challenge paper

BIODIVERSITY

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The Challenge of Ecosystems and Biodiversity

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Table of Contents

List of tables

List of figures

Table of acronyns

BAU	Business As Usual
AKST	Agricultural Knowledge, Science and Technology
BRIC	Brazil, Russia, India and China
CBD	Convention on Biological Diversity
CGE	Computable General Equilibrium
COP	Conference of Parties
ESS	Ecosystem Services
EJ	Exa Joules(20^{18} Joules)
FAO	Food and Agricultural Organization
FARM	Future Agricultural Resources Model
GIS	Geographical Information System
GLC	Global Land Classification
GLOBIO	Global Methodology for Mapping Human Impacts on the Biosphere: a dose response biodiversity model
GTAP	Global Trade Analysis Project
GTEM	Global Trade and Environment Model: a computable general equilibrium model used to assess agricultural yields
IAASTD	International Assessment of Agricultural Knowledge, Science and Technology for Development
IDENT	Identification Number
IEEP	Institute for European Environmental Policy
IMAGE	Integrated Model to Assess the Global Environment: an integrated assessment model
IMPACT	International Model for Policy Analysis of Agriculutural Commodities and Trade: a partial equilibrium model used to assess agricultural yields
IPCC	Intergovernmental Panel for Climate Change
IUCN	International Union for the Conservation of Nature
MAC	Marginal Abatement Cost
MD	Marginal Damages
MEA	Millennium Ecosystem Assessment
MSA	Mean Species Abundance
NTFP	Non-timber Forest Products
OECD	Organization for Economic Cooperation and Development
OLS	Ordinary Least Squares
PA	Protected Areas
PBL	Netherlands Environmental Assessment Agency (Planbureau voor de Leefomgeving)
POLES	Prospective Outlook on Long-term Energy Systems
R & D	Research and Development
REDD	Reducing Emissions from Deforestation and Forest Degradation
RICE	Regional Integrated Model of Climate and the Economy
sCBD	Secretariat of the Convention on Biological Diversity
SPSS	Statistical Package for the Social Sciences
TEEB	The Economics of Ecosystems and Biodiversity
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
UNEP	United National Environment Programme
UNEP-WCMC	United Nations Environment Programme: World Conservation Monitoring Centre
USD	United States Dollars
WCERE	World Congress of Environmental and Resource Economists
WDPAs	World Database on Protected Areas
WSSD	World Summit on Sustainable Development

1 Introduction

In this challenge paper we look at the costs and benefits of three possible interventions that would enhance the planet's biodiversity and improve its ecosystems over the next 40 years. The results are based on a study carried out across four research institutes and coordinated by the Scottish Agricultural College (Hussain et al., 2011)[1] that combined a global biophysical model (IMAGE-GLOBIO) which analyses the biophysical impacts of different development scenarios compared to the counter-factual, with a set of valuation studies that placed monetary values on the outcomes resulting from the different policy options in terms of biodiversity and ecosystem services.

While reference is frequently made in the popular press to biodiversity losses, in practice it is difficult to quantify and value them. There are several studies that attempt to do this in specific cases but no one has successfully estimated the value of the loss of biodiversity at a global level[2]. This is because the links between biodiversity and biological systems and the economic and social values that they support are extremely complex. Even the measurement of biodiversity is problematic, with a multi-dimensional metric being regarded as appropriate (Purvis and Hector, 2000; Mace et al., 2003), but with further work being considered necessary to define the appropriate combination.

For this reason the focus, initiated by the Millennium Ecosystem Assessment (2005), has been on measuring ecosystem services, which are derived from these complex biophysical systems. The Millennium Ecosystem Assessment (MEA) defines ecosystem services under four headings: provisioning, regulating, cultural and supporting and under each there are a number of sub-categories.

The most important fact about these services is that they have also been facing major losses. During the last century the planet has lost 50 per cent of its wetlands, 40 per cent of its forests and 35 per cent of its mangroves. Around 60 per cent of global ecosystem services have been degraded in just 50 years (MEA, 2005).

While working at the ecosystem level makes things somewhat easier it is still important to understand the causes of the loss of these services and the links between losses of biodiversity and the loss of ecosystem services. Indeed this is a major field of research for ecologists and one thesis that has been articulated over a long period is that more diverse ecosystems are more stable and less subject to malfunction (Haines-Young and Potschin, 2010; McCann, 2000, Tilman and Downing, 1994). The current state of knowledge on the links between biodiversity and ecosystem services is still a topic of research and while some clear lines are emerging, they are not strong enough to allow

[1] Anil Markandya was an advisor to the Project and a reviewer of the report and Salman Hussain was the coordinating lead author. We are interpreting some of the results of the original study in a way that involves additional assumptions and analysis. For that reason the other authors of the original paper cannot be held responsible for the contents of this paper.

[2] For a review see ten Brink (ed.) 2011, Chapter 5.4

a formal modelling to be carried out at a level that would produce credible estimates of the global value of biodiversity. The latter therefore remains a topic for research[3].

If the aim is to obtain estimates of changes in the economic values of services from natural systems at the global level, as it is in this paper, one has, of necessity, to estimate these services through the ecosystem valuation framework, recognizing that there is a complex link between changes in such values and the changes in the measures of biodiversity (appropriately defined). However, the ecosystem methodology used in the paper does take into account the quality of an ecosystem and the services it produces, based on the species abundance within it. This is derived from the Mean Species Abundance (MSA) approach, which is explained more fully in the next section. To some extent therefore, the study does build on the linkages between the biodiversity of a biome and its ecosystem functions.

What we attempt to do in this paper is to link bio-physical modelling (PBL, 2010) with non-market valuation (Hussain et al. 2010). The bio-physical modelling set out projections for global business-as-usual (BAU) scenarios for terrestrial ecosystems as well as scenarios with three policy options (increases in agricultural productivity; extending protected areas; reduced reforestation). There is thus a bio-physical comparison between BAU and each policy option in turn, with outputs presented in terms of changes in the extent of terrestrial land cover and also a measure of biodiversity (MSA).

The core economic benefit appraisal is based on changes in land cover. This analysis is carried out at patch level using a geographical information system (GIS). We analyse changes in the extent of circa 2.3 million patches of grassland, temperate forest and tropical forest. If the size of a particular patch changes (BAU *versus* policy option) then the value of that patch also changes, owing to a modified provisioning of the suite of ecosystem services provided by that patch. The novelty in the economic benefit analysis is that unlike previous analyses the value change estimates are patch-specific and then aggregated across all circa 2.3 million patches, with the values of some patches increasing under policy-on and the value of others decreasing.

We apply benefits transfer using meta-regression analysis. We have constructed a comprehensive spatially-referenced database of primary valuation estimates for the three biomes. We use GIS databases on various variables that might influence the per hectare value estimate at patch level. Typically in benefits transfer the only variable which is routinely used in meta-regression-based transfer is the income variable. We extend this by considering a range of variables that are proxies for the ecological characteristics of the patch (e.g. human appropriation of net primary product, a proxy for habitat intactness) as well context characteristics (e.g. proximity to roads). We use extant databases (where available) to infer the projected values of these variables at the end of the study period (which is 2030 or 2050 depending on the policy). The value of a projected increase in extent for a particular patch in 2030 or 2050 depends on these variables as described by the biome-level meta-regression analyses.

[3] Theoretical models of the economic values attached to biodiversity have been developed. See for example, Brock and Xepapadeas (2003). Such models draw simple links between harvesting rates, system biodiversity and overall system value. As yet, however, they are not supported by empirical estimates that be used to apply the methods to derive these system values.

The analysis presented here aims at meeting the setup of the Copenhagen Consensus where a hypothetical, additional budget of 75 billion USD over the next four is assumed. The four years and the budget is not viewed as an one-time increase, but rather as a perpetual increase.

We argue in the conclusions, that spending a smaller amount than proposed will not lower the benefit cots ratios we derive because none of our programs have significant set up costs. In fact the benefit cost ratios could be higher for a policy option limited to 75 billion USD or less as that would allow one to pick out the options with higher net benefit. We could not undertake such an exercise for this study because it would have involved a lot more disaggregated work.

We also note that our costs and benefits start in 2000 and go forward to 2030 or 2050, depending on the policy option being considered. The starting date is defined by the bio-physical modelling: the model runs use the Global Land Cover 2000 (GLC2000)[4] as a baseline, as described in section 2. All benefits and costs are standardised to 2007 USD. This year for standardisation was chosen as, at the time the analysis was being carried out (2010), 2007 was the last year for which World Bank purchasing power parity (PPP) data were available across all primary valuation studies in our valuation databases (see Appendix). To update all benefit and cost figures to 2012 would have involved a lot of detailed re-estimation; we do not expect such an updating to make much difference to the estimates but there was not enough time and resources to undertake that task.

The structure of this paper is as follows:

Section 2 sets out the bio-physical modelling and the baseline data and assumptions, the latter providing the rationale for some form of policy intervention, *viz*. projected losses in biodiversity under BAU. The policy options are a response to this agenda but our assessment is not based on biodiversity loss per se, as set out above; we consider benefits to arise from land cover change and the associated changes in ecosystem service provision. Section 3 sets out the three policy measures in turn, and section 4 cost estimates for each of these policy options. We do not carry out any primary analysis for cost estimation.

Sections 5 and 6 then turns to the benefit estimation. Section 5 sets out the value functions and underpinning GIS analysis (the discussion of the primary valuation databases underpinning these value functions is found in the Appendix. We present overall results for benefit estimates in section 6.

Section 7 provides the overall summary of cost-benefit estimates including sensitivity analysis, and a discussion vis-à-vis the wider implications of the study, limitations and the outcomes with regards the Copenhagen Consensus framework.

