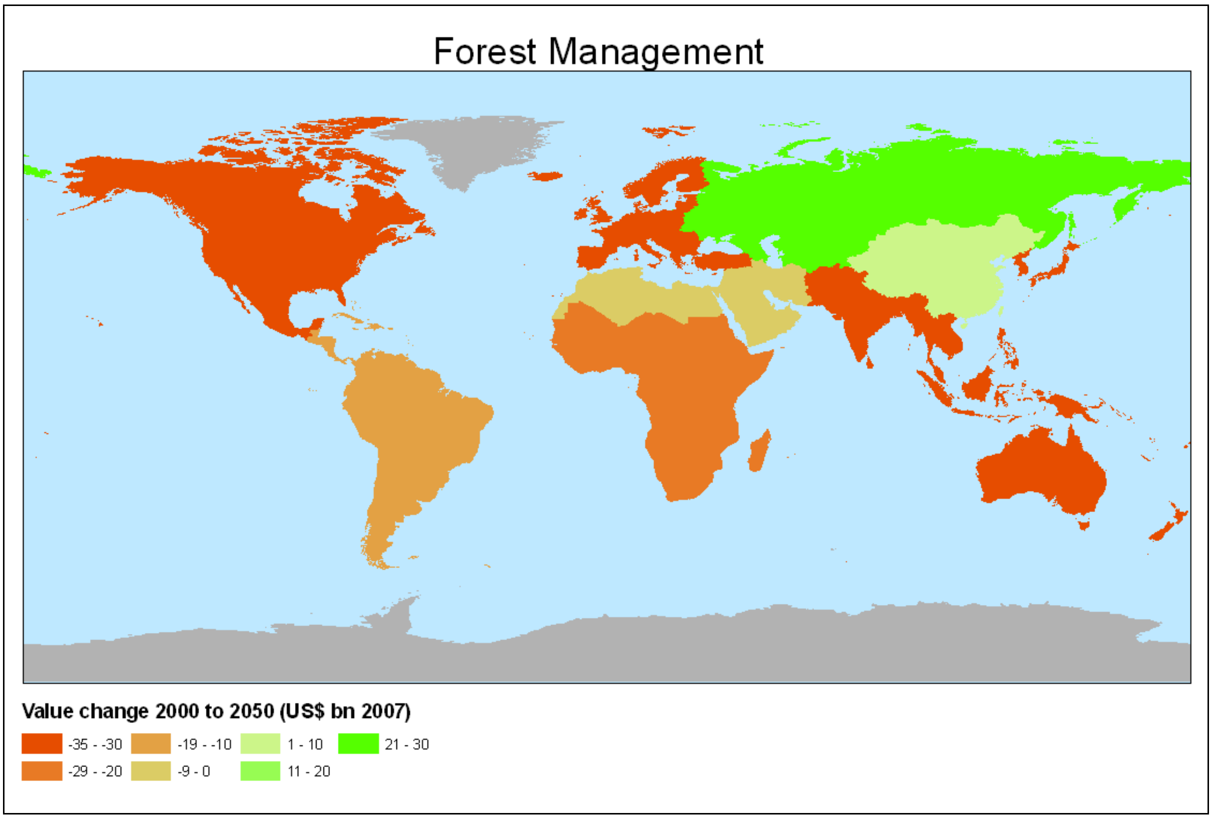

Figure 26 Forest management: map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 73 Forest management: value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-2.1	645.9	-0.1
Central and South America	-6.0	253.5	-0.1
Middle East and North Africa	-0.3	326.6	0.0
Sub-Saharan Africa	6.6	63.1	0.0
Russia and Central Asia	11.9	351.2	0.3
South Asia	-3.4	148.2	0.0
China Region	-15.2	232.5	-0.2
Total	**-8.5**		**-0.1**
Temperate Forest			
OECD	-6.8	21056.7	-1.1
Central and South America	1.6	17041.0	0.5
Middle East and North Africa	0.0	16503.8	0.0
Sub-Saharan Africa	-1.2	9684.4	-0.1
Russia and Central Asia	6.1	18250.8	0.6
South Asia	-7.3	10318.3	-1.6
China Region	2.5	16222.9	0.4
Total	**-5.0**		**-1.3**
Tropical Forest			
OECD	0.0	9962.7	0.0
Central and South America	-6.3	8259.3	-0.7
Middle East and North Africa			
Sub-Saharan Africa	-34.4	4015.9	-1.4
Russia and Central Asia			
South Asia	-0.7	7611.5	-0.1
China Region	0.0	8586.3	0.0
Total	**-41.5**		**-2.2**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 74 Annual and discounted aggregated regional benefits of forest management versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	-1.2	-48.2	-34.7	-14.5
Central and South America	-0.3	-21.7	-15.6	-6.5
Middle East and North Africa	0.0	-0.1	-0.1	0.0
Sub-Saharan Africa	-1.4	-34.4	-24.8	-10.3
Russia and Central Asia	1.0	29.2	21.0	8.8
South Asia	-1.7	-47.2	-34.0	-14.2
China Region	0.2	1.8	1.3	0.5
Total	**-3.6**	**-120.7**	**-86.9**	**-36.2**

12.6.1 Discussion

Table 75 presents the overall benefit/cost ratios for change scenario 4. Summing the costs from land-use change (-86.9 billion 2007 US$ at 1% discount rate) with the net benefits from enhanced carbon sequestration (2912.8 billion 2007 US$ for the SCC estimate) provides overall net benefits, but these must be juxtaposed with the cost estimates from Section 9, set out in Table 38. The benefit/cost ratio only exceeds 1 when using the lower-bound cost estimate and the SCC; the mid-bound costs and the SCC at a 4% discount rate (i.e. costs are discounted below the SCC carbon benefit); or the lower and mid-bound costs when benefits include the highest, POLES, carbon values.

There are likely to be many instances locally where reduced impact logging is in fact less costly than conventional logging and the cost section (Section 9) provides evidence to support this assertion. Further, as described in Section 12.1, our methodology systematically under-states benefits as we only consider benefits that arise from land-use change, not changes in management *within land uses*.

The evidence presented in this study does not support the economic efficiency of the sustainable forestry option with respect to its land-use change impacts alone. Economic efficiency is achievable when including carbon values, but is sensitive to cost assumptions.

Table 75 Overall benefit-cost ratios for forest management

		Discount rate		
		0%	**1%**	**4%**
Benefits (bn US$2007)				
Change in biome areas		-120.6	-86.9	-36.2
Carbon values (bn US$2007)				
	POLES	7817.4	7817.4	7817.4
	SCC	2912.8	2912.8	2912.8
	RICE high	1746.4	1746.4	1746.4
	RICE low	639.8	639.8	639.8
Costs (bn US$2007)				
	Mid	5195.0	4072.5	2232.0
	Lower	2080.0	1630.6	893.7
	Upper	11885.0	9316.9	5106.3
Benefit/cost ratios				
No carbon value	Mid	0.0	0.0	0.0
	Lower	-0.1	-0.1	0.0
	Upper	0.0	0.0	0.0
Social Cost of Carbon	Mid	0.5	0.7	1.3
	Lower	1.3	1.7	3.2
	Upper	0.2	0.3	0.6
High carbon value (POLES model)	Mid	1.5	1.9	3.5
	Lower	3.7	4.7	8.7
	Upper	0.6	0.8	1.5
Low carbon value (RICE model low)	Mid	0.1	0.1	0.3
	Lower	0.2	0.3	0.7
	Upper	0.0	0.1	0.1

12.7 Protected areas

The baseline scenario assumes that the current system of protected areas in maintained, with 14.6% of terrestrial areas having protected status, albeit with differences between eco-regions. No further policy interventions are assumed. The change scenario assumes an increase in protected area coverage to 20% and 50% in 65 identified ecological regions. We treat each sub-option (20% and 50%) in turn below.

12.7.1 Expansion to 20 per cent of each eco-region

Figure 27 presents the bio-physical changes in land-use relative to the baseline. Note that the BAU and change scenarios pertain to 2030 (not 2050) as the bio-physical modelling is to 2030 (PBL, 2010). The area of grassland increases in each region under this scenario; however, these increases are off-set by decreases in forest area in three regions, although land-use change does not exceed 4.2% of biome area in any region compared to the baseline. These reductions in forest are not direct conversions to grassland, but rather result from deforestation to provide land for agriculture following protection of grassland areas. A value map by region is presented in Figure 28; value results by region and by biome are presented in Table 76; and Table 77 presents overall results with three discount rates.

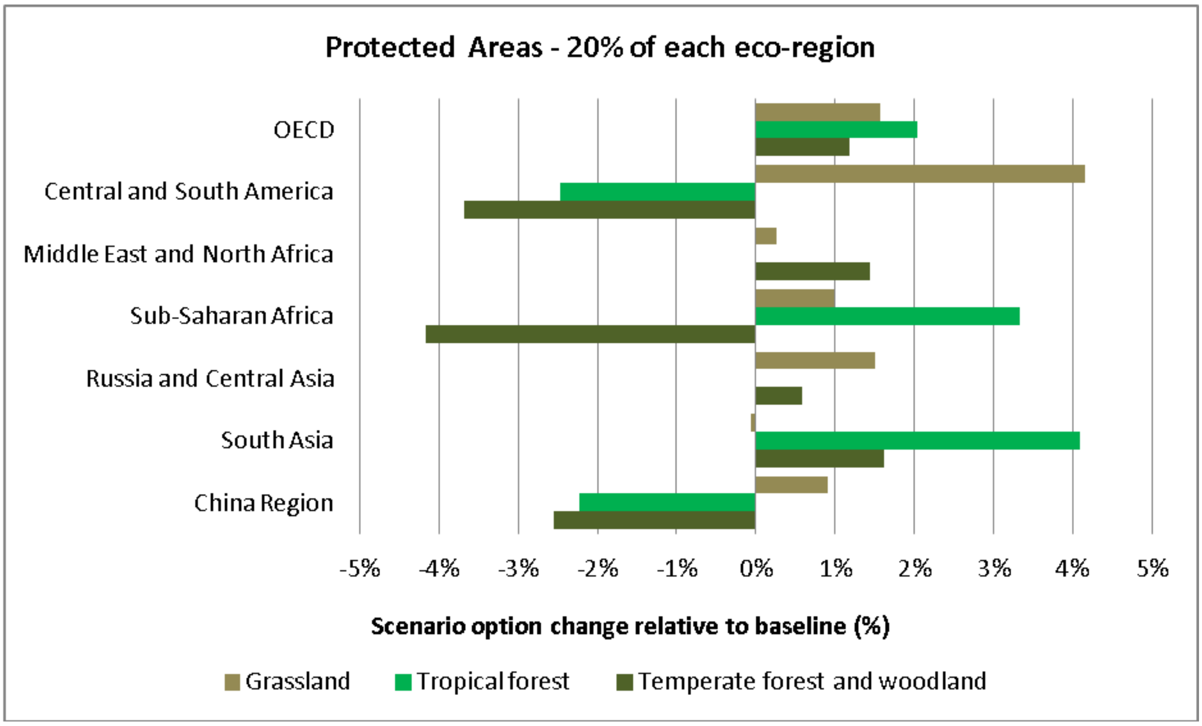

Figure 27 Protected Areas 20%: change in area of biomes for scenario option relative to the baseline