[4] http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php

2 Bio-physical modelling and Baseline Data

The Global Biodiversity Model that is used here (IMAGE-GLOBIO3) analyses biodiversity as "the remaining mean species abundance (MSA) of original species, relative to their abundance in pristine or primary vegetation, which are assumed to be not disturbed by human activities for a prolonged period" (Alkemade et al., 2009: 375). Species abundance is a measure of the population size of a species, not a measure of species richness. MSA as estimated by IMAGE-GLOBIO is a composite indicator that indexes the average abundance of original species remaining in disturbed ecosystem patches relative to the abundance in a pristine, undisturbed state.

MSA is calculated based on five drivers of biodiversity change: land use; nitrogen deposition; infrastructure; habitat fragmentation; and climate change. The model consists of a relationship between each of these pressures and the effect on MSA. Based on these 'meta analysis' functions an MSA value is calculated for a chosen area, given information on the different pressure indicators. The total MSA effect in a particular area is a multiplication of the individual pressure effects.

As the IMAGE-GLOBIO model is a geographical explicit model, the MSA value of a geographical region is calculated as the area-weighted mean of MSA values for the constituent parts, actually grid cells of about 1 by 1 km. The GLOBIO3 model is used to assess the probable impacts of the selected drivers on MSA for a number of world regions and a future no new policies scenario, as well as the impacts of specific pre-defined policy options.

2.1 Input data and assumptions in IMAGE-GLOBIO

Data for land cover and land-use in IMAGE-GLOBIO came from the IMAGE model at a resolution of 0.5 by 0.5º grid cells (*circa* 50 km by 50km). The spatial detail was increased by calculating the proportion of each land cover type within each grid cell from the Global Land Cover 2000 (GLC2000) map (Alkemade et al., 2009); GLC2000 data was at a resolution of 1 by 1km. The ten GLC2000 forest classes were converted into four land use categories using national data on forest use[5] with fractions assigned on a regional basis. The five scrubland and grassland classes were converted into three IMAGE-GLOBIO categories. Livestock grazing area was based on estimates from IMAGE, and herbaceous areas were assigned to 'pasture' if those areas were originally forest. The cultivated and managed areas class was categorised as either low-input or intensive agriculture based on regional distributions of intensity[6]; where no estimates of distribution were available intensive agriculture was assumed. The GLC2000 class of mosaic of cropland and tree cover was treated as a 50/50 mix of low-input agriculture and lightly used forest. This is summarised in Figure 1.

[5] 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

[6] 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

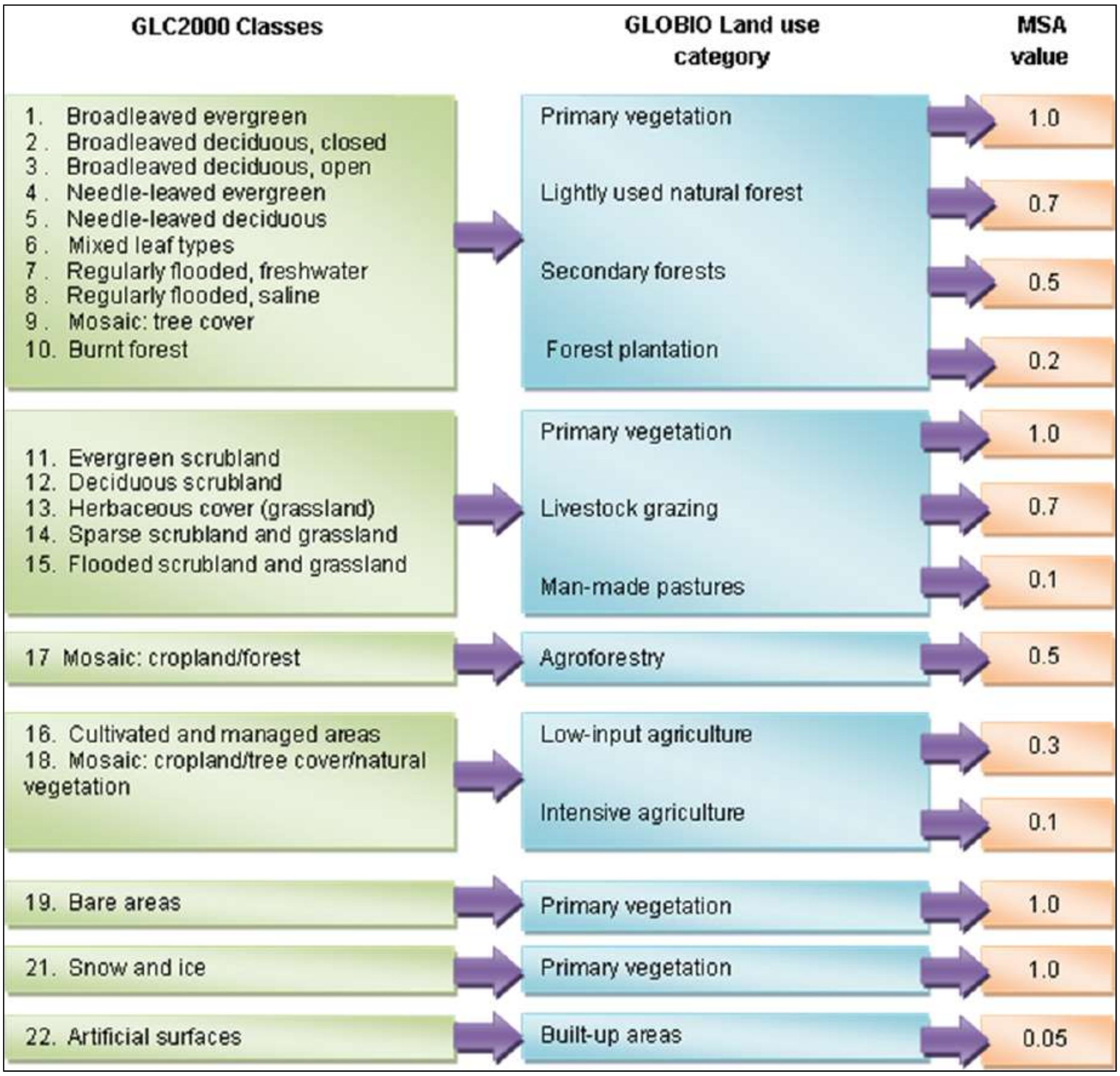

Figure 1 Conversion of GLC2000 classes to IMAGE-GLOBIO land use categories and respective MSA values (Adapted from Alkemade et al., 2009)

Land-use change within the IMAGE model was 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) were adjusted through a process of iteration between IMAGE and GTAP in which the effects of climate change and land conversion were calculated in IMAGE (PBL, 2010).

IMAGE was also used to calculate nitrogen (N) deposition, as this also affects the MSA measure. This is based on agricultural and livestock production, energy consumption and the mix of energy sources (Alkemade et al., 2009). Infrastructure data were derived from a GIS map of linear infrastructure (roads, railways, power lines and pipelines) derived from the Digital Chart of the World database[7]. Buffers representing low, medium and high impact zones were calculated for each biome (Alkemade et al., 2009).

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

The IMAGE-GLOBIO modelling framework is illustrated in Figure 2.

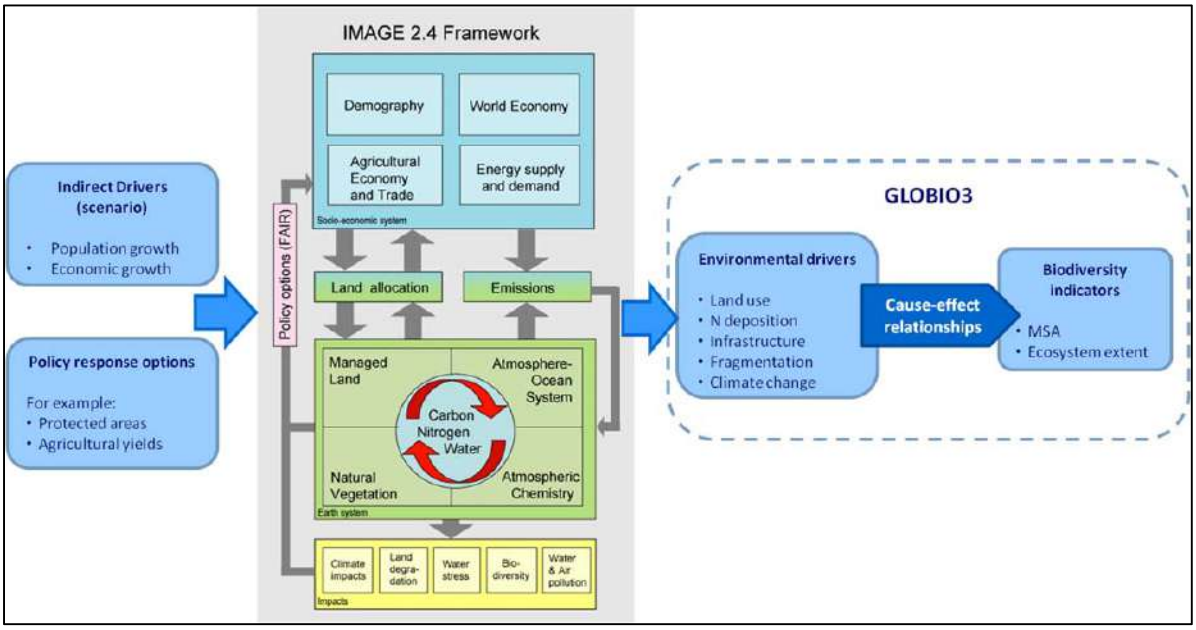

Figure 2 IMAGE-GLOBIO modeling framework

(Adapted from http://themasites.pbl.nl/en/themasites/image/model_details/index.html and http://www.globio.info/what-is-globio/how-it-works)

2.2 Losses in Mean Species Abundance 1900 to 2000

We start by examining the losses we have suffered in MSA and in terms of 'MSA area' across the main biomes and in the different regions of the world. Loss in MSA is, as stated above, the average decrease in all original populations in a particular area as compared to the original population sizes (as a percentage). Lost 'MSA area' means the hypothetical size of intact area (100% MSA) entirely lost due to human interventions (0% MSA). Table 1 provides this summary from 1900 to 2000.

Table 1 Changes in MSA area by biome and world region, period 1900-2000 (1000ha)

Biome	OECD	Central and South America	Middle East and North Africa	Sub-Saharan Africa	Russia and Central Asia	South Asia	China, East and SE Asia	Total	% on 1900 levels
Ice and tundra	-18,158	-3,200	0	0	-11,629	-1,627	-19,701	-54,316	-6%
Grassland and steppe	-174,658	-34,955	-33,475	-46,282	-58,187	-26,498	-76,689	-450,744	-27%
Scrubland and savannah	-71234	-135,528	-13,465	-234,860	0	-119,254	-118	-574,459	-26%
Boreal forests	-118,677	-4,062	0	0	-171,040	-3,478	-35,195	-332,452	-16%
Temperate forests	-278,755	-82,342	0	-11,238	-41,840	-33,326	-79,433	-526,934	-45%
Tropical forests	-13,590	-191,819	0	-76,072	0	-133,360	-1,579	-416,421	-25%
Desert	-42,642	-3,905	-49,790	-44,567	-2,793	-22,872	-9,321	-175,890	-8%
Total	**-717,715**	**-455,812**	**-96,729**	**-413,019**	**-285,489**	**-340,414**	**-222,036**	**-2,531,216**	
% on 1900 levels	-23%	-26%	-9%	-19%	-14%	-40%	-24%		-21%

Source: Derived from PBL (2010)

Globally we have lost about 21 per cent of biodiversity expressed in MSA terms in the last century, with the greatest losses in South Asia, followed by Central and South America, China, East and South East Asia and the OECD regions. In simple terms, this means that on average all populations in the world have been decreased by about 21% due to direct and indirect human interventions (see for more information www.globio.info).

2.3 Baseline scenario and results from biophysical analysis

The baseline for the IMAGE-GLOBIO projections, a no-new-policies scenario, was based on the OECD 'Environmental Outlook to 2030' report (Bakkes et al., 2008; OECD, 2009)[8], which 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:

a) Population growth from 6 to 9 billion following the UN medium scenario.
b) Per capita incomes increase in all regions, particularly in dynamic emerging economies such as the Brazil, Russia, India and China (BRIC) countries.
c) Global economic output increases fourfold (approximately 2.8% per annum) with attendant shifts in consumption patterns including increased luxury foodstuffs and livestock products.
d) Technical progress and the productivity of labour converge across countries to the long-term industrialised nation trend.
e) Yields of major staple crops increase by a factor of 1.6, i.e. an average 0.94% per annum. This is insufficient to keep pace with either increasing population or changing consumption patterns necessitating an increase in land under production.
f) Global energy use increases from 400 EJ to 900 EJ primarily from fossil fuel sources. Global average temperature increases by 1.6ºC above pre-industrial levels over the period.
g) No new policies are introduced on the environmental and global trade fronts.
h) No new measures will be taken to promote bio-fuel use or to reduce CO_2 emissions from deforestation and forest degradation.
i) There are no incentives to promote sustainable forestry. Demand for timber, pulp and firewood increase with economic and population growth.
j) The size of protected areas will remain constant (approx 10% of terrestrial area).

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

a) Biodiversity, as measured by MSA, declines from 71% in 2000 to 62% in 2050. PBL (2010). Note that it is unlikely that MSA globally could drop below 30-35% as 20% of the terrestrial land area is inaccessible; land converted to agriculture has a lower MSA limit of 5-10%.
b) The extent of natural areas will decline by 8% (10 million km^2). Natural areas are those not used for urban construction, agriculture or infrastructure. Natural areas can be intact (MSA 100%) but also be affected (low MSA). This indicator is another one produced by the IMAGE/GLOBIO model and derived from the 2010 indicators of the CBD.
c) Wilderness areas (areas with are highly intact MSA > 80%) will decline by over 11% (15 million km^2).

[8] http://www.pbl.nl/en/publications/2008/BackgroundreporttotheOECDEnvironmentalOutlookto2030
http://www.oecd.org/document/20/0,3343,en_2649_34283_39676628_1_1_1_37465,00.html

Within these global projections, some regional variations occur with MSA forecast 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). A summary is provided in Table 2.

Table 2 Changes in wilderness area by biome and world region, period 2000-2050 (1000ha)

Biome	OECD	Central and South America	Middle East and North Africa	Sub-Saharan Africa	Russia and Central Asia	South Asia	China	Total	% on 1900 levels
Ice and tundra	-23,962	-1,942	0	0	-20,945	-1,483	-12,339	-60,672	-7%
Grassland and steppe	-63,278	-14,656	-6,869	-31,315	-37,606	-17,395	-24,171	-195,289	-16%
Scrubland and savannah	-46,646	-42,645	-2,273	-197,689	0	-64,980	-39	-354,272	-22%
Boreal forests	-56,870	-2,302	0	0	-78,001	-2,079	-11,780	-151,033	-9%
Temperate forests	-52,321	-9,476	0	-9,627	-17,456	-12,091	-19,775	-120,744	-18%
Tropical forests	-1,897	-48,078	0	-58,835	0	-35,314	-436	-144,560	-12%
Desert	-38,357	-1,660	-30,055	-31,671	-9,790	-22,385	-5,205	-139,122	-7%
Total	**-283,332**	**-120,759**	**-39,197**	**-329,137**	**-163,798**	**-155,726**	**-73,745**	**-1,165,693**	
% on 2000 levels	-12%	-9%	-4%	-18%	-10%	-30%	-11%		-12%

Source: Derived from PBL (2010)

The bio-physical modelling of a suite of policy options set out in the next section is a direct response to the 2010 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).

3 Description of Policies and Measures

In order to reduce this loss in MSA the paper considers three policy options. For each of these an estimate is made of both the biophysical impacts with regards MSA and also the range of costs and possible benefits under various assumptions. The methodology used for benefit-cost estimation is set out in the next section; here we first outline the three policy options.

Underpinning this choice of policy options 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).

3.1 Investment in Agricultural Knowledge, Science and Technology

The first policy 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 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.*, 2006; Vitousek *et al.*, 1997).

The baseline scenario for this policy option is based on Rosegrant *et al.* (2009) and van Vuuren *et al.* (2009). Average yields, aggregated at global level, are projected to increase by about 1% for cereals, 0.35% for soybeans, roots and tubers, and 0.8% for fruits and vegetables, yet, gradually levelling off in most regions. Average livestock and dairy yields (i.e. production per animal) increase by 0.74% and 0.29% per year respectively.

Yield differentials can be due to a variety of factors such as soil and climatic conditions. With regards 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 Fischer *et al.* (2009): '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, 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, 2006). These parallel developments are expected to put pressure on land conversion of natural areas; expansion is projected to be of the order of a 10% increase in current agricultural area , occurring mostly in the tropical and sub-tropical zones (OECD, 2008).

The following policy 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.

The scope of our analysis is restricted to the benefits arising from changes in land cover relative to the baseline, namely reduced conversion of forest or other natural areas. However, there are a number of other benefits and costs that might arise from investment in AKST but which are not included in our assessment. Previous research focusing on market impacts has indicated that the rate of return on investment in AKST can vary between 36 and 67% (IAASTD, 2009 p539) arising from the benefits of investments in an accelerated technological growth to the existing agricultural areas, therein enhancing farmer livelihoods and potentially food security.

However, the distribution of these returns will clearly impact on the incentives driving investments. Unless private companies are able to capture benefits through technology licensing, increased input prices (seeds, agrochemical etc.) or expanded markets the incentives for investment may not be sufficient. This suggests that a role remains for public sector investment in both research and development and extension services, and consequently the implementation of the option becomes contingent on its relative position in governments' spending priorities. Beyond the need for political and commercial will to make the necessary investments in AKST, there remain barriers to technology adoption that will also need to be overcome. PBL (2010) note that new technologies and techniques need to be extensively trialed, tested and adapted to local conditions and that existing social, institutional and political conditions may need to change before farmers are prepared to make necessary management changes.

3.2 Extension of Protected Areas

This policy option explores the effects of increasing protected area coverage to 20% of 65 terrestrial ecological regions (Olson *et al.*, 2001). The 20% target was developed from earlier work for the second Global Biodiversity Outlook (sCBD and PBL, 2007). Despite the (pre-Nagoya) 10% objective set by the CBD[9], figures vary across biomes and even more across eco-regions.[10] According to Coad *et al.* (2009a), 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 is applied and thus the baseline (current) level of protection of around 10% is maintained.

This policy option results in an even representation of protected areas not only per biome but also per eco-region within each biome. While the 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.*,

[9] The post-Nagoya target for protected areas is for 2020, and includes "at least 17 per cent of terrestrial and inland water areas, and 10 per cent of coastal and marine areas"(see http://www.cbd.int/decision/cop/?id=12268 Target 11). The policy option obviously talks to the outcomes established at the Nagoya CBD COP 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.

[10] Defined by Brunckhorst (2000) as a "recurring pattern of ecosystems associated with characteristic combinations of soil and landform that characterize that region".

2001). The IMAGE-GLOBIO model 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.