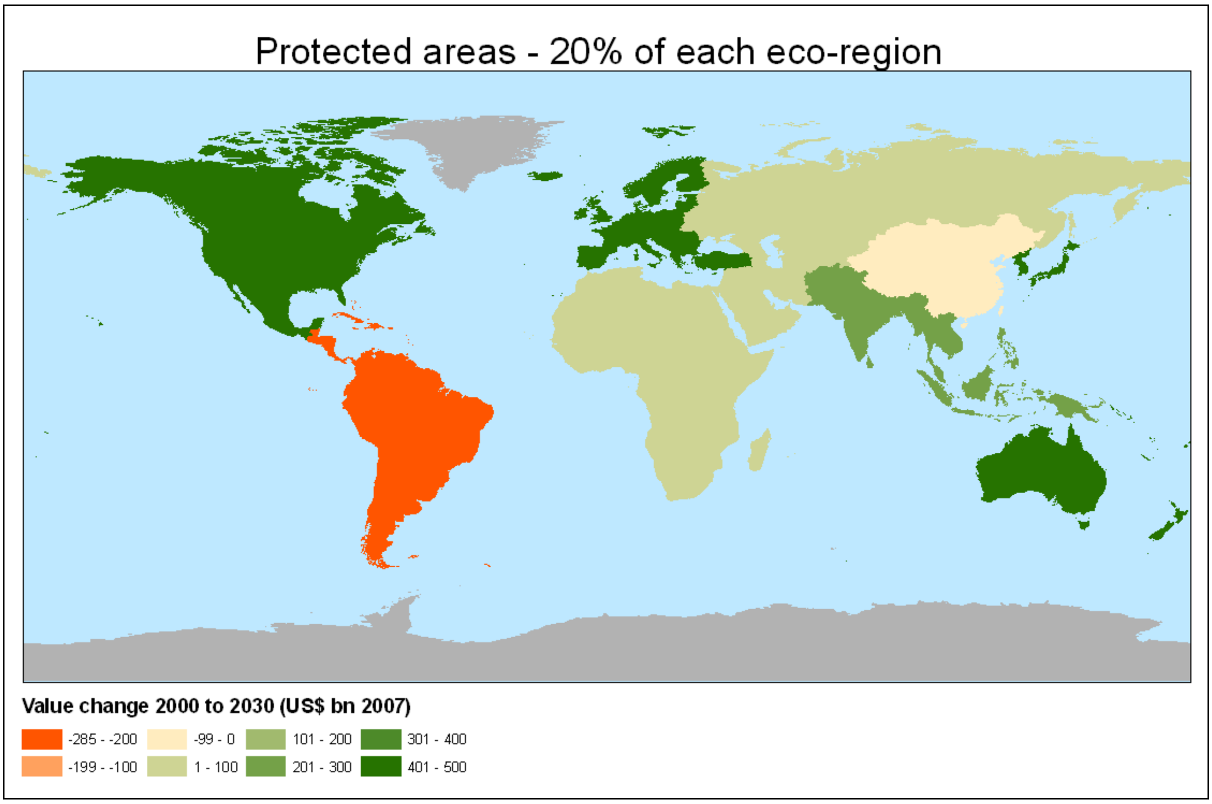

Figure 28 Protected areas 20%: map of value changes 2000 to 2030 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 76 Protected areas 20%: value results by region and by biome relative to 2030 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	210.3	536.5	8.7
Central and South America	182.2	217.1	2.8
Middle East and North Africa	3.9	275.2	0.1
Sub-Saharan Africa	93.1	53.1	0.6
Russia and Central Asia	89.3	295.7	2.2
South Asia	-0.9	120.6	0.0
China Region	31.8	188.4	0.4
Total	**609.7**		**14.7**
Temperate Forest			
OECD	110.8	20132.1	17.8
Central and South America	-27.0	16988.0	-8.7
Middle East and North Africa	1.0	15582.6	0.4
Sub-Saharan Africa	-5.9	10468.4	-0.6
Russia and Central Asia	49.2	17217.6	4.9
South Asia	8.1	9118.5	1.6
China Region	-47.2	15281.9	-6.7
Total	**89.2**		**8.7**
Tropical Forest			
OECD	17.8	9785.8	6.2
Central and South America	-161.3	8063.5	-16.6
Middle East and North Africa			
Sub-Saharan Africa	154.3	4219.7	6.5
Russia and Central Asia			
South Asia	102.9	7071.8	15.3
China Region	-1.8	8104.3	-0.4
Total	**111.9**		**11.0**

Mean per ha values are the average of 2030 baseline and 2030 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.
Note that IMAGE/GLOBIO suggests overall reductions in temperate and tropical forest area, our analysis using the same change factors indicate aggregate gains. This disparity arises from differences in the areas of forest patches and across regions between our respective datasets.

Table 77 Annual and discounted aggregated regional benefits of protect areas 20% versus 2030 baseline

	2030 undiscounted annual benefit	2000 – 2030 discounted total benefit		
		0%	1%	4%
OECD	32.7	506.4	414.7	237.8
Central and South America	-22.4	-347.9	-284.9	-163.4
Middle East and North Africa	0.4	6.8	5.6	3.2
Sub-Saharan Africa	6.5	100.7	82.5	47.3
Russia and Central Asia	7.1	109.4	89.6	51.4
South Asia	16.9	261.6	214.3	122.9
China Region	-6.7	-104.3	-85.4	-49.0
Total	**34.4**	**532.7**	**436.3**	**250.2**

The overall results for value changes from land-use change are positive with the exception of ‘Central and South America’ and ‘China region’, and there are wide variations regionally As indicated in Table 76 the benefits from this policy arise mainly from increases in the area of the grassland biome (634,000 km^2) this compares to a more modest, although valuable increases in temperate forest (107,000 km^2) and tropical forests (114,900 km^2). The large contribution of grassland values to the total value (43%) arises largely from the predominance of increases in grassland areas across the majority of regions. In contrast there is a greater degree of balancing of gains and losses for the forest biomes particularly with respect to values.

Table 78 presents overall benefit/cost results for the PA 20% option. Taking the mid-range value for costs (Option 3 for PAs reported in Section 9.5), the benefit/cost ratio is 1.1 at 1% discount rate, and with the SCC the ratio is 2.0. However the benefit/cost ratio is <1 for the upper-range cost estimate (0.5 with SCC included for 1% discount rate).

The issue of the distribution of winners and losers regionally should also be considered, particularly with respect to the large losses estimated for ‘Central and South America’. A case can be made for promoting the PA 20% option on global welfare grounds with compensation to such affected regions. This is particularly the case for this option as a main driver of PA establishment is biodiversity conservation, whereas the value estimates derived in this study are focused on a wider range of ESSs. The protected status of a site will alter the mix of ESS it provides, i.e. fewer provisioning services but more supporting services, with perhaps more or less regulating and cultural services depending on context. We are unable to ‘unpick’ the relative values of ESS for different sites as out value functions implicitly assume an ‘average’ level of ESS provision. The fact that the benefits estimated for this PA option scenario in terms of land-use change are significantly larger than costs for the mid-range estimate is thus significant.

Table 78 Overall benefit-cost ratios for protected areas 20%

		Discount rate		
		0%	1%	4%
Benefits (bn US$2007)				
Change in biome areas		532.7	436.3	250.2
Carbon values (bn US$2007)				
	POLES	395.4	395.4	395.4
	SCC	367.1	367.1	367.1
	RICE high	211.1	211.1	211.1
	RICE low	94.0	94.0	94.0
Costs (bn US$2007)				
	Mid	465.1	400.4	269.0
	Upper	1995.7	1717.1	1151.3
Benefit/cost ratios				
No carbon value	Mid	1.1	1.1	0.9
	Upper	0.3	0.3	0.2
Social Cost of Carbon	Mid	1.9	2.0	2.3
	Upper	0.5	0.5	0.5
High carbon value (POLES model)	Mid	2.0	2.1	2.4
	Upper	0.5	0.5	0.6
Low carbon value (RICE model low)	Mid	1.3	1.3	1.3
	Upper	0.3	0.3	0.3

12.7.2 Expansion to 50 per cent of each eco-region

This sub-section considers the expansion to 50% of each eco-region, with the study period being 2000 to 2030. Figure 29 presents bio-physical changes relative to the baseline. As would be expected from this more radical departure from BAU (compared to the PA 20% option), several regions show changes which are clearly non-marginal. As with the 20% option there are increases in the area of grassland across all regions of between 1% and 26%. Again, these are frequently off-set by reductions in forest area of as much as 33%. However, an increase in temperate forest and woodlands of 38% is observed in 'Sub-Saharan Africa'.

A value map by region is presented in Figure 30; value results by region and by biome are presented in Table 79; and Table 80 presents overall results with three discount rates.

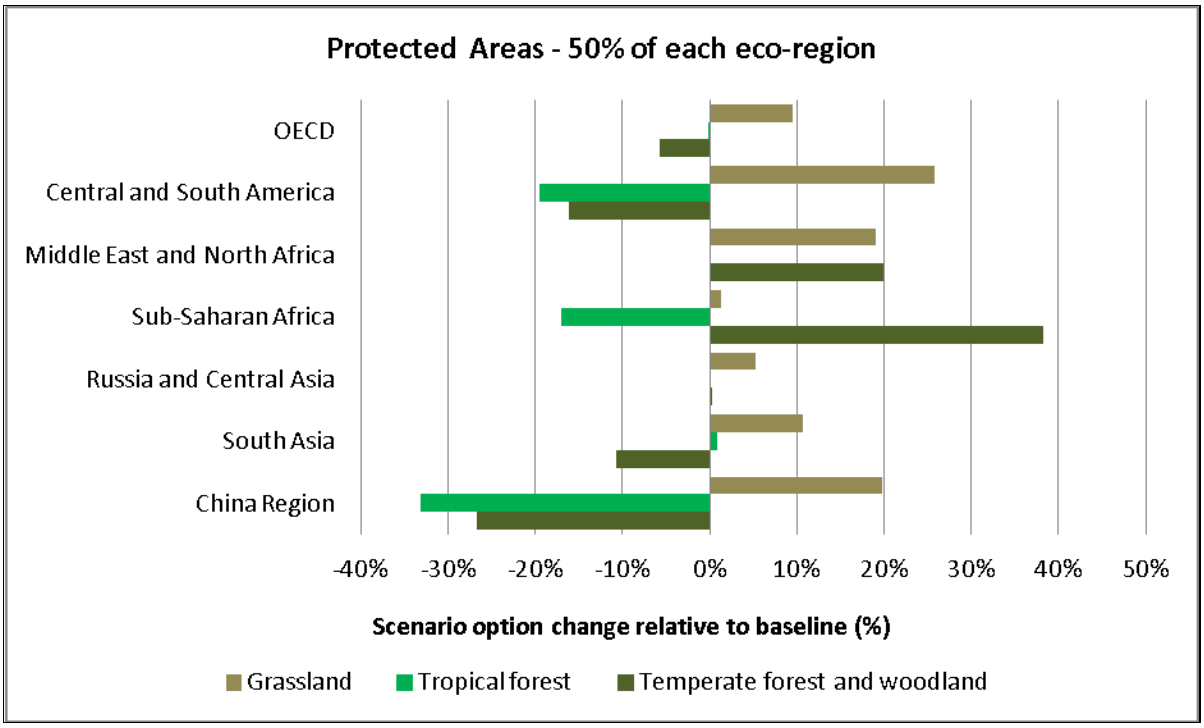

Figure 29 Protected Areas 50%: change in area of biomes for scenario option relative to the baseline

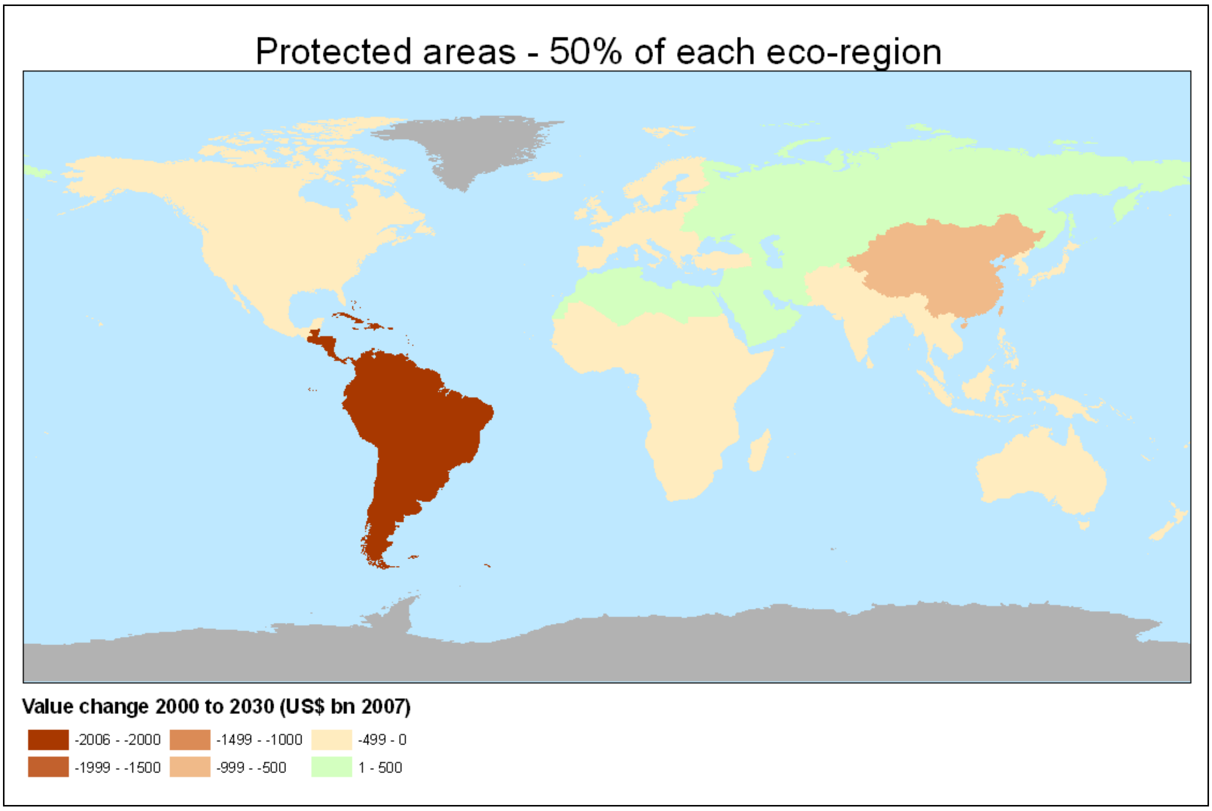

Figure 30 Protected areas 50%: map of value changes 2000 to 2030 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 79 Protected areas 50%: value results by region and by biome relative to 2030 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	1278.0	535.9	52.7
Central and South America	1135.3	216.5	17.6
Middle East and North Africa	277.1	274.5	6.3
Sub-Saharan Africa	124.6	53.1	0.8
Russia and Central Asia	316.4	295.5	7.7
South Asia	169.0	120.4	1.3
China Region	692.9	187.9	7.9
Total	**3993.2**		**94.1**
Temperate Forest			
OECD	-543.4	20435.5	-88.8
Central and South America	-118.6	17499.9	-39.4
Middle East and North Africa	14.2	15058.7	4.7
Sub-Saharan Africa	53.6	9723.0	5.0
Russia and Central Asia	18.6	17230.7	1.8
South Asia	-53.9	9369.6	-10.6
China Region	-493.8	16246.1	-74.3
Total	**-1123.3**		**-201.5**
Tropical Forest			
OECD	-1.2	9798.2	-0.4
Central and South America	-1271.1	8247.3	-136.3
Middle East and North Africa			
Sub-Saharan Africa	-785.6	4424.9	-35.1
Russia and Central Asia			
South Asia	19.8	7110.8	3.0
China Region	-27.4	8576.0	-6.8
Total	**-2065.6**		**-175.7**