3.3 Reduced deforestation (REDD variant)

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 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 account of the co-benefits of reduced deforestation, (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; this is consistent with current and proposed programmes. However our analysis focuses on value changes derived from land-cover changes. As such our methodology 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 (PBL, 2010).

The policy option assumes that *all* forests with closed tree cover (areas of closed tree cover excluding savannah, scrub and wooded tundra) are protected from conversion to agricultural land from 2000 onwards (PBL, 2010)[11]. 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 (the tropical forest biome includes both woodlands and dense forest).

The analysis of reduced deforestation through IMAGE-GLOBIO allows the assessment of potential trade-offs 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; thus an attempt is made under IMAGE-GLOBIO to capture leakage effects.

[11] The PBL (2010) bio-physical model applies this assumption, which we accept is politically unfeasible. Futurer runs of the iMAGE-GLOBIO model are set to modify this assumption.

The assumption that all dense forests are protected from deforestation will have knock on effects in that land available for agriculture (to meet rising global demand) will be reduced; consequently there will be pressure to convert other biomes such as grassland to cropping, and more land will also be required for plantation forestry to meet timber demand. The opportunity costs arising from this restriction on land conversion to agriculture form the bulk of the costs of this option as discussed further below.

4 Estimation of costs for the policy options

No primary research was carried out for the estimation of the costs. However, there are extant cost estimates that can be modified to reflect the three policy options. We have reviewed the extant literature to provide such cost estimates for each policy option in so far as this is possible. Each of the policy options is treated in turn below.

The analysis in Hussain et al. (2011) is based on a 50 year time horizon from 2000 to 2050 for the first option (agricultural productivity) and a 30 year time horizon for the second and third options (extending protected areas and reduced deforestation-REDD variant). We report the results for these time horizons. Costs and benefits in Hussain et al. (2011) are also standardised to 2007 USD, and the figures herein are also reported in 2007 USD, as discussed in section 1 above.

4.1 Agricultural productivity

This policy option 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 in 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 policy option 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.[12] 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 between IAASTD (2009) and Schmidhuber *et al* (2009) as the outcomes for which costs are estimated differ. Notwithstanding

[12] The total cost includes investments in agricultural research, irrigation, rural roads, education and clean water. The figure of US$30 billion includes spending to achieve the baseline growth in agricultural productivity.

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, and therein not achieved outcomes as predicted *ex ante*. 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, 1996; 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 both the direction and the extent of these impacts.

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 over and above BAU, 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.

4.2 Protected areas

Despite their numerous benefits, most notably in terms of positive externalities (Naidoo *et al*., 2008), 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*., 2003; 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, 2008).

This policy option refers to a precise percentage increase of 20% for the earth's 834 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.

We found seven studies on the global costs of conserving PAs with some variability with regards the types of costs estimated (

Table 3). 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.

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 billion 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.

Table 3 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

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.

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 policy option requires a cost estimate per eco-region, as defined by the CBD (Coad *et al*, 2009b) which in turn entails inevitable complexities. Data on the total coverage (km^2) of eco-regions were obtained from Coad *et al* (2009b) and Naidoo and Iwamura (2007). While the World Database on Protected Areas (WDPA[13]) provides specific regional data, and maps protected areas, we did not find a specific percentage for eco-regions. The policy option developed in PBL (2010) assumes current protection to be 10% of all eco-regions and this figure is used as the counter-factual.

We therefore estimate the total costs of achieving 20% 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 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. 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 4 summarises the outcomes across the three scenarios for an expansion across all global 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.

13 See: http://www.wdpa.org/Statistics.aspx

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.

Table 4 Estimated costs of expanding protection (billions 2007 US$).

	Scenario	Costs/annum	Marginal annual costs (assuming 10% current protection)
All eco-regions	1	30.9	15.4
	2	132.9	66.4
	3	71.3	35.6

Source: Authors' calculations (see text)

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. 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. 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 (Naidoo *et al*., 2006; Carret and Lover, 2003) that are not accounted for in our analysis. 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. TEEB, 2010).

The most defensible estimate for costs for expansion to 20% from the 10% baseline is 15.4 billion billion 2007 US$ per annum. The upper bound estimate for costs for expansion to 20% from the 10% baseline is 66.4 billion 2007 US$ per annum. 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 and 66.4 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% coverage of global eco-regions.

4.3 Reduced deforestation (REDD-variant)

There are three principal costs of implementing the 'reduced deforestation' policy option, 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[14]. The estimations used by the Stern review, for example, are in this category (Stern, 2007; Grieg-Gran, 2006).

(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 5.

[14] 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.

Table 5 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 and Robledo (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 5.

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 6).

Table 6 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).

All the estimates presented in

Table 5 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 that table would be defensible. Perhaps the most significant estimate with regards the policy perspective is Grieg-Gran (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 (2008) 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.

4.4 Summary of policy option cost estimates

Table 7 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 7 Overall summary of cost estimates for policy options (all figures billions 2007 US$).

Policy Option	Annual Cost Estimate	Programme Cost Estimate[a] 3%	5%
Agricultural productivity	14.5	372.1	264.7
Protected Areas			
'Best-guess':	15.4	304.6	239.4
Upper bound:	66.4	1304.6	1023.7
Reduced deforestation			
Lower bound:	7.9	162.6	127.2
Upper bound:	22.5	441.0	345.9

[a] Agricultural productivity 2000 to 2050; both Protected Areas and Reduced Deforestation 2000 to 2030.

[b] Plus up to US$500m pa management and transaction costs by year 10 of programme (cost rising in US$50m increments each year form year 1)

5 Benefit Assessment: value functions and GIS framework

The IMAGE-GLOBIO bio-physical model estimates: (i) ecosystem extent, (ii) Mean Species Abundance (MSA) 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 benefit estimate depends on the change in these parameters. IMAGE-GLOBIO models the Protected Areas and reduced deforestation (REDD-variant) option to 2030 and the AKST option to 2050. Thus there is a 2000 base year, a business-as-usual (no-new policies) 2030 or 2050 baseline projection, and a 2030 or 2050 scenario on which the above policy options have been superimposed (e.g. with-Protected Areas expansion). The premise for benefit valuation is as follows:

a) patches of land (e.g. a contiguous area of tropical rainforest) deliver a range of ecosystem services. Including a proxy value for biodiversity located in that patch (the ecosystem service termed ‘gene pool’);
b) one of the ecosystem services relates to carbon, and this is treated independently;
c) the patches are valued vis-à-vis ecosystem service provisioning in 2000, in 2030/2050 in the baseline scenario) and in 2030/2050 with the policy option; and
d) the marginal change in provisioning in 2030/2050 defines the net benefit of the policy option.

Patch-level analysis is applied across 2.3 million patches. A key requirement of the analysis is to determine value appropriately at patch-level. There are two elements to the estimation of benefits: (1) valuing overall changes in land-use, and (2) valuing changes in carbon.

5.1 Biome-level value functions

The valuation of changes in the extent of ecosystem patches follows the methodology set out by Brander et al. (2011), which combines meta-analytic value functions with GIS to transfer and scale up ecosystem service values. 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 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; 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).

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 8 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.

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 below after discussing the biome-level meta-regressions.

We now present the benefit functions used in this study for each biome in turn. The dependent variable in each case is US$/ha/annum in 2007 price levels. (Note that the dependent variable is not total value per site). In all cases the value functions are estimated by ordinary least squares regression (OLS) using SPSS 16.0[15].

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.

The transfer pertains to the aggregate value of ecosystem services at the study site. Although considerable effort was expended in developing the biome-level valuation database (see Appendix) the number of studies did not allow benefit transfer for individual ecosystem services or indeed for a broader split between use and non-use values.

Table 8 sets out the spatial variables used in the benefit function development and corresponding data sources.

[15] 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 8 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.
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.
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

5.2 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 benefit function outlined in Table 9 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.

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.

Table 9 Temperate forest and woodland value function

Variable	Beta	Std. Error	Sig.
Constant	28.627	6.124	0.000
Natural log of the study site area	-0.420	0.076	0.000
Natural log of Gross Cell Product within 50km radius	0.247	0.150	0.104
Natural log of urban area within 50km radius of study site	0.245	0.143	0.092
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		

5.3 Tropical forests

The average tropical forest value is US$444.98/ha/annum and the median is US$14.86/ha/annum. Table 10 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 ecosystem services. 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 10 Tropical forest value function

Variable	Beta	Std. Error	Sig.
Constant	12.960	4.071	0.002
Natural log of the study site area	-0.230	0.070	0.001
Natural log of Gross Cell Product within 50km radius	0.402	0.173	0.022
Natural log of urban area within 50km radius of study site	0.424	0.121	0.001
Natural log of human appropriation of NPP within 50km radius of study site	-0.394	0.292	0.181
Natural log of area of forest within 50km radius of study site	-0.336	0.202	0.100
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		

5.4 Grasslands

The value function for grasslands is presented in Table 11. The estimated coefficients on the explanatory variables all have the expected signs but are mostly not statistically significant[16]. 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 ecosystem services 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).

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; an alternative would be to transfer mean

[16] 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.

values. Transferred values are checked for estimates that lie outside of the range of values observed in the literature.

Table 11 Grasslands value function

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

5.5 GIS analysis: upscaling values

As mentioned above, 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)

5.6 Estimation of benefits arising from land use change

The value changes for each of the policy options are based on the three terrestrial biomes (temperate forests, tropical forests and grasslands) which were modelled in IMAGE-GLOBIO[17]. 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 3.

Table 12 presents the area of the terrestrial biomes in each region; discussions of land-use change under the policy 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.