Mean per ha values are the average of 2030 baseline and 2030 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 80 Annual and discounted aggregated regional benefits of protect areas (50%) versus 2030 baseline

	2030 undiscounted annual benefit	2000 – 2030 discounted total benefit		
		0%	1%	4%
OECD	-36.6	-566.7	-464.1	-266.1
Central and South America	-158.0	-2449.3	-2005.9	-1150.2
Middle East and North Africa	11.0	169.8	139.1	79.7
Sub-Saharan Africa	-29.4	-455.7	-373.2	-214.0
Russia and Central Asia	9.5	147.7	120.9	69.3
South Asia	-6.3	-98.1	-80.3	-46.1
China Region	-73.3	-1135.9	-930.3	-533.4
Total	**-283.1**	**-4388.3**	**-3593.8**	**-2060.6**

The fact that changes in the biomes are non-marginal implies that the methodology is not reliable with respect to determining benefits. Notwithstanding this, the land-use change values indicate a substantial loss in welfare. For instance, net benefits in 2007 US$ at the 1% discount rate are 436.3 billion for PA20% compared to a 3593.8 billion loss for PA50%. The benefit cost test is therefore unnecessary, but we set this out in Table 81 for completeness.

Table 81 Overall benefit-cost ratios for protected areas 50%

		Discount rate		
		0%	1%	4%
Benefits (bn US$2007)				
Change in biome areas		-4388.3	-3593.8	-2060.6
Carbon values (bn US$2007)				
	POLES	3063.1	3063.1	3063.1
	SCC	2844.0	2844.0	2844.0
	RICE high	1582.8	1582.8	1582.8
	RICE low	699.1	699.1	699.1
Costs (bn US$2007)				
	Mid	1853.8	1595.1	1069.5
	Upper			
Benefit/cost ratios		7972.9	6859.0	4596.5
No carbon value	Mid			
	Upper	-2.4	-2.3	-1.9
Social Cost of Carbon	Mid	-0.6	-0.5	-0.4
	Upper	-0.8	-0.5	0.7
High carbon value (POLES model)	Mid	-0.2	-0.1	0.2
	Upper	-0.7	-0.3	0.9
Low carbon value (RICE model low)	Mid	-0.2	-0.1	0.2
	Upper	-2.0	-1.8	-1.3

Although there are co-benefits to establishing PAs, the extension of protected areas to 50% of eco-regions turns out to be inferior in terms of economic efficiency compared to the 20% option. The *significant* caveat that we would apply in this case is that our analysis does not fully capture biodiversity loss *per se*. As such, PA 50% may conceivably be economically optimal if the benefits from biodiversity conservation out-weigh costs, and it is plausible that the extra biodiversity conservation benefits (comparing 20% versus 50%) are very large.

Notwithstanding this, the PA20% option is economically efficient for our lower bound cost estimate even without adding the core benefits of biodiversity conservation.

12.8 Reduced deforestation (REDD variant)

The baseline scenario assumes no additional actions compared to current standards: in short, deforestation and forest degradation continue due to additional pressures of population and economic growth, with subsequent land-use change for agriculture and logging practices. The change scenario assumes the protection of all forests and woodlands from agricultural expansion. Note that the study period based on the IMAGE-GLOBIO modelling is 2000 to 2030 and all calculations are made to 2030; the analysis is fundamentally the same as is applied to those change scenarios that are modelled to 2050.

Figure 31 presents changes in the extent of grassland and forest biomes for the reduced deforestation change scenario relative to the baseline. The changes in land use for this scenario option are largely marginal across the seven regions and average 2.7% and do not exceed 8.7%. The exception to this is 'Sub-Saharan Africa' where changes range from a loss of 15.8% of grassland to increases of 34% and 62.5% of tropical forests respectively. In absolute terms these changes are large for grassland and tropical forest (see Table 82) and reflect an increased conversion of grassland to cultivation and comparable preservation of forest relative to the baseline. The large percentage change for temperate forest and woodland relates to a relatively small change in physical area.

A value map by region results showing breakdowns of value changes and by biome is presented in Figure 32; value results by region and by biome are presented in Table 82; and Table 83 presents overall results with three discount rates.

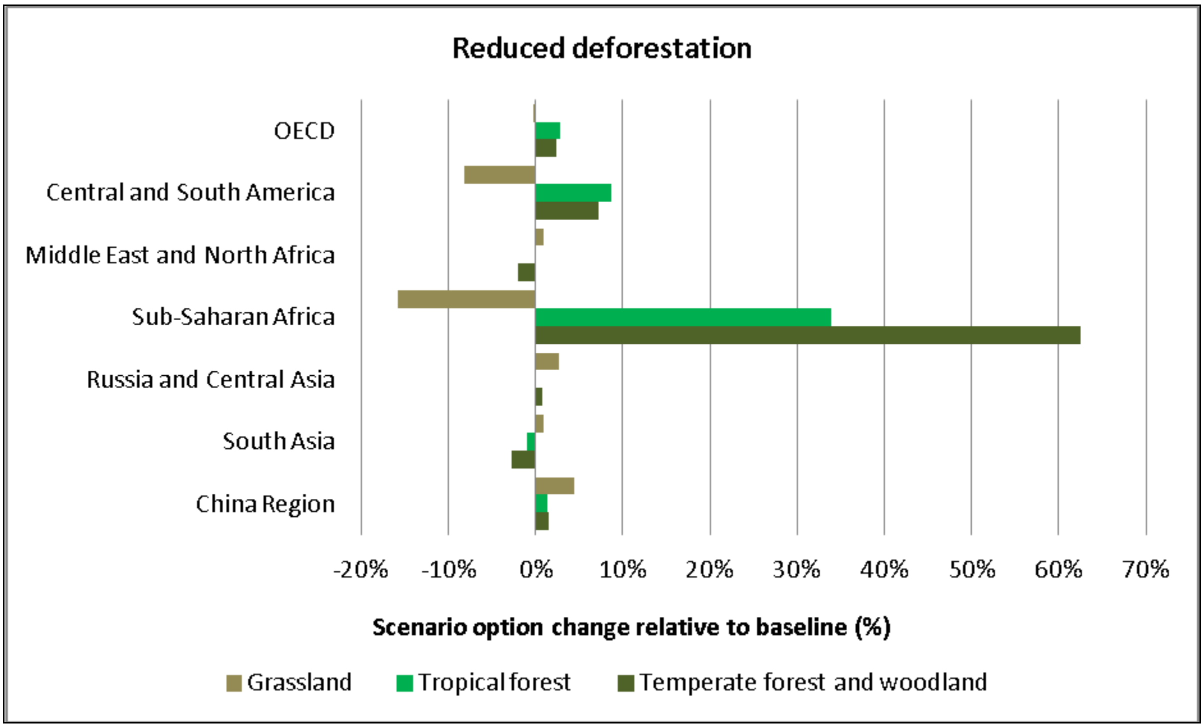

Figure 31 Reduced deforestation: change in area of biomes for scenario option relative to the baseline

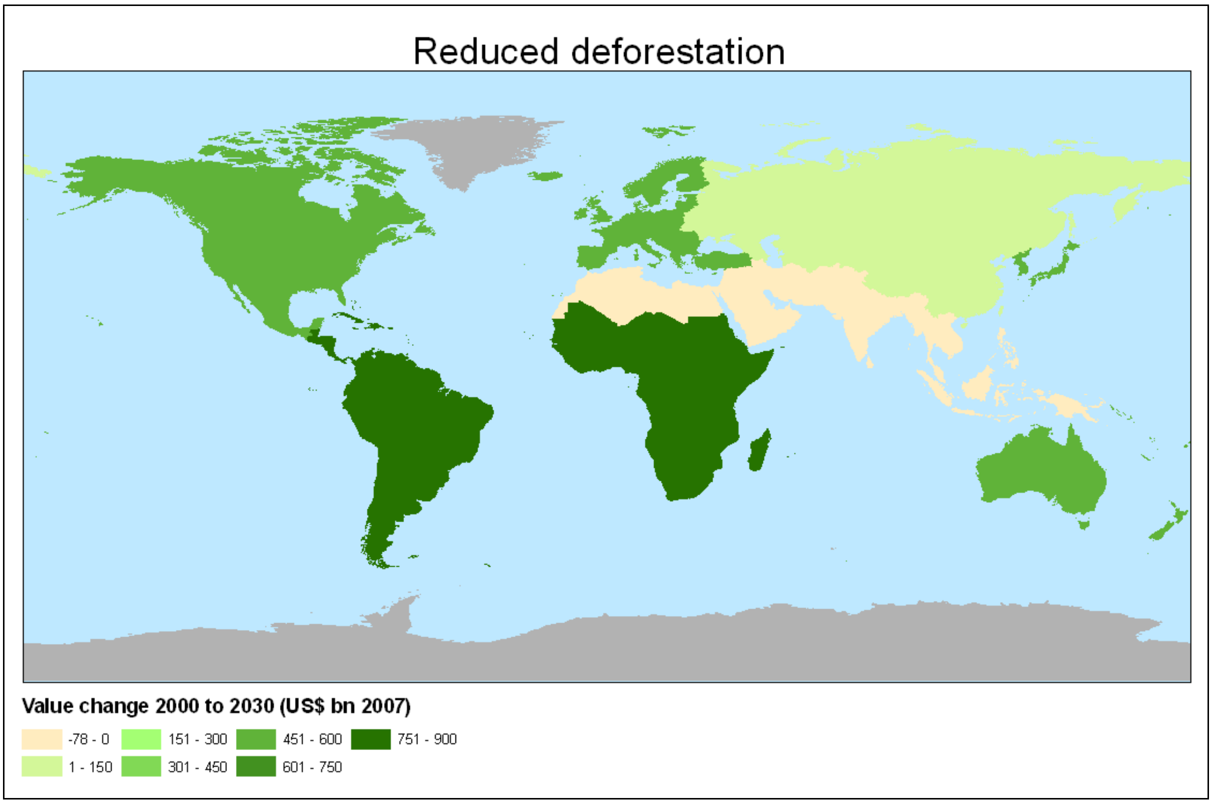

Figure 32 REDD: map of value changes 2000 to 2030 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 82 Reduced deforestation: value results by region and by biome relative to 2030 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-25.9	536.6	-1.1
Central and South America	-360.6	217.4	-5.6
Middle East and North Africa	13.8	275.2	0.3
Sub-Saharan Africa	-1481.2	53.2	-9.1
Russia and Central Asia	154.7	295.6	3.8
South Asia	14.6	120.5	0.1
China Region	155.2	188.3	1.8
Total	**-1529.4**		**-9.8**
Temperate Forest			
OECD	226.2	20081.6	36.3
Central and South America	53.0	16612.6	16.7
Middle East and North Africa	-1.4	15695.4	-0.5
Sub-Saharan Africa	87.8	9426.7	7.9
Russia and Central Asia	66.8	17210.1	6.6
South Asia	-13.8	9202.2	-2.7
China Region	28.0	15152.3	3.9
Total	**446.6**		**68.3**
Tropical Forest			
OECD	24.8	9781.3	8.6
Central and South America	567.5	7955.8	57.1
Middle East and North Africa			
Sub-Saharan Africa	1571.9	3991.8	62.1
Russia and Central Asia			
South Asia	-23.8	7131.7	-3.6
China Region	1.1	8057.2	0.2
Total	**2141.5**		**124.3**

Mean per ha values are the average of 2030 baseline and 2030 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 83 Annual and discounted aggregated regional benefits of reduced deforestation option.

	2030 undiscounted annual benefit	2000 – 2030 discounted total benefit		
		0%	1%	4%
OECD	43.8	679.1	556.1	318.9
Central and South America	68.1	1056.1	864.9	495.9
Middle East and North Africa	-0.2	-2.8	-2.3	-1.3
Sub-Saharan Africa	60.9	943.4	772.6	443.0
Russia and Central Asia	10.4	161.1	131.9	75.6
South Asia	-6.1	-95.2	-77.9	-44.7
China Region	5.9	92.1	75.5	43.3
Total	**182.8**	**2833.8**	**2320.8**	**1330.7**

The overall results show strongly positive net benefits for the reduced deforestation option. ‘Central and South America’ benefits significantly, as does ‘Sub-Saharan Africa’. Considerable benefits are also observed for the ‘OECD’ region. It is unclear what is driving the loss in forest area observed in the ‘South Asia’ region (and to a lesser extent ‘Middle East and North Africa’). It is possible that the reductions are due to losses of woodlands of lower density that are not protected by the scenario option.

12.8.1 Discussion

We set out the benefit/cost results in for reduced deforestation in Table 84. A worst-case scenario (applying the upper estimate for costs and removing additional carbon storage benefits) realises a benefit/cost ratio of 4.0 at 1% discount rate. With the SCC added and the lower cost estimate, the benefit/cost ratio is a very high value (51.4 at 1% discount rate).

Given the overall confidence in the results including the assessment of marginality, there is an unequivocally strong case for supporting the reduced deforestation option as economically efficient on a global basis.

Table 84 Overall benefit-cost ratios for reduced deforestation

		Discount rate		
		0%	1%	4%
Benefits (bn US$2007)				
Change in biome areas		2833.8	2320.8	1330.7
Carbon values (bn US$2007)				
	POLES	6466.5	6466.5	6466.5
	SCC	8655.7	8655.7	8655.7
	RICE high	4238.6	4238.6	4238.6
	RICE low	1961.7	1961.7	1961.7
Costs (bn US$2007)				
	Lower	248.3	213.7	142.2
	Upper	679.0	584.6	392.6
Benefit/cost ratios				
No carbon value	Lower	11.4	10.9	9.4
	Upper	4.2	4.0	3.4
Social Cost of Carbon	Lower	46.3	51.4	70.2
	Upper	16.9	18.8	25.4
High carbon value (POLES model)	Lower	37.5	41.1	54.8
	Upper	13.7	15.0	19.9
Low carbon value (RICE model low)	Lower	19.3	20.0	23.2
	Upper	7.1	7.3	8.4

12.9 Mitigating climate change with bio-energy

In the baseline scenario, bio-energy developments are modest, and land needed for biomass fuels is of the order of 0.5 million km^2 in 2050. In the change scenario considered in our study, GHG concentration is limited to 445 ppm CO_2-equivalent by including an expansion in bio-energy with an associated bio-energy land requirement of 4 million km^2 by 2050.

Figure 33 provides changes in the extent of grassland and forest biomes for the with-bio-fuel expansion change scenario relative to the baseline. This change scenario sees reductions in grassland area in all regions except ‘South Asia’ as a result of land conversion into either biomass production, or compensating food cultivation. The majority of changes are marginal with the exception of the loss of grassland (21.3%) and increase in temperate forest (27.4%) in ‘Sub-Saharan Africa’. A value map by region results showing breakdowns of value changes and by biome is presented in Figure 34; value results by region and by biome are presented in Table 85; and Table 86 presents overall results with three discount rates.

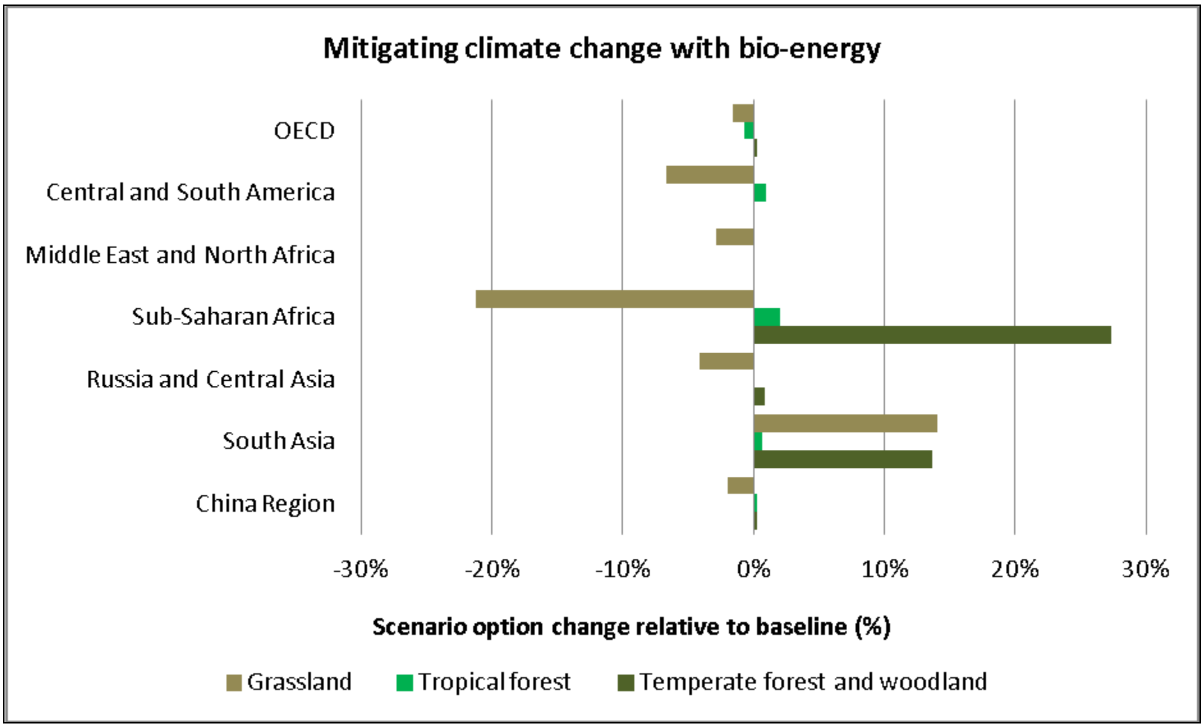

Figure 33 Mitigating climate change with bio-energy: change in area of biomes for scenario option relative to the baseline

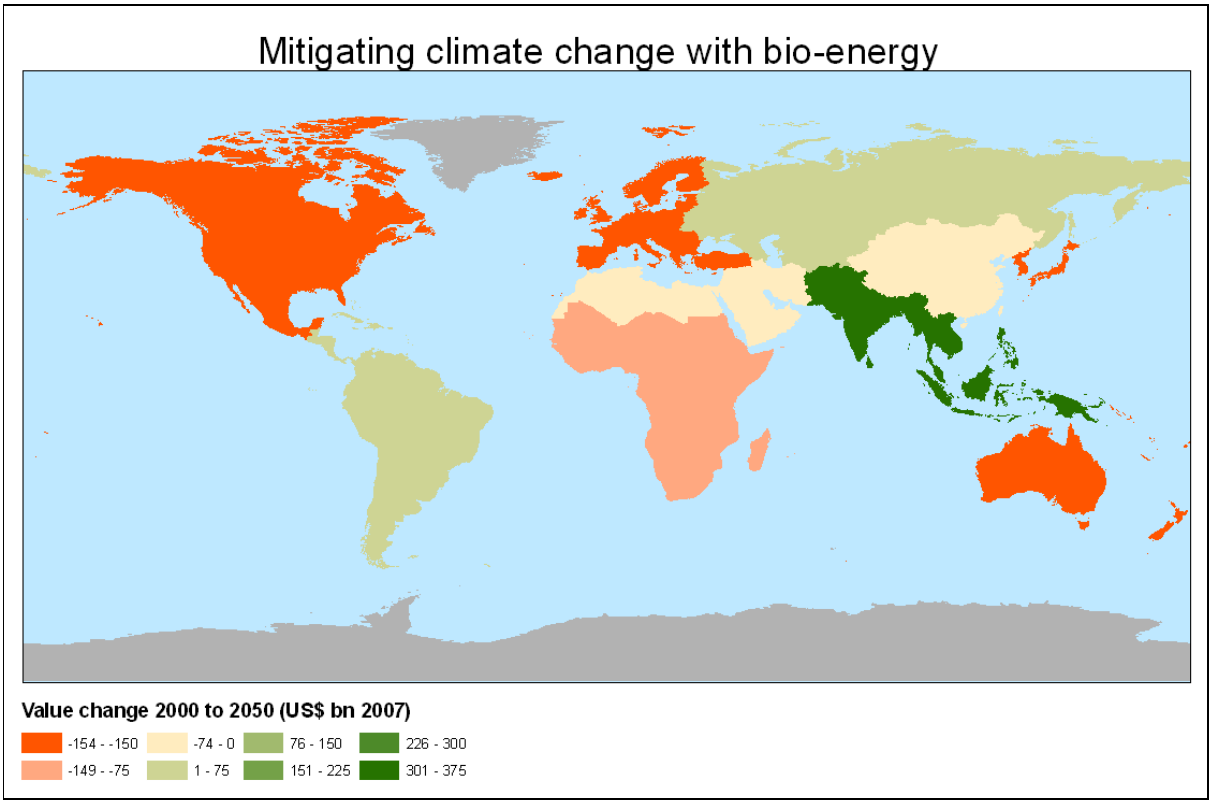

Figure 34 Mitigating climate change with bio-energy: map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 85 Mitigating climate change with bio-energy: value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-210.5	646.1	-10.2
Central and South America	-273.8	253.7	-4.9
Middle East and North Africa	-40.7	326.7	-1.1
Sub-Saharan Africa	-1840.2	63.3	-13.6
Russia and Central Asia	-230.6	351.4	-6.5
South Asia	134.5	147.9	1.3
China Region	-75.0	232.5	-1.0
Total	**-2536.2**		**-36.1**
Temperate Forest			
OECD	24.7	21012.1	4.1
Central and South America	0.9	17203.7	0.3
Middle East and North Africa	0.0	16428.8	0.0
Sub-Saharan Africa	51.4	9270.6	4.5
Russia and Central Asia	71.3	18210.3	7.5
South Asia	60.6	10032.4	12.8
China Region	5.7	16175.2	0.8
Total	**214.6**		**30.1**
Tropical Forest			
OECD	-6.2	9962.7	-2.3
Central and South America	65.1	8259.3	7.0
Middle East and North Africa			
Sub-Saharan Africa	111.9	4015.9	4.5
Russia and Central Asia			
South Asia	15.3	7611.5	2.5
China Region	0.3	8586.3	0.1
Total	**186.4**		**11.8**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 86 Annual and discounted aggregated regional benefits of mitigating climate change with bio-energy option versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	-8.4	-214.0	-154.2	-64.2
Central and South America	2.4	61.0	43.9	18.3
Middle East and North Africa	-1.1	-27.3	-19.7	-8.2
Sub-Saharan Africa	-4.5	-115.1	-82.9	-34.6
Russia and Central Asia	1.0	24.3	17.5	7.3
South Asia	16.6	424.5	305.8	127.4
China Region	-0.1	-3.5	-2.6	-1.1
Total	**5.9**	**149.9**	**108.0**	**45.0**

The value changes for the with-bio-energy extension change scenario are positive with a net benefit globally of 108.0 billion 2007 US$ at 1% discount rate. There are moderate losses regionally in 'OECD' and 'Sub-Saharan Africa' and small losses in 'Middle East and North Africa'. In each case the losses in value arise from large scale conversion of grassland to biomass production. Although the change scenario does allow for increased agricultural productivity, this alone does not allow for the 3.5 million km^2 expansion of biomass production, hence the 2.5 million km^2 reduction in grassland.

12.9.1 Discussion

As stated above (Section 12.2) no results for carbon storage values are available. However, given that this option scenario is concerned primarily with mitigating climate change, i.e. a final 450 ppm outcome is assumed, and that agricultural productivity increase are assumed to reduce land-use change pressures, these benefits are likely to be highly significant. As such, our analysis is only concerned with the *co-benefits* vis-à-vis land-use change. The overall benefit/cost results are set out in Table 87 and indicate that this change scenario does not pass the benefit/cost with respect to land-use change. As noted in section 9.7 the cost estimates reflect only direct costs, however as the change scenario is concerned with second generation bio-energy indirect costs are not considered to be as big an issue as with first generation bio-energy.