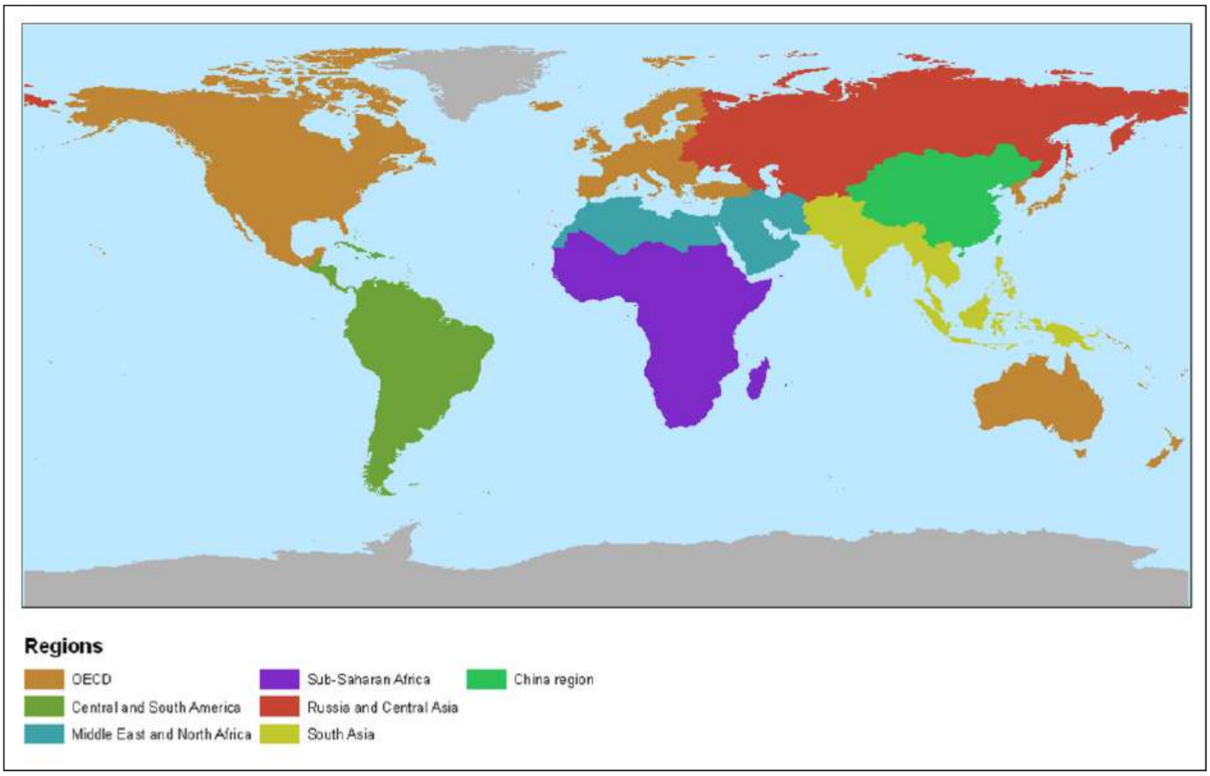

Figure 3 Map of regions

For each landscape patch (i.e. site) the value per hectare under the baselines and policy options 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 policy option vis-à-vis land-use change. Note that

[17] 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' respectively.

in some cases land-use change can have net costs if there is a switch from higher-value to lower-value land-use types.

Table 12 Baseline area of terrestrial biomes considered in analysis ('000 km^2)

	Grassland	Temperate Forest	Tropical Forest
OECD	14,197.4	9,663.4	872.7
Central and South America	4,255.2	789.3	6,963.9
Middle East and North Africa	1,464.3	71.6	10.0
Sub-Saharan Africa	7,692.1	296.9	6,401.4
Russia and Central Asia	5,952.6	8,356.8	0.0
South Asia	1,723.9	509.8	2,525.9
China Region	3,983.7	1,940.7	84.7
Total	**39,269.1**	**21,628.4**	**16,858.6**

5.7 Estimation of benefits from mitigating carbon release

The costs of carbon (and therefore the benefits of mitigating carbon) released into the atmosphere can be derived in two ways. One is the additional cost of removing the last ton of carbon so as to meet a given target reduction in emissions by a given date. This 'marginal cost of abatement or MAC' depends of course on how strict the target is and how quickly it is to be achieved. The other method is to estimate the damages per ton based on the future pathway for the economy and the amount of damages caused by a small increase in emissions in terms of losses through higher temperatures, extreme events, changes in rainfall etc. These 'marginal damage or MD' based estimates depend very much on the expected physical changes from increased carbon emissions and on the monetary values associated with them. Both are highly uncertain.

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 (CO_2 and Nitrogen fertilisation). The policy options generally impact upon all three. For instance, a reduction 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. Table 13 provides the bio-physical data on changes in carbon storage for the three policy options. The economic benefit appraisal is set out in each respective sub-section of Section 6 Results.

Table 13 Modeled projections of changes in net carbon storage relative to the baseline (billion tonnes CO_2-equivalent)

	Agricultural Productivity (high AKST)	Protected Areas (20%)	REDD
2000	0.00	-0.29	0.71
2005	-0.57	-0.04	6.34
2010	0.04	-0.02	6.71
2015	1.39	0.66	9.16
2020	0.96	0.49	8.65
2025	2.49	0.31	8.05
2030	2.41	0.76	6.97
2035	3.05		
2040	2.85		
2045	3.70		
2050	3.74		
Total (all years)	**90.88**	**8.18**	**216.53**

In the benefits reported in this study the MAC estimates are taken from the POLES model when the goal is to reduce global emissions by 80 per cent relative to 1990 levels by 2050 (Criqui et al., 1999). The MD damages are taken from the RICE model, which provides a range based on different assumptions about temperature and precipitation changes and their physical impacts (Nordhaus and Yang, 1996). The difference between the two is significant especially over time. In 2010 the MAC estimate is US$8/ton CO_2, while the MD damages and between US$7 and US$13/ton CO_2. By 2050, however the MAC estimate is US$406/ton CO_2, while the MD range is US$21-61/ton CO_2. In the analysis of the different options we consider both the MAC estimate based on POLES and the MD estimate based on the mean of the figures from the RICE model.

5.8 Assessment of benefits: systemic under-estimation

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

(1)This study focuses on valuing changes in land-cover, i.e. the quantity of land-cover 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, e.g. 'human appropriation of net primary product' (HANPP) as a proxy for intensity of land-use and 'roads' as a proxy for habitat fragmentation; such variables are likely to only partially capture changes in habitat quality. The only alternative is to infer changes in quality from MSA changes, but this requires mapping changes in MSA to changes in ecosystem service provision; the evidence base from the scientific literature to support these inferences is limited and thus no attempt is made to do so in this study. 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, values are not transferred across ecosystem service categories. Valuation estimates from primary studies are used 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. 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 ecosystem services is generic to environmental valuation studies and to site-level benefits transfer (as opposed to ecosystem service-level transfer).

(3) Our value estimations for the policy options 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, but these are not considered in this paper.

6 Overall Results

Results are presented in turn for agricultural productivity (high investment in Agricultural Knowledge, Science and Technology), extension of Protected Areas to 20% of eco-regions and reduced deforestation (REDD-variant).

6.1 Agricultural productivity

As noted earlier, 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: cumulated growth in productivity of 60% to 2050 relative to productivity in 2000. Under the policy option, 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 4 presents the percentage changes in land-use for each biome under the high AKST policy option 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. The results are presented by region and by biome (Table 14) with the overall aggregated results from 2000 to 2050 at two discount rates (3% and 5%) for the high AKST scenario (Table 15).

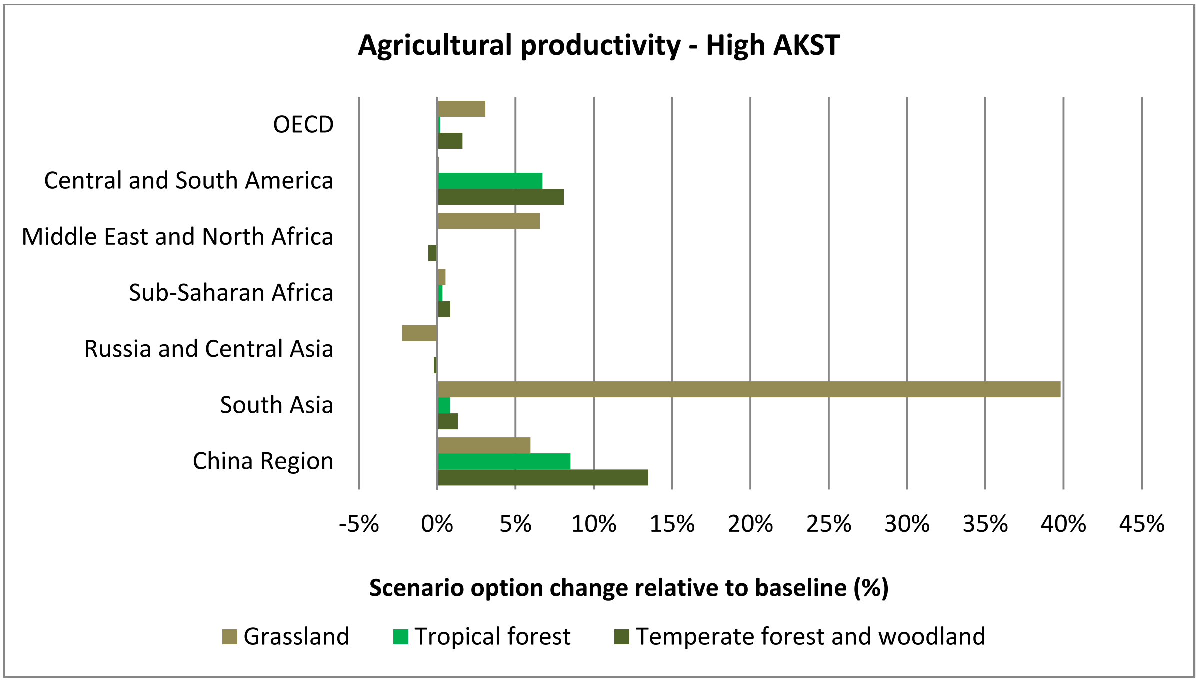

Figure 4 Agricultural productivity: change in area of biomes for policy option relative to the baseline

Table 14 shows the breakdown on a biome-by-biome basis. The sum of the value changes across the three biomes are US$ 27.8 billion for grasslands; US$81.7 billion for temperate forest; US$52.6 billion for tropical forests. These are summed in the summary table (Table 15) second column, i.e. US$162.1. All values are in 2007 prices. This presentational style is repeated for all the policy options below.

Table 14 Agricultural productivity: 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.

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[18]. The patch size coefficient for each biome is negative indicating that larger patches have lower per ha values.

Globally, the land-use value change is significantly positive, i.e. around US$1,631 billion at the 3% discount rate (see Table 15). Note that these gains are against the 'moderate' counter-factual of (on

[18] 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.

average) 0.94% growth in productivity per year. The results show that there are significant welfare gains associated with the high AKST policy option across the three biomes; however there are some variations across regions. Specifically the ‘Russia and Central Asia’ region sees a loss in welfare of US$5.5 billion 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.

Table 15 Annual and discounted aggregated regional benefits (billions 2007 US$) of agricultural productivity increase versus 2050 baseline

	2050 undiscounted annual benefit	2000 – 2050 discounted total benefit	
		3%	5%
OECD	47.3	476.4	280.4
Central and South America	65.1	655.4	385.7
Middle East and North Africa	2.3	23.0	13.5
Sub-Saharan Africa	1.3	13.1	7.7
Russia and Central Asia	-5.5	-55.4	-32.6
South Asia	9.6	96.6	56.9
China Region	41.9	422.1	248.4
Total	**162.1**	**1631.3**	**960.1**

The undiscounted annual benefit in 2050 (162.1 billion US$) is the figure for the *year* 2050, i.e. the end of the study period 2000 to 2050. We assume a linear trajectory of benefits from 2000 to 2050. This is represented in Figure 5.

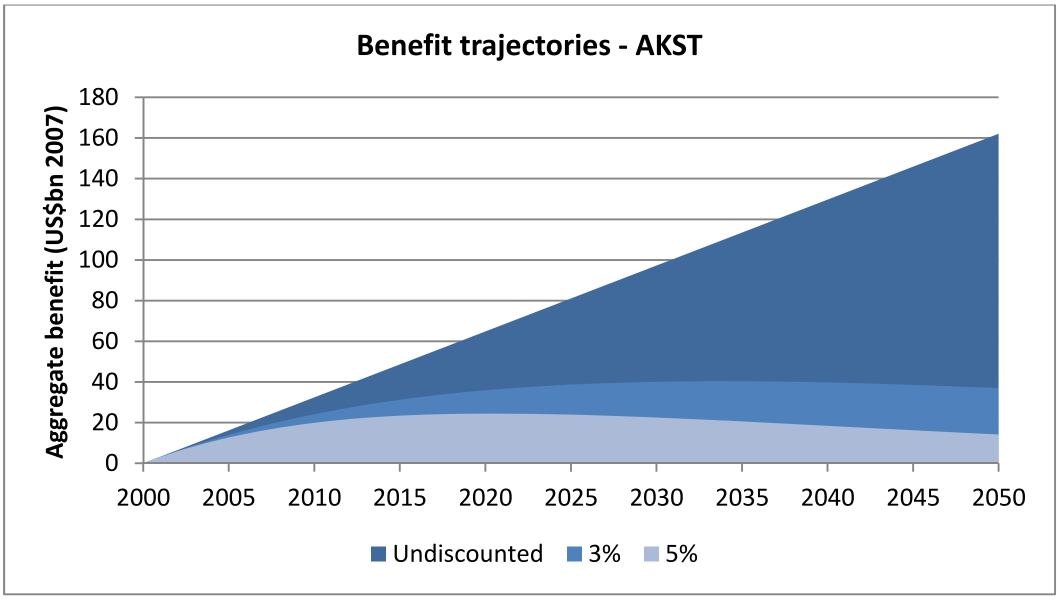

Figure 5 Linear benefit trajectory for increased productivity: undiscounted and discounted benefit estimates over the study period 2000 to 2050

Given the development-focused nature of the policy option, 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 between US$720 million and US$6,019 billion (see Table 16). Costs were estimated to be US$373 billion at a 3% discount rate. The benefit-cost ratios across a range of discount rates and carbon values are set out in Table 16.

Table 16 Overall benefit-cost ratios for agricultural productivity (2000 to 2050)

		Discount rate	
		3%	**5%**
Benefits of change in biome areas (bn US$2007)		1,631	960
Carbon values (bn US$2007)			
	POLES	6,019	3,166
	RICE-Mean	1,182	720
Costs (bn US$2007)		373	265
Benefit/cost ratios			
No carbon value		4.4	3.6
Carbon value Based on MAC (POLES)		20.5	15.6
Carbon value based on MD (Rice-Mean)		7.5	6.3

Even without adding the additional carbon storage estimated to occur with the policy option, the benefit/cost ratio is significantly positive, i.e. 3.6 with the higher 5% discount rate. The majority of the benefits from land-use change come from the forest biomes: of the US$162.1 billion undiscounted annual benefit US$27.8 billion is attributed to the grasslands biome (see Table 14), i.e. 17%. This is significant. We also note that 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 (US$510.3 bn) and the carbon benefits leaves a significant positive benefit-cost ratio (~3 at a 3% discount rate). Adding the carbon benefits raises the benefit/cost ratio to between 8 and 21 at the 3% discount rate and between 6 and 16 at the 5% discount rate.

We can say therefore, with very high confidence, that this policy option is economically efficient on the basis of land-use change alone.

6.2 Protected areas

The baseline scenario assumes that the current system of protected areas in maintained. No further policy interventions are assumed. The policy option assumes an increase in protected area coverage to 20% in 65 identified ecological regions.

Figure 6 presents the bio-physical changes in land-use relative to the baseline. Note that the BAU and policy option pertain to 2030 (not 2050); the bio-physical modelling is to 2030 (PBL, 2010). The area of grassland increases in each region under this policy option; 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. Grasslands are underrepresented in the existing network of protected areas and under the baseline these are more likely to be converted to agricultural use than forests. The policy option specifically aims to achieve a more representative network of protected areas across biomes, and consequently there will be displacement of land conversion into forest areas. Value results by region and by biome are presented in Table 17 and Table 18 presents overall results with two discount rates.

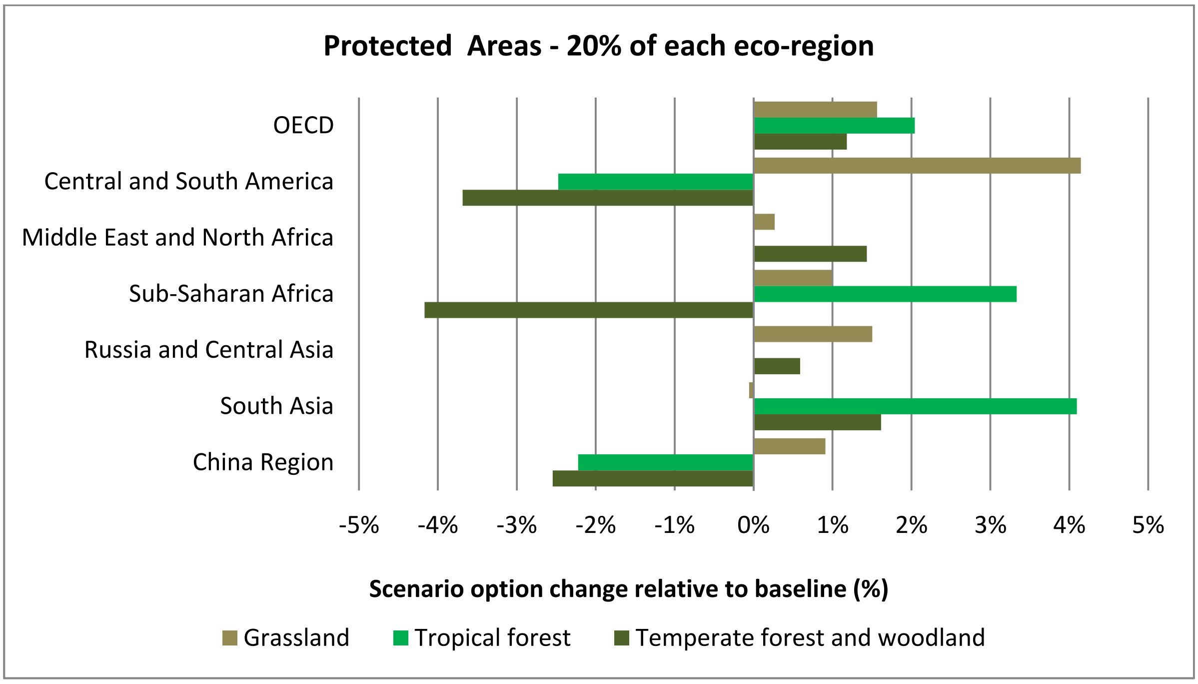

Figure 6 Protected Areas: change in area of biomes for scenario option relative to the baseline

The overall results for value changes from land-use change are positive with the exception of 'Central and South America', 'Sub-Saharan Africa' and 'China region', and there are wide variations regionally. As indicated in Table 17 the benefits from this policy arise mainly from increases in the area of the grassland biome (609,000 km^2). This compares to a more modest, although valuable increases in temperate forest (89,200 km^2) and tropical forests (111,900 km^2). The large contribution of grassland values to the total value (43%) arises 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 19 presents overall benefit/cost results for the protected areas option. Taking the 'best guess' estimate for costs (Scenario 1 for PAs reported above), the benefit/cost ratio at a 3% discount rate is 1 with no carbon values and 1.4 with carbon values based on MAC and 1.2 with carbon values based on MD. With a discount rate of 5% the benefit/cost ratio is <1 with no carbon value and between 1.0 and 1-2 with carbon values, depending on whether we take the MAC or MD figures. If, however, we take the higher estimate of costs the benefit cost ratio is below one in all cases, including those where carbon is added to the benefits.