Table 87 Overall benefit/cost ratios for mitigating climate change with bio-energy option

	Discount rate		
	0%	1%	4%
Benefits (bn US$2007)			
Change in biome areas	149.9	108.0	45.0
Costs (bn US$2007)			
	1297.5	985.9	489.5
Benefit/cost ratios			
No carbon value	0.1	0.1	0.1

In summary the benefit/cost ratio of 0.1 at 1% discount rate implies that the change scenario of climate change mitigation with an expansion in of bio-energy production is not efficient with respect to land-use change, and that further benefits including carbon values would need to be considered.

12.10 Global dietary patterns

In the baseline scenario, livestock production doubles as a consequence of population and increased per capita consumption, driven notably by increased consumption in developing countries. The following two change scenarios are evaluated: (1) a global transition to vegetarianism through the complete substitution of meat protein intake by plant-based protein consumption; (2) a transition to a Willett diet which features a large proportion of fruits, vegetables, whole grains and vegetable oils, with a reduced intake of animal protein relative to a typical Western diet. Each sub-option is considered in turn below.

12.10.1 Global dietary patterns: Willett diet

Figure 35 presents changes in biome extent for the less extreme Willett diet. Changes in extent are, as would be expected, less extreme than the 'no meat' option. However, there is still a large, 23.4%, reduction observed for grassland in 'Central and South America' (versus a 28.7% reduction under 'no meat'). Increases in forest area are observed in all regions and do not exceed the 5.7% increase in temperate forest observed for the 'OECD'.

A value map by region is presented in Figure 36; value results by region and by biome are presented in Table 88; and Table 89 presents overall results with three discount rates.

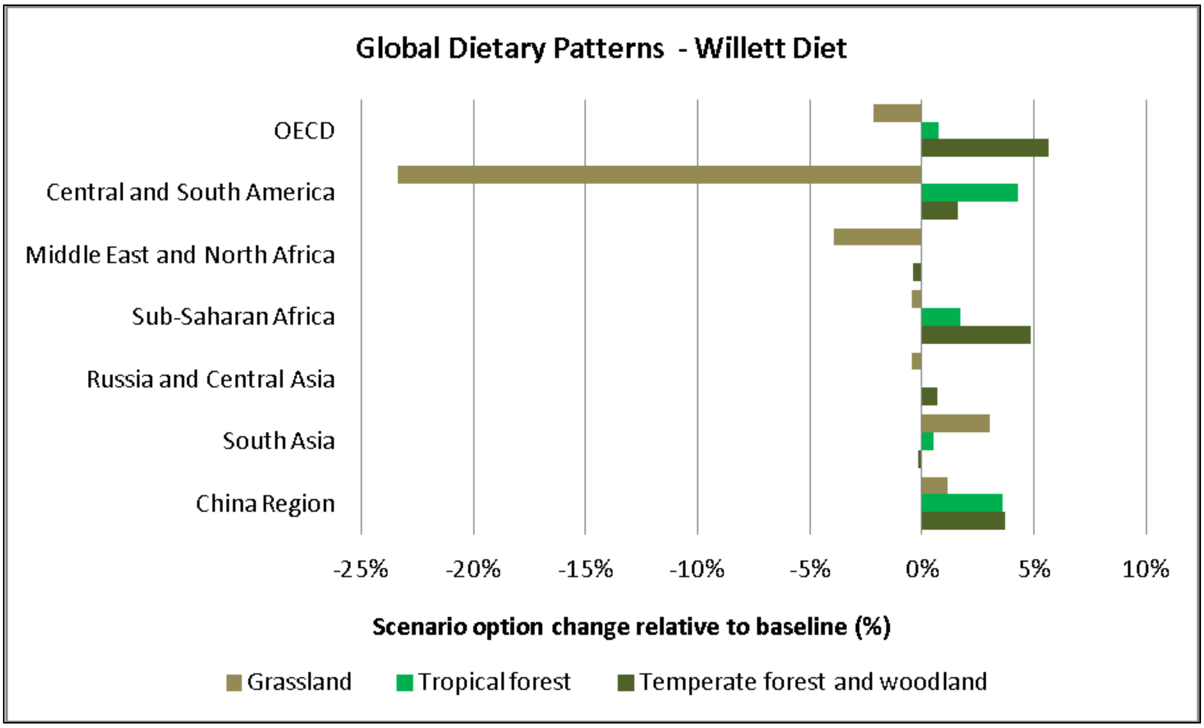

Figure 35 Global dietary patterns (Willett diet): change in area of biomes for scenario option relative to the baseline

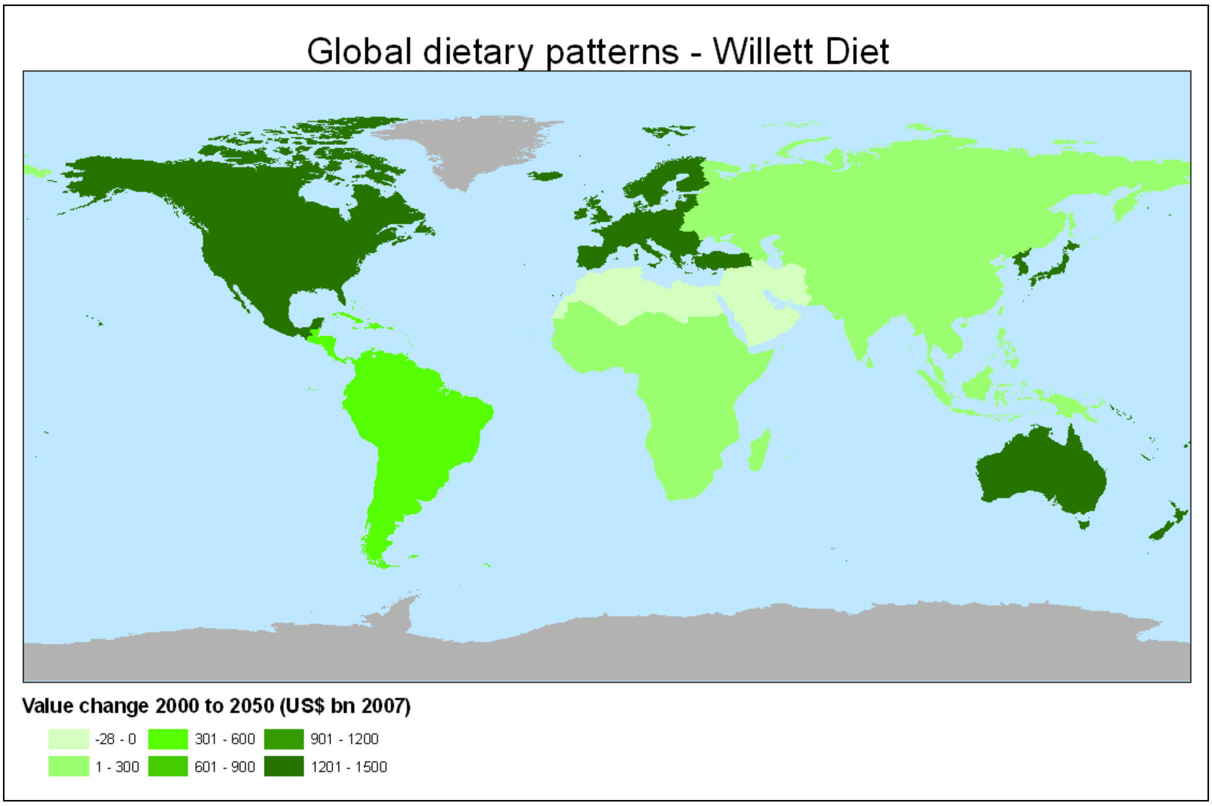

Figure 36 Global dietary patterns (Willett diet): map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 88 Global dietary patterns (Willett diet): value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-299.0	645.6	-14.5
Central and South America	-941.1	254.5	-17.0
Middle East and North Africa	-53.8	327.0	-1.4
Sub-Saharan Africa	-30.8	63.4	-0.2
Russia and Central Asia	-25.9	350.8	-0.7
South Asia	48.4	145.9	0.5
China Region	46.6	232.1	0.7
Total	**-1255.7**		**-32.8**
Temperate Forest			
OECD	550.5	20794.1	91.1
Central and South America	11.7	17673.5	3.9
Middle East and North Africa	-0.3	16449.5	-0.1
Sub-Saharan Africa	13.3	8227.3	1.0
Russia and Central Asia	57.7	18138.4	6.0
South Asia	-0.8	9720.2	-0.2
China Region	74.8	15580.8	10.8
Total	**706.9**		**112.6**
Tropical Forest			
OECD	6.8	9962.7	2.5
Central and South America	282.8	8259.3	30.5
Middle East and North Africa			
Sub-Saharan Africa	105.5	4015.9	4.2
Russia and Central Asia			
South Asia	13.2	7611.5	2.2
China Region	3.1	8586.3	0.8
Total	**411.5**		**40.3**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 89 Annual and discounted aggregated regional benefits of Willett Diet option.

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	79.1	2016.8	1453.1	605.3
Central and South America	17.5	445.9	321.3	133.8
Middle East and North Africa	-1.5	-39.3	-28.3	-11.8
Sub-Saharan Africa	5.1	128.9	92.9	38.7
Russia and Central Asia	5.3	134.8	97.1	40.4
South Asia	2.5	63.3	45.6	19.0
China Region	12.2	311.2	224.2	93.4
Total	**120.1**	**3061.7**	**2205.9**	**918.9**

There is no policy option per se pertaining to the Willett option. Our results should be taken as indicative as the changes in some regions are non-marginal. Notwithstanding this caveat, our analysis provides an interesting insight into the potential co-benefits of such a dietary change which would likely be promoted in order to promote human health outcomes.

12.10.2 Global dietary patterns: No meat

Figure 37 presents changes in biome extent for the more extreme, no meat sub-option relative to the baseline. Modest increases in forest area (up to 7.3%) are observed in six of the regions, with the exception of 'Middle East and North Africa' which sees a 111% increase in temperate forest. However, this represents a 79,000 km2 out of a 1.1 million km2 global increase in this biome. The area of grassland decreases in six regions (the exception being 'China region'), with the largest decrease of 28.9% in 'Central and South America'. The decline in grassland area arises from the conversion of pasture to arable cultivation to replace animal with vegetable protein sources.

A value map by region is presented in Figure 38; value results by region and by biome are presented in Table 90; and Table 91 presents overall results with three discount rates.

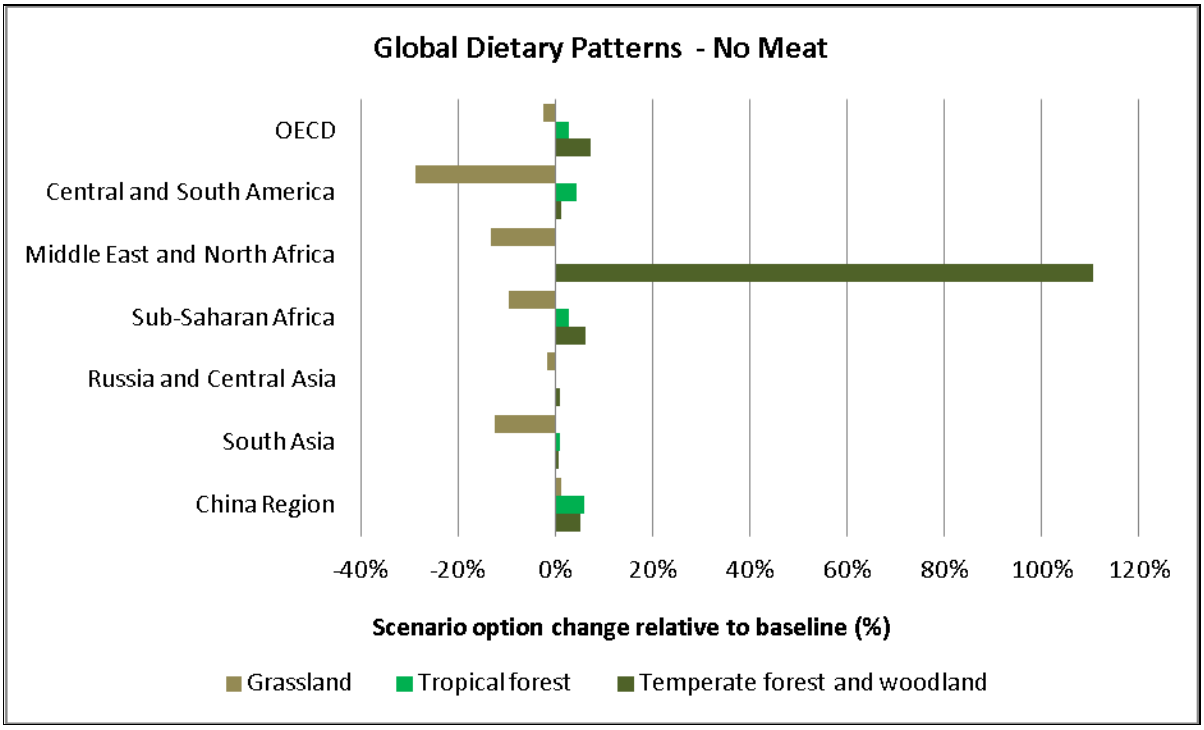

Figure 37 Global dietary patterns (no meat): change in area of biomes for scenario option relative to the baseline