Table 17 Protected areas: 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	20,132.1	17.8
Central and South America	-27.0	16,988.0	-8.7
Middle East and North Africa	1.0	15,582.6	0.4
Sub-Saharan Africa	-5.9	10,468.4	-0.6
Russia and Central Asia	49.2	17,217.6	4.9
South Asia	8.1	9,118.5	1.6
China Region	-47.2	15,281.9	-6.7
Total	**89.2**		**8.7**
Tropical Forest			
OECD	17.8	9,785.8	6.2
Central and South America	-161.3	8,063.5	-16.6
Middle East and North Africa			
Sub-Saharan Africa	154.3	4,219.7	6.5
Russia and Central Asia			
South Asia	102.9	7,071.8	15.3
China Region	-1.8	8,104.3	-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.

Table 18 Annual and discounted aggregated regional benefits of protect areas versus 2030 baseline

	2030 undiscounted annual benefit	2000 – 2030 discounted total benefit	
		3%	5%
OECD	32.7	284.2	200.4
Central and South America	-22.4	-195.3	-137.7
Middle East and North Africa	0.4	3.8	2.7
Sub-Saharan Africa	6.5	56.5	39.8
Russia and Central Asia	7.1	61.4	43.3
South Asia	16.9	146.8	103.5
China Region	-6.7	-58.5	-41.3
Total	**34.4**	**299.0**	**210.8**

The undiscounted annual benefit in 2030 (34.4 billion US$) is the figure for the *year* 2030, i.e. the end of the study period 2000 to 2050. We assume a linear trajectory of benefits from 2000 to 2030. This is represented in Figure 7.

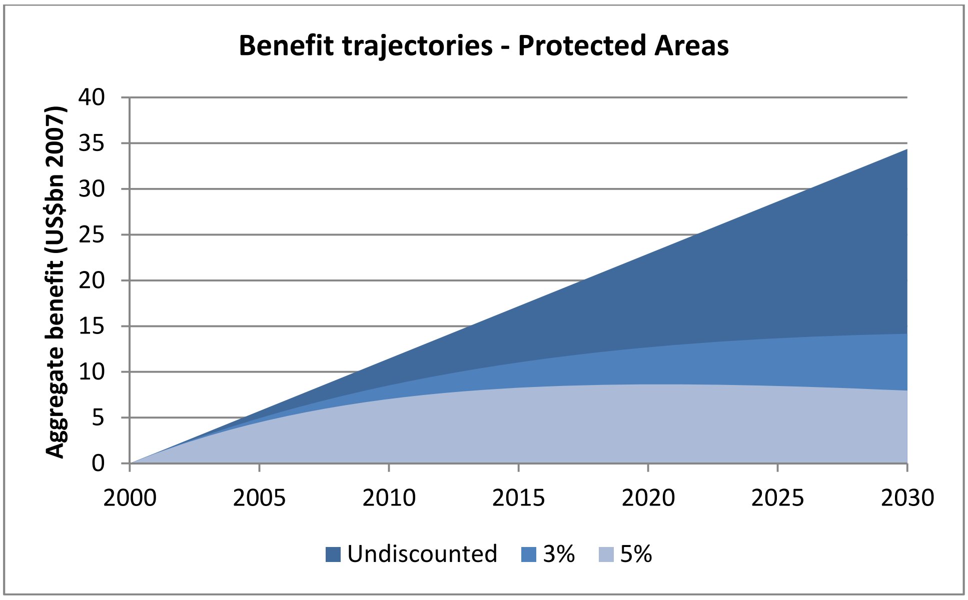

Figure 7 Linear benefit trajectory for protected areas: undiscounted and discounted benefit estimates over the study period 2000 to 2030

Table 19 Overall benefit-cost ratios for protected areas

		Discount rate	
		3%	5%
Benefits of change in biome areas (bn US$2007)		299	211
Carbon values (bn US$2007)			
	POLES	132	70
	RICE-Mean	63	39
Costs (bn US$2007)			
	'Best guess'	305	239
	Upper	1,305	1,024
Benefit/cost ratios			
No carbon value	'Best guess'	1.0	0.9
	Upper	0.2	0.2
Carbon value based on MAC (POLES)	'Best guess'	1.4	1.2
	Upper	0.3	0.3
Carbon Value based on MD (Rice-Mean)	'Best guess'	1.2	1.0
	Upper	0.3	0.2

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 protected areas 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 ecosystem services. The protected status of a site will alter the mix of ecosystem services 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 ecosystem services for different sites as our value functions implicitly assume an 'average' level of ecosystem service provision.

We conclude for the chosen discount rates that the protected areas option is only economically efficient for our lower bound cost estimate and then only when the associated carbon benefits are included.

6.3 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 changes as land-use is converted for agriculture. The policy option assumes the protection of all dense forests 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.

Figure 8 presents changes in the extent of grassland and forest biomes for the reduced deforestation policy option relative to the baseline. The changes in land use for this 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 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.

Value results by region and by biome are presented in Table 20and Table 21 presents overall results for the two discount rates.

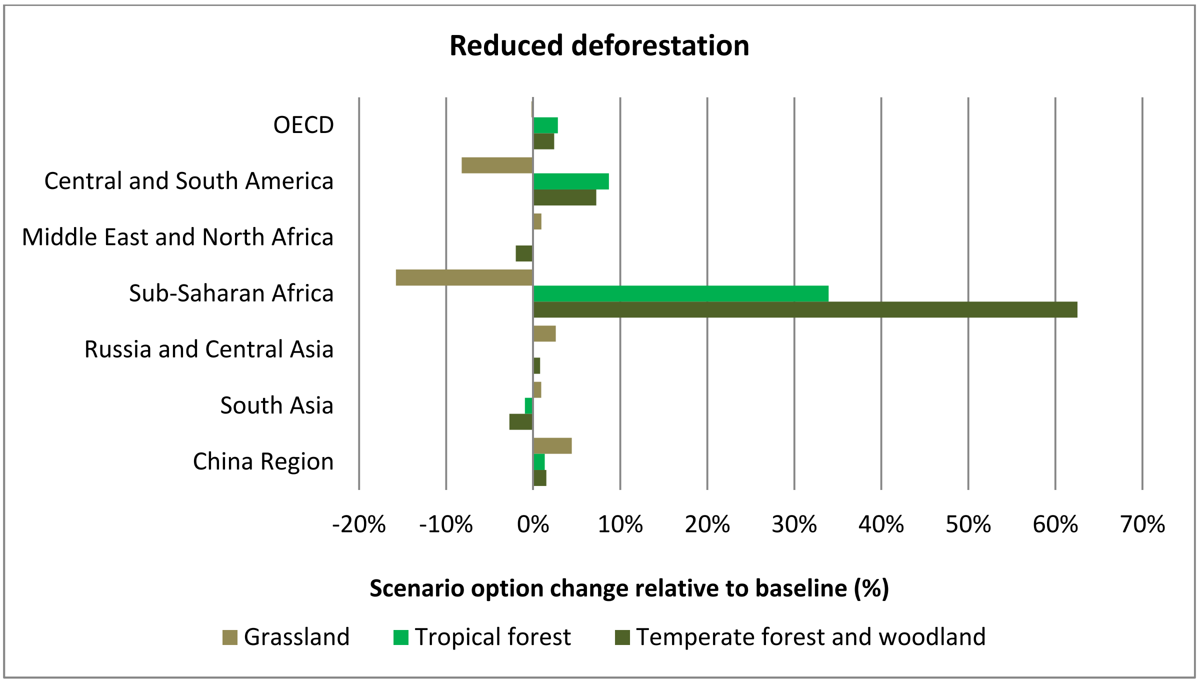

Figure 8 Reduced deforestation: change in area of biomes for scenario option relative to the baseline

Table 20 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	-1,481.2	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	**-1,529.4**		**-9.8**
Temperate Forest			
OECD	226.2	20,081.6	36.3
Central and South America	53.0	16,612.6	16.7
Middle East and North Africa	-1.4	15,695.4	-0.5
Sub-Saharan Africa	87.8	9,426.7	7.9
Russia and Central Asia	66.8	17,210.1	6.6
South Asia	-13.8	9,202.2	-2.7
China Region	28.0	15,152.3	3.9
Total	**446.6**		**68.3**
Tropical Forest			
OECD	24.8	9,781.3	8.6
Central and South America	567.5	7,955.8	57.1
Middle East and North Africa			
Sub-Saharan Africa	1,571.9	3,991.8	62.1
Russia and Central Asia			
South Asia	-23.8	7,131.7	-3.6
China Region	1.1	8,057.2	0.2
Total	**2,141.5**		**124.3**

Mean per ha values are the average of 2030 baseline and 2030 policy option scenario per ha values.
Total value changes are sum of individual patch values and are not calculated from regional mean per ha values.

Table 21 Annual and discounted aggregated regional benefits of reduced deforestation (REDD) option.