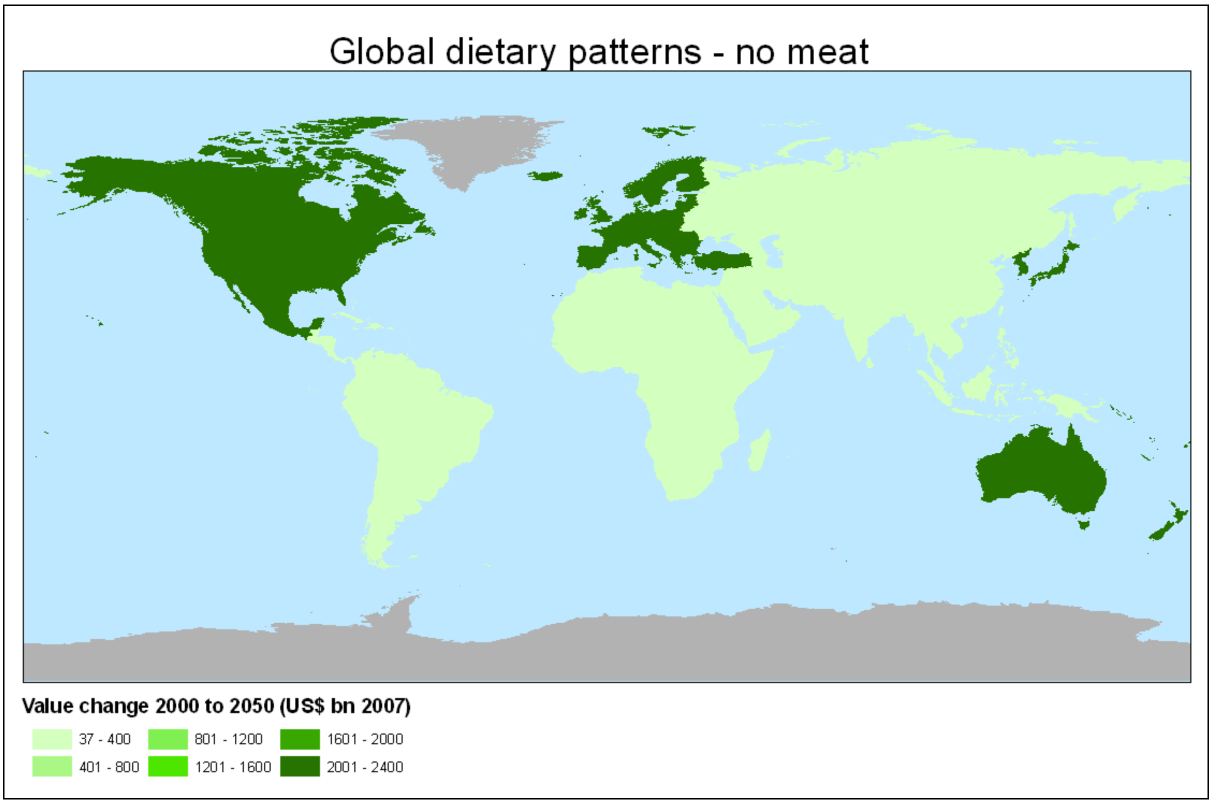

Figure 38 Global dietary patterns (no meat): map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 90 Global dietary patterns (no meat): value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	-349.3	645.6	-17.0
Central and South America	-1156.1	254.8	-20.9
Middle East and North Africa	-182.8	327.5	-4.9
Sub-Saharan Africa	-712.6	63.5	-5.3
Russia and Central Asia	-99.4	350.8	-2.8
South Asia	-198.5	146.3	-1.9
China Region	45.0	232.1	0.6
Total	**-2653.5**		**-52.1**
Temperate Forest			
OECD	710.3	20728.4	117.2
Central and South America	8.4	17690.1	2.8
Middle East and North Africa	79.3	14237.8	24.8
Sub-Saharan Africa	16.8	8207.3	1.3
Russia and Central Asia	87.3	18125.2	9.1
South Asia	2.7	9706.2	0.6
China Region	102.3	15538.9	14.7
Total	**1007.0**		**170.5**
Tropical Forest			
OECD	24.5	9962.7	9.0
Central and South America	292.7	8259.3	31.6
Middle East and North Africa			
Sub-Saharan Africa	173.6	4015.9	7.0
Russia and Central Asia			
South Asia	19.9	7611.5	3.3
China Region	5.1	8586.3	1.3
Total	**515.9**		**52.2**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 91 Annual and discounted aggregated regional benefits of no meat option.

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	109.2	2785.1	2006.6	835.8
Central and South America	13.5	345.1	248.7	103.6
Middle East and North Africa	19.9	507.9	365.9	152.4
Sub-Saharan Africa	3.0	77.0	55.5	23.1
Russia and Central Asia	6.3	160.5	115.7	48.2
South Asia	2.0	51.1	36.8	15.3
China Region	16.6	423.7	305.3	127.2
Total	**170.6**	**4350.5**	**3134.4**	**1305.6**

The benefits vis-à-vis land-use are large for this change scenario, but underpinning these changes in values are non-marginal changes in biome extent, especially for grasslands. Although the radical transformation provides an interesting context for the other change scenarios evaluated, they have little policy relevance *per se* as such a change is both inconceivable and, associated with this, it is not possible to provide a cost estimate.

The results for this sub-option are strongly positive in terms of value changes arising from land-use change. The results for the less extreme Willett diet sub-option are weaker than the 'no meat' sub-option, e.g. 2205.9 billion 2007 US$ at 1% discount rate for 'Willett diet' versus 3134.4 billion for 'no meat'. The 'OECD' region benefits greatly from the Willett diet sub-option whereas 'Middle East and North Africa' sees a small loss, which compares to a significant benefit under the 'no meat' sub-option.

12.11 Global agricultural trade

The baseline scenario assumes that the current structure of global agricultural trade persists: no dismantling of current tariff and non-tariff barriers is conceived. The option scenario assumes these barriers are progressively dismantled by 2015. Figure 39 shows the changes in biome extent relative to the baseline. Grasslands are most significantly affected in the 'Central and South America' region with a 6.8% increase. However this increase is offset by loss of forest area in that region, perhaps indicating a change in land use in favour of grazing in that region. The 4.3% reduction in temperate forest in 'South Asia' is not balanced by an increase in the grassland indicating that this is deforestation in favour of cultivation. Figure 40 maps the changes in values by IMAGE region at 1%; Table 92 provides value results by region and by biome and Table 93 provides aggregate results at 0%, 1% and 4% discount rates.

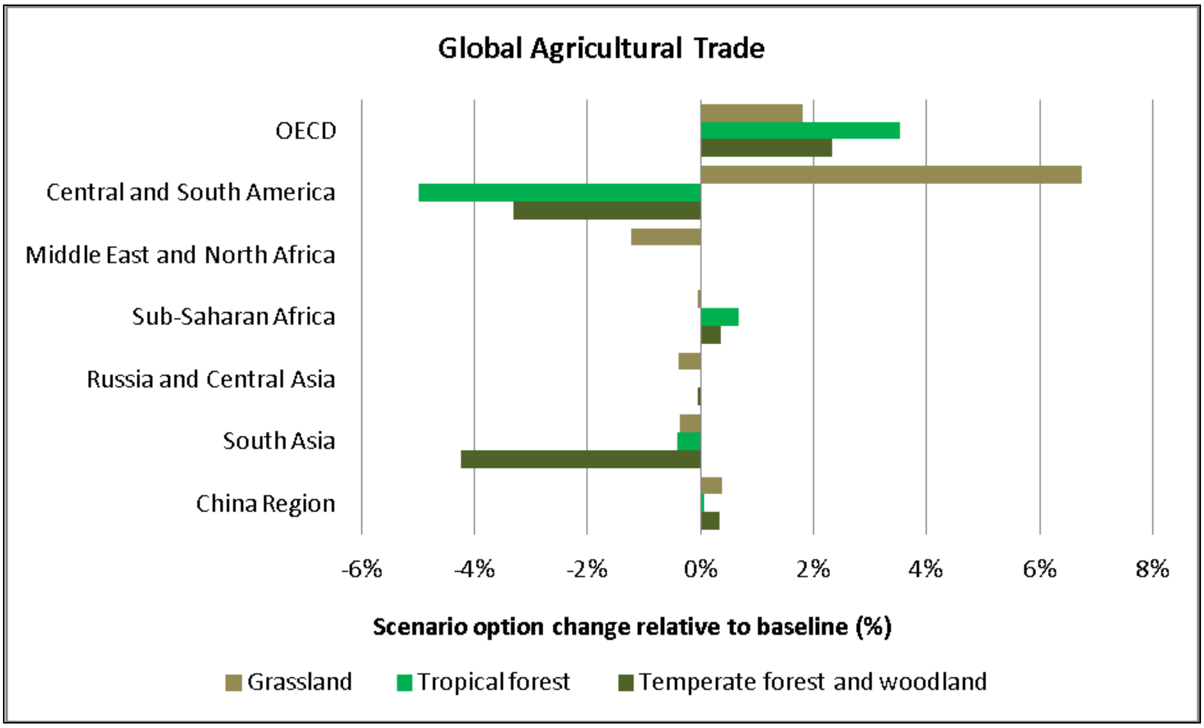

Figure 39 Global agricultural trade: change in area of biomes for scenario option relative to the baseline

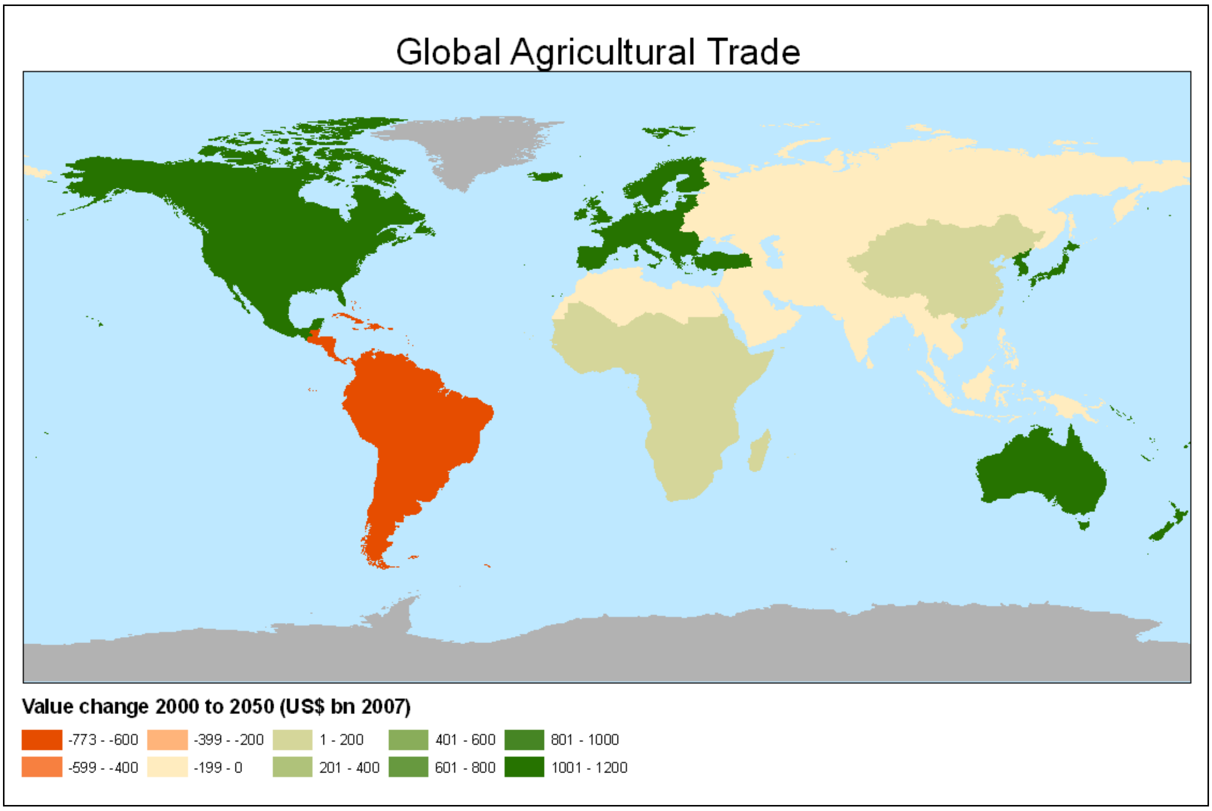

Figure 40 Global agricultural trade: map of value changes 2000 to 2050 (1% discount rate) relative to 2050 baseline assuming a linear uptake path from 2000

Table 92 Global agricultural trade: value results by region and by biome relative to 2050 baseline

	Change in area ('000 km^2)	Mean per ha value (US$ 2007)	Annual value (bn US$ 2007)
Grassland			
OECD	246.2	645.6	12.0
Central and South America	276.2	253.2	5.0
Middle East and North Africa	-17.6	326.5	-0.5
Sub-Saharan Africa	-4.3	63.1	0.0
Russia and Central Asia	-23.5	350.7	-0.7
South Asia	-3.6	148.0	0.0
China Region	15.0	232.4	0.2
Total	**488.4**		**15.9**
Temperate Forest			
OECD	229.3	20845.4	38.0
Central and South America	-26.4	17183.5	-8.6
Middle East and North Africa	0.0	16435.1	0.0
Sub-Saharan Africa	0.7	9515.7	0.1
Russia and Central Asia	-3.7	18229.8	-0.4
South Asia	-19.5	10226.9	-4.2
China Region	6.4	16202.7	1.0
Total	**186.7**		**25.9**
Tropical Forest			
OECD	31.5	9962.7	11.6
Central and South America	-356.1	8259.3	-38.5
Middle East and North Africa			
Sub-Saharan Africa	39.4	4015.9	1.6
Russia and Central Asia			
South Asia	-10.3	7611.5	-1.7
China Region	0.1	8586.3	0.0
Total	**-295.4**		**-27.0**

Mean per ha values are the average of 2050 baseline and 2050 scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 93 Annual and discounted aggregated regional benefits of global agricultural trade option.

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	61.6	1571.2	1132.0	471.5
Central and South America	-42.1	-1073.1	-773.1	-322.0
Middle East and North Africa	-0.5	-11.8	-8.5	-3.6
Sub-Saharan Africa	1.6	41.1	29.6	12.3
Russia and Central Asia	-1.1	-26.8	-19.3	-8.1
South Asia	-6.0	-152.1	-109.6	-45.7
China Region	1.2	30.0	21.6	9.0
Total	**14.8**	**378.4**	**272.6**	**113.6**

12.11.1 Discussion

Globally, the land-use value change is positive at 0%, 1% and 4% discount rate. There is a wide regional variation in costs and benefits from land-use change which might be expected. Developing world regions show the biggest net losses ('Central and South America' and 'South Asia') whereas 'OECD' countries see a significant gain (~96% of the observed benefits).

The net global value change arising from land-use change is small relative to regional variations (e.g. at 1% discount rate, the global total net benefit of 272.6 billion 2007 US$ is around 35% of the losses in 'Central and South America' alone).

No value is available for changes in carbon storage. Further, the evidence from the cost estimation is equivocal; we assume zero net cost but with the potential for regional variability.

In summary, the balance of evidence from land-use change and cost assessment (without any assessment of carbon storage changes being available) is marginal: the change scenario may or not be economically efficient, and agricultural trade liberalisation has a disproportionately negative effect on many developing world countries.

12.12 Change scenario package

PBL (2010) also present a 'change scenario package' that applies elements of various scenarios cumulatively[73], primarily with the intention of predicting the extent to which MSA decline might be mitigated were several change scenarios to be applied. PBL note that the preceding change scenarios cannot be applied independently as a series of individual measures, and that some combination of measures yields synergies. However, there is no obvious way to determine the effects on land cover of the constituent options within the package. As such we present value results in Table 94 for completeness but do not discuss the package further.

Table 94 Annual and discounted aggregated regional benefits of package option.

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit		
		0%	1%	4%
OECD	148.5	3787.1	2728.5	1136.6
Central and South America	34.0	867.2	624.8	260.3
Middle East and North Africa	-19.6	-499.0	-359.5	-149.8
Sub-Saharan Africa	44.6	1137.0	819.2	341.2
Russia and Central Asia	19.8	505.1	363.9	151.6
South Asia	3.3	85.2	61.4	25.6
China Region	-63.3	-1613.9	-1162.8	-484.4
Total	**167.4**	**4268.6**	**3075.5**	**1281.1**

[73] The 'package' includes elements of: expanding protected areas (for biodiversity and carbon stocks totalling 29% of global land area); agricultural productivity; reducing post-harvest losses; reduced meat consumption; improved forest management; and mitigation of climate change (with bio-fuel only grown on abandoned agricultural land).

13 Discussion

13.1 Change scenario benefit-cost summary

The results of the benefit transfer exercise and benefit/cost assessments presented in Section 12 are summarised in Table 95 and Table 96. The scenario option with the highest estimated land-use change benefits is the reduced deforestation option (US$182.8bn annual benefits in 2030) where the high benefits value reflects the high per ha values estimated for the forest biomes. High values were also observed for the extreme ‘no meat’ scenario option (US$170.6bn p.a. in 2050). This scenario bears useful comparison to the scenario in which there is no investment in agricultural knowledge, science and technology (AKST) where there is an annual loss in welfare of US$783.8bn in 2050. These scenarios illustrated the role of global demand for agricultural land in driving land-use change. In the ‘no meat’ scenario modest changes in forest area drive the benefits which off-set the losses due to conversion of pasture to arable for plant protein production. In contrast without investment in AKST, there are particular pressures on the conversion of forest land to agriculture. Attendant with these changes in land use there are significant gains or losses in carbon benefits in the no meat and no AKST scenarios respectively.

However we should restate the caveats that the no-AKST and ‘no meat’ option lead to non-marginal changes, therein implying that our analysis for these options is less methodologically robust. By contrast, the reduced deforestation (REDD-variant) *is* based on marginal changes and thus, although ‘extreme’ in the sense of achieving a very high benefit estimates, is defensible vis-à-vis the methodology applied in our study.

Within the bounds set by these extreme scenarios is a range of more modest options. High investment in AKST results in estimated annual benefits of US$162.1bn in 2050. When combined with carbon benefits of US$471.8bn (Social Cost of Carbon) the benefits exceed costs by at least a factor of 7. Benefit/cost ratios in excess of 7 are observed for the reduced deforestation option scenario; as would be expected the benefits are driven by large increasing in forest area with associated benefits and substantial carbon benefits. It is particularly noteworthy that the positive benefit-cost ratio from the reduced deforestation option does not depend on the carbon benefits; the ratio is >3.4 even with the upper-bound estimate for costs.

The protected areas scenarios (20% and 50% of eco-regions) produce notable results in that both see increases in grassland areas but large reductions in forest coverage in some regions[74]. This arises as a more representative selection of ecosystems is protected across the 65 eco-regions that are either protected from conversion into, or converted out of, intensive agricultural use; however the increase in protected area is offset by continued demand for food production and a shifting of production to non-protected habitats. In the 20% protected areas scenarios it is this protection of grassland that drives 43% of the benefits. In the 50% protected area scenario the losses of forest area and benefits become

[74] Note that IMAGE/GLOBIO suggests overall reductions in temperate and tropical forest area, our analysis using the same change factors indicate aggregate gains in the PA 20% option. This disparity arises from differences in the areas of forest patches and across regions between our respective datasets.

considerable and off-set the gains in grassland area, leaving a considerable US$283.1bn total annual loss. These changes in benefits across the two protected areas scenarios mean that when combined with carbon values the benefit costs ratio fall from between 1.9 and 2.3 for 20% to 0.7 for 50% protected areas under the highest 4% discount rate. Again, we note that the 50% protected area option scenario, as well as being politically unrealistic, is also out with the bounds of ‘marginality’ for land-use change and thus the results for PA 50% are at best only indicative.

We would re-emphasise that our results reflect the ecosystem services lost or gained through land-use change and not the total value of services in ecosystem patches that do not change. This latter reason perhaps also explains the welfare losses estimated for the forest management scenario.

Two final scenarios for which no carbon benefits or cost estimates were available were post harvest losses and global agricultural trade. The first of these involves halving the approximately 30% of food production that is lost globally through a range of causes including waste, spoilage and lack of access to markets. In essence this scenario leads to an effective yield increase and consequently reduces the area of land needed for a given output. The agricultural trade scenario highlights some of the distributional issues of the different scenario with welfare gains in OECD, Sub-Saharan Africa and China. Losses are seen in Central and South America, Middle East and North Africa, Russia and Central Asia, and South Asia.

Table 95 Summary global land-use change and benefits of change scenarios by biome

Change scenario	Time frame	Grassland		Temperate forest		Tropical forest		Total annual value (bn US$ 2007)
		Area change ('000 km^2)	Annual value (bn US$ 2007)	Area change ('000 km^2)	Annual value (bn US$ 2007)	Area change ('000 km^2)	Annual value (bn US$ 2007)	
Agricultural productivity (high AKST)	2050	1167.8	27.8	456.8	81.7	471.7	52.6	162.1
Post harvest losses	2050	117.2	3.7	84.0	15.3	119.4	10.6	29.5
Forest Management	2050	-8.5	-0.1	-5.0	-1.3	-41.5	-2.2	-3.6
Protected areas (20%)	2030	609.7	14.7	89.2	8.7	111.9	11.0	34.4
Reduced deforestation	2030	-1529.4	-9.8	446.6	68.3	2141.5	124.3	182.8
Mitigating climate change with bio-energy	2050	-2536.2	-36.1	214.6	30.1	186.4	11.8	5.9
Dietary change (Willett diet)	2050	-1255.7	-32.8	706.9	112.6	411.5	40.3	120.1
Global agricultural trade	2050	488.4	15.9	186.7	25.9	-295.4	-27.0	14.8

Table 96 Summary change scenario aggregate benefits, costs and benefit/cost ratios

Change scenario	Discount rate	Discounted total benefit (bn US$ 2007)	Carbon benefit (SCC, bn US$ 2007)	Discounted total cost (bn US$ 2007)	Benefit/ cost ratio
Agricultural productivity (high AKST)	0%	4133.2	6342.8	725.0	14.4
	1%	2977.9		568.3	16.4
	4%	1240.4		311.5	24.3
Post-harvest losses	0%	816.8	Not estimated		
	1%	588.5			
	4%	245.1			
Forest Management	0%	-120.6	2912.8	5195.0	0.5
	1%	-86.9		4072.5	0.7
	4%	-36.2		2232.0	1.3
Protected areas (20%) [a]	0%	532.7	367.1	465.1	1.9
	1%	436.3		400.4	2.0
	4%	250.2		269.0	2.3
Reduced deforestation [a, b]	0%	2833.8	8655.7	679.0	16.9
	1%	2320.8		584.6	18.8
	4%	1330.7		392.6	25.4
Mitigating climate change with bio-energy	0%	149.9	Not estimated	1249.0	0.1
	1%	108.0		938.8	0.1
	4%	45.0		452.7	0.1
Dietary change (Willett diet)	0%	4350.5	17293.0	Not estimated	
	1%	3134.4			
	4%	1305.6			
Global agricultural trade	0%	378.4	Not estimated		
	1%	272.6			
	4%	113.6			

[a] Aggregation between 2000 and 2030
[b] Upper bound cost estimates reported

13.2 Uncertainty in the estimated benefits

This section discusses the sources and degree of uncertainty in the benefits presented in the global quantitative assessment. The estimated values for changes in the extent of biomes are recognised to be subject to high uncertainty. There are multiple sources of uncertainty in the analysis, which are layered one on top of the other. Each step in the analysis introduces an additional degree of uncertainty. The quantification of the uncertainty introduced in each step is difficult (e.g. in terms of confidence intervals) and the combined uncertainty of the final results even harder to assess. We provide here a qualitative description of each source of uncertainty.

13.2.1 Uncertainty in the bio-physical modelling of changes in biome extent

The spatial resolution at which the modelled changes in land use are valid is not high. Initial land cover in the IMAGE model is derived from 1 by 1km resolution land cover data (GLC2000); this is then aggregated to a 0.5 degree resolution (approx 50km) grid. Land is then allocated to agricultural use or natural vegetation on a ranking scheme until allocations match with FAO land use data. Consequently, although general land use patterns are reproduced in IMAGE-GLOBIO their detailed spatial distribution is lost. This means that change factors for different biomes can only be applied at the aggregate level (such as IMAGE regions) and not the 0.5 degree grid level. PBL (2010) provide further discussing of factors affecting the robustness of the IMAGE-GLOBIO model results including baseline and change scenarios.

Notwithstanding the fact that the need to apply the same change factor (e.g. 10% increase in temperate forest) across an entire IMAGE region decreases the spatial resolution, the analysis carried out in Part III is still at patch level. Thus the simplification is that (say) all the temperate forest patches in the specific IMAGE region increase in size by 10%, but the value of this change is patch-specific and estimated using the temperate forest value function applied individually to each and every patch in the region.

13.2.2 Uncertainty in GIS data and spatial modelling

The GIS data underlying the global biome maps and spatial variables included in the value functions is also subject to a degree of uncertainty. The selection of spatial data followed a set of criteria to identify reliable data sets. These criteria include the consistency, accuracy and credibility of the data. Nevertheless, the requirement for data with global coverage necessitates the use of data that is subject to a degree of imprecision. Sources for this imprecision include measurement and analyst errors but also deliberate generalisations, e.g. reclassification of land use classes into a limited number of subclasses. The consistency issues in global datasets can especially be found in socioeconomic data originating from different data collection periods within a dataset and/or between datasets and/or different data collection methods between countries or regions. The global road density dataset used is based on an original data collection from 1993, with improvements made in 1997. Although more current global road datasets are in development, this was the best documented and internally consistent dataset publically available. Another potential source of error is the assignment of geographic coordinates to the selected study sites that are subsequently used to calculate the value functions. Depending on the provided geographic information the centroids of some study sites can be located accurately, others can only be located on much coarser scale levels, e.g. the centre of a region in which the studied biome is located. Furthermore, although checked rigorously, analyst errors cannot be excluded in the manual recording, transcription and conversion between different geographic formats of x-y coordinates.

Although the combination and processing of above described error sources and uncertainty can lead to a significant propagation of error in the GIS data used, it is considered of relatively small importance compared to the uncertainties caused by the much lower spatial accuracies used in the modelling of biome extent changes.

In addition to imprecision in the spatial data itself, the processing of data to the format required for our analysis adds a further degree of uncertainty. A significant processing error is caused by the fact that different global datasets with different spatial accuracies have different geographic extents. For example, the global human appropriation of net primary production (HANPP) dataset has an accuracy of 5 arc minutes (grid cells of ca. 10 x 10 kilometres at the equator), resulting in different boundaries at the continent edges. Other processing errors concern the reprojection and resampling of data to the resolution and exact grid cell locations of the reference grid used.

Probably the largest source of potential error in the spatial data processing is caused by the application and global upscaling of the developed spatial variables on patch level. The chosen analysis resolution of 1 by 1 km grid cells leads to the formation of 'super patches',

patches of connecting grid cells of the same land use up to millions of square kilometres in which biomes are connected which are in reality dissected, e.g. by road infrastructure.

Another but much smaller issue in the use of the spatial variables is the allocation of GDP per capita values to the biomes coral reefs and mangroves, which are located in the ocean outside the continental areas for which the GIS databases contain data. Depending on the exact location of a coral reef it can be that a GDP per capita value is assigned based on the most proximate country, while the coral reef belongs to a country at a larger distance with a different GDP per capita value. However it should be noted that these coastal biomes were not part of the core assessment of eight change scenarios.

There are many more error sources and uncertainties relating to known GIS science issues (for an overview see e.g. Mark, 2003) in both the use of geodata and geoprocessing that will influence the final results.

13.2.3 Uncertainty in the primary valuation data underlying the value functions

Measurement error in primary valuation estimates may result from weak methodologies, unreliable data, analyst errors, and the whole gamut of biases and inaccuracies associated with valuation methods (EEA, 2010). Inaccuracies in the value data underlying the estimated value functions is a potential source of unexplained variance in the meta-regressions. Variation in observed values is to a large extent due to methodological variation in primary valuation studies rather than variation in the characteristics and context of the ecosystems valued. Meta-analyses of the ecosystem service value literature have tended to find that variables indicating differences in methodology are highly important in explaining differences in value results (e.g. Woodward and Wui, 2001; Brander *et al*., 2006; Johnston *et al*. 2006). In addition to biases in the primary studies themselves, there is also an observed bias in the publication of study results. Publication selection bias arises when the publication process through which valuation results are disseminated results in an available stock of knowledge that is skewed to certain types of results and that does not meet the information needs of value transfer practitioners. In the economics literature there is generally an editorial preference to publish statistically significant results and novel valuation applications rather than replications, which may result in publication bias.

13.2.4 Uncertainty in the transfer of values across ecosystems with widely varying characteristics

So-called generalisation error occurs when values for study sites are transferred to policy sites that are different without fully accounting for those differences. Such differences may be in terms of population characteristics (income, culture, demographics, education *etc*.) or environmental/physical characteristics (quantity and/or quality of the good or service, availability of substitutes, accessibility *etc*.). In the context of transfer using meta-analysis based value functions, generalisation error can arise due to the common limitation of meta-analyses to fully explain variation in primary value estimates. This is clearly the case for the biome value functions used in the global assessment, which at best explain 40% of variation in values (mangroves, tropical forests) and at worst explain only 17% (lakes and rivers, coral reefs), although the latter two biomes do not constitute part of the core analysis of eight option scenarios.

Generalisation error will also be driven by the extent of differences between the characteristics of the study sites used in the primary valuation literature and the characteristics of the sites to which values are transferred. This again varies across the biomes included in the quantitative assessment, with some (e.g. wetlands) showing reasonably broad geographic representation in the data and others relying on unrepresentative samples (e.g. lakes and rivers).

There is also a temporal source of generalisation error in that preferences and values for ecosystem services may not remain constant over time. Using value transfer to estimate values for ecosystem services under future policy scenarios may therefore entail a degree of uncertainty regarding whether future generations hold the same preferences as current or past generations.

A limited measure of generalisation error can be obtained by examining the differences between observed primary values and predicted values (using the estimated value functions). In other words we can use the meta-analytic value functions to predict the values found in the primary valuation literature and compute transfer errors (percentage difference between primary and predicted values). We perform this type of analysis for the temperate forest, tropical forest, and grassland data, i.e. those biomes used in the core analysis of the eight option scenarios.

The analysis of the temperate forest data reveals a mean transfer error of 277% and a median error of 23%. This median error compares favourably with values in the literature, for example Lindhjem and Navrud (2008) report mean transfer errors of between 33% and 126% and median errors between 29% and 70% using meta-analytic transfer for multi-use Scandinavian temperate forests. However, as the authors note the studies they included in the meta-analysis were homogeneous in terms of the goods valued, the valuation methods employed and socio-economic conditions. The analysis of tropical forest data reveals a mean transfer error of 381% and a median error of 29%.

For the grassland data we find median transfer errors of 354%. This high transfer error is largely driven by over prediction of low primary values. In many cases, small over predictions in absolute terms result in large percentage errors if the primary value is low. Although the estimated value function for grasslands is limited in its explanatory power, it does result in lower transfer errors than if mean grassland value were transferred instead. In this case the median transfer error would be approximately 500%.

This approach to measuring generalisation error is described as limited because it is restricted to examining the precision of predicting the available primary value data. There are two reasons why this does not provide an accurate view of actual transfer errors. First, primary value estimates are treated as ‘true’ observations of welfare whereas they are in fact also imprecise estimations. Second, in the global assessment we are transferring values outside of the set of study sites represented in the primary data. If the study sites used in the primary valuation literature are not representative of the policy sites to which values are transferred, the ‘in-sample’ transfer error will not be a good reflection of transfer errors to policy sites.

Uncertainty regarding the precision of value estimates may be larger at the level of specific ecosystem sites than at the regional or global level. The various sources of uncertainty inherent in the global assessment are expected to lead to substantial transfer errors at a site specific level. The value of some ecosystem sites may be under estimated and others over estimated. It is expected, however, that to some extent these transfer errors will cancel out when site level values are aggregated to the regional and global level. For this reason we place more confidence on regional values than site specific value estimates.

13.3 Future research requirements

This research has highlighted considerable uncertainties in the estimates of the global benefits and costs of the change scenarios. The sources of these uncertainties fall into four categories: biophysical modelling; spatial data and analysis; primary valuation studies; and value transfer. In considering these in turn we now identify future research requirements.

Biophysical modelling

This is necessarily an abstraction and will always be subject to the inherent uncertainties resulting from the use of stylised physical and economic relationship, i.e. the specification of baseline and policy scenarios and the algorithms within the model. However in this study we have been restricted to considering only estimated land-use changes in area terms, i.e. quantity, and not the *quality* of those changes to ecosystems as expressed in through Mean Species Abundance (MSA) in PBL (2010). Our analysis is also partial in that we were unable to consider benefits of freshwater, coastal and marine biomes. The MSA measure is not readily amenable to economic valuation as it is not directly comparable across biomes and values all species equally regardless of their scarcity or contribution to ecosystem services, i.e. valuation endpoints. We therefore recommend that:

- Research is directed towards developing biodiversity indicators that reflect both the intactness of an ecosystem (as per MSA) and also the delivery of ecosystem services that can be more readily valued in economic terms.
- Future biophysical modelling should aim to capture changes to terrestrial, aquatic and marine biomes.
- In the interim, analysis should be undertaken to determine the extent to which our estimated values reflect the changes in MSA estimated by IMAGE-GLOBIO, i.e. is there a positive correlation between land-use change benefits based on ecosystem service provision and MSA?
- The biophysical models should be periodically re-run to reflect changing realities, e.g. new agreements on policies such as the extent of protected areas and the mechanisms used in their selection and designation. Such reanalysis could be undertaken following consultation with bodies such as WCPA.

In estimating the total benefits of each change scenario, we have only considered the changes in land use by each scenario's end point (either 2030 or 2050) with a linear change trajectory from the 2000 baseline.

- Further intermediate points such as 2010 and 2030 (for 2050 scenarios) should be estimated to determine more realistic trajectories. These data are available in some cases but were made available to the QA team close to the report submission deadline. The precision and reliability of results would be improved markedly by running the analysis for these intermediate points.

Spatial data and analysis

This reflects both the biophysical modelling approach of IMAGE-GLOBIO and the nature of the spatial data we have incorporated in our value functions. On the first point we recognise the inherent difficulty in scaling up from relatively fine resolution land cover maps (GLC 2000) to the regional land use model within IMAGE-GLOBIO. The regional level land-use changes estimated by IMAGE-GLOBIO do not then easily scale back down to individual patches, and it is not apparent how land-use change is distributed either within regions or across different types of patch (e.g. patches of different size). Regarding the second point, our choice of spatial variables for use in the value function estimation was restricted to publically available global datasets and arbitrary radii (10, 20 and 50km) from patch centres. The latter were decided by the project team, and alternative specifications were not considered due to resource, particularly time, constraints. We suggest that:

- Expert opinion or sub-regional models could be used to identify likely distributions of land-use change across patches, e.g. thresholds of gains or losses based on patch size or location. This approach has potential for use in sensitivity analysis given the dependency of per hectare values on patch size and other spatial variables.
- Primary valuation studies should consider site-specific spatial variables in the estimation of value functions and be precise on the spatial definition of the sites and services they are valuing. This would assist and guide future benefit transfers.
- Future benefit transfers should revisit the available spatial data for inclusion in value functions. More detailed country or region level spatial data should be use where available and appropriate.

Primary valuation studies

We have identified uncertainties arising from the primary valuation data; these will always be present when using meta-analysis to derive benefit functions due to variations between studies, in terms of both sites and methodology. Ultimately questions as to the precision of any individual value estimate may remain unresolved. However, it is also the case that many of the primary valuation studies used in this research pre-date the widespread adoption of the ecosystem approach and the resultant values can only be crudely mapped onto particular ecosystem services. We recommend that:

- Primary valuations are explicit in the use of the ecosystem approach and consistent in their definitions of ecosystem services.
- Publication bias can be overcome through a greater willingness by journals to publish papers using non-novel methodologies that add consistent and robust value estimates to the literature.

Value transfer

Many of the uncertainties that we have described with respect to the value transfer are well recognised and applicable to all such exercise, e.g. the assumption that preferences and values are constant over time. Other uncertainties are more specific to this analysis. For instance, values have been used where available across a range of ecosystem services; the quantity and quality of those services has not been considered in either all of the primary valuation or the transfer sites, therefore per hectare values may not fully reflect the true degree of ecosystem service delivery at individual sites. It might be the case that there is selection bias within the primary valuation literature in favour of sites providing higher levels of particular ecosystem services. The degree to which this biases our results upwards is, however, debatable as not all possible ecosystem services are typically valued at any one site. The value functions used in this study rely on a small number of primary valuation studies (relative to the number of patches) with often a limited geographical spread. This is particularly the case with grasslands in terms of study numbers, and rivers, lakes and wetlands in terms of dispersion (although the latter are not included in the benefit/cost assessment). We recommend that:

- As additions to the primary valuation literature allow, value transfers should consider ecosystem services either individually or in groups.
- Value functions should better reflect the degree of ecosystem service delivery at transfer sites.
- Efforts should be made to expand the coverage of the primary valuation literature in terms of the number of studies available (e.g. for grasslands) or the geographical coverage of studies.

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