	2030 undiscounted annual benefit	2000 – 2030 discounted total benefit	
		3%	5%
OECD	43.8	381.1	268.7
Central and South America	68.1	592.7	417.9
Middle East and North Africa	-0.2	-1.6	-1.1
Sub-Saharan Africa	60.9	529.4	373.3
Russia and Central Asia	10.4	90.4	63.7
South Asia	-6.1	-53.4	-37.7
China Region	5.9	51.7	36.5
Total	**182.8**	**1,590.4**	**1,121.3**

The undiscounted annual benefit in 2030 (182.8 billion US$) is the figure for the *year* 2030, i.e. the end of the study period 2000 to 2050. We assume a linear trajectory of benefits from 2000 to 2030. This is represented in Figure 9.

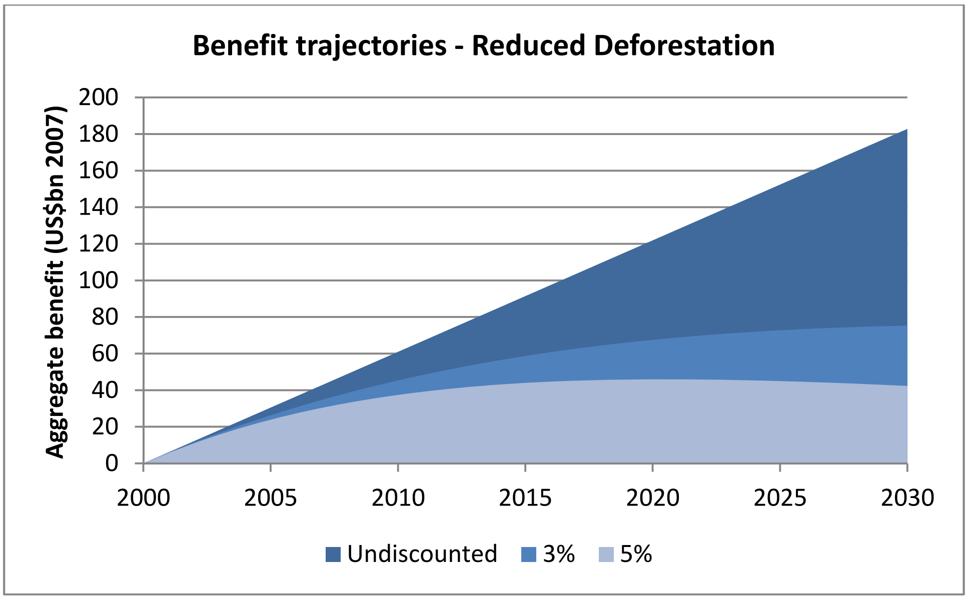

Figure 9 Linear benefit trajectory for reduced deforestation: undiscounted and discounted benefit estimates over the study period 2000 to 2030

The overall results show strongly positive net benefits for the reduced deforestation/REDD option. 'Central and South America' benefits significantly, as does 'Sub-Saharan Africa'. Considerable benefits are also observed for the 'OECD' region. The loss in forest area observed in the 'South Asia' region (and to a lesser extent 'Middle East and North Africa') arises due to losses of woodlands of lower density that are not protected by the policy option.

We set out the benefit/cost results in for reduced deforestation in Table 22. A worst-case scenario (applying the upper estimate for costs and excluding additional carbon storage benefits) realises a benefit/cost ratio of 3.2 at a 5% discount rate. With the value of carbon added and the lower cost estimate, the benefit/cost ratio is a very high value (as much as 31.3 at a 3% discount rate and 27.8 at a 5% discount rate). These values do account for changes in the opportunity cost of agricultural land.

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 22 Overall benefit-cost ratios for reduced deforestation (REDD)

		Discount rate	
		3%	5%
Benefits of change in biome areas (bn US$2007)		1,590	1,121
Carbon values (bn US$2007)			
	POLES	3,522	2,408
	RICE - Mean	1,866	1,369
Costs (bn US$2007)			
	Lower	163	127
	Upper	441	346
Benefit/cost ratios			
No carbon value	Lower	9.8	8.8
	Upper	3.6	3.2
Carbon value based on MAC (POLES)	Lower	31.3	27.8
	Upper	11.6	10.2
Carbon Value based on MD (Rice-Mean)	Lower	21.2	19.6
	Upper	7.8	7.2

7 Conclusion

This paper has analysed the challenge of ecosystems and biodiversity. Under business as usual there will be a significant loss of biodiversity over the next 40 years: our estimates indicate that globally it could be around 12 percent, with South Asia facing a loss of around 30 percent and Sub-Saharan Africa of 18 percent. These losses have a significant value, based on the services that the different biomes provide. These include timber and other forest products, genetic materials, recreational and cultural uses of the biomes, non-use values and carbon values. They have been estimated in monetary terms in a number of studies for the three main biomes (temperate and tropical forests, and grasslands, using a meta-analysis linking the unit values of the services in each biome to the characteristics of the particular patch of biome over which the estimates were made. From this meta-analysis we derive figures for the losses that will occur when any patch of the same biome is lost. This approach is applied to all biomes and ecosystem services except for carbon values which are based on a review of the literature. For the carbon values a range is taken, with the lower bound based on marginal damage studies and an upper bound based on the marginal costs of abatement arising from a target of a 50 per cent global reduction in emissions by 2050.

The study looked at three interventions relative to the business as usual. The first was an increase in agricultural productivity (20 percent for crops and 40 percent for livestock), which reduces pressure on land. The benefit cost ratios for this program were very favourable: with a total cost over the period 2000 to 2050 of US$373 billion at a 3% discount rate the non-carbon benefits alone were well in excess of that. If we take the carbon benefits valued using the MD (the lower of the two unit values) the ratio goes to 7.5 at a 3% discount rate and around 6 at a discount rate of 5%. Hence we would argue that there is a strong case for such a program.

The second program was to increase the amount of protected areas globally to around 20 percent of all land across a large number of ecological regions. Currently such areas account for around 10 percent of all land. There are obvious benefits from this but there are also significant costs, principally the loss of output from the land that is taken out of use. The net benefits are very much dependent on what cost estimates are taken as valid. With these figures set at the best guess, the program was just beneficial with the lower of the carbon values. If, however, the costs were at the upper end, the program did not have a benefit cost ratio of more than one even with higher carbon values. This suggests that only a selective increase in protected areas is warranted – in situations where the opportunity costs are low and the ecosystem services gained are high.

A further comment about protected areas is warranted. The main reason for these programmes is really to enhance biodiversity conservation and our methods of estimation do not fully capture those benefits. Hence the assessment made here underestimates the benefits of such policies.

The final program was one that sought to prevent all dense forests from conversion. In this case the benefits are very high and while there is considerable uncertainty about the costs (the upper bound is more than four times the lower bound) the benefit cost ratio exceeds one even with the higher cost figures and without the carbon values. When the latter were included the ratio went well above one, indicating that such a program would be very attractive.

The challenge specifies that the amount available is around US$75 billion per year for four years. The amounts involved in these programs are in excess of that figure but there are spread out over a longer period as well: over 50 years in the case of the first intervention and over 30 years in the case of protected areas and reduced deforestation. The detailed analysis did not indicate that there was any notable non-linearity in the programs; in other words the benefit-cost ratios should not be significantly different if the programs were conducted at a fraction of the scale considered here. In fact one could argue that a smaller program could have a higher benefit to cost ratio if one could pick out the areas where it was applied so as to keep the costs lower and the benefits higher. This should certainly be possible in the case of the reduced deforestation option, although perhaps less so to the increasing agricultural productivity option. In any event these two options could easily share the budget of US$75 billion over four years (possibly spending it over a longer period) and generate benefits that would result in benefit cost ratios similar to the ones reported here. One caveat we have add is that these programs cannot be readily aggregated. Consequently there is likely to be significant double counting if the programs are combined, i.e., changes in land cover for any one patch may apply over more than one program, e.g. any given patch of forest might not be converted to pasture under high AKST and would not be converted to another use under reduced deforestation.

One final remark about the methodology that merits consideration is the fact that it is based on a partial equilibrium analysis. That is to say, changes in biomes are valued on the assumption that the amounts involved are small compared to the total size of the biome and the services it provides. If that assumption is not valid then the estimates of changes will be flawed to the extent that other prices, as well as unit values of the services themselves may change. We have been at pains to note the size of the change in biomes and services are relatively small but that is a matter of judgment and in one or two cases the proposed measures may be considered as possibly non-marginal. In that case there may have been an overestimate of the benefits.

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Appendix: Databases of biome-level primary valuation studies

In this section 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.

Benefit database development

The valuation studies used for the benefit transfer were identified from the TEEB valuation database (van der Ploeg *et al.*, 2010) developed at Wageningen (forest and grassland biomes). The TEEB database contains 1,298 individual entries across 14 biomes 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.

The major task in our 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. 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 value function estimation.

Following completion of the review a number of studies in each biome were considered unsuitable for inclusion in our 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 (ESSs), 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 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[19] (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

[19] Estimates were obtained for household numbers in Denmark, Finland and Australia (Queensland) from national statistical agency online databases.

national GDP deflators and then converted to US$ using the relevant purchasing power parity exchange rate[20].

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 to the dataset used in this study. These site-specific variables are used in value function estimation and also for the subsequent value transfer.

Forest biome database description

Following the review of the TEEB database 58 temperate forest and 103 tropical forest values were selected for inclusion in our 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[21]. Table A1 summarises the ecosystem service categories represented by the values for temperate and tropical forest biomes. 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 A2 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

[20] 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.

[21] We refer in this section and throughout this report to two forest biomes: (1) temperate forests and woodland and (2) tropical forests. However Table A1 and Table A2provide the disaggregated analysis for completeness, i.e. temperate forests and presented separately to woodlands.

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 A1 (note that multiple values may have been obtained for individual sites).

Table A1 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 A2 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	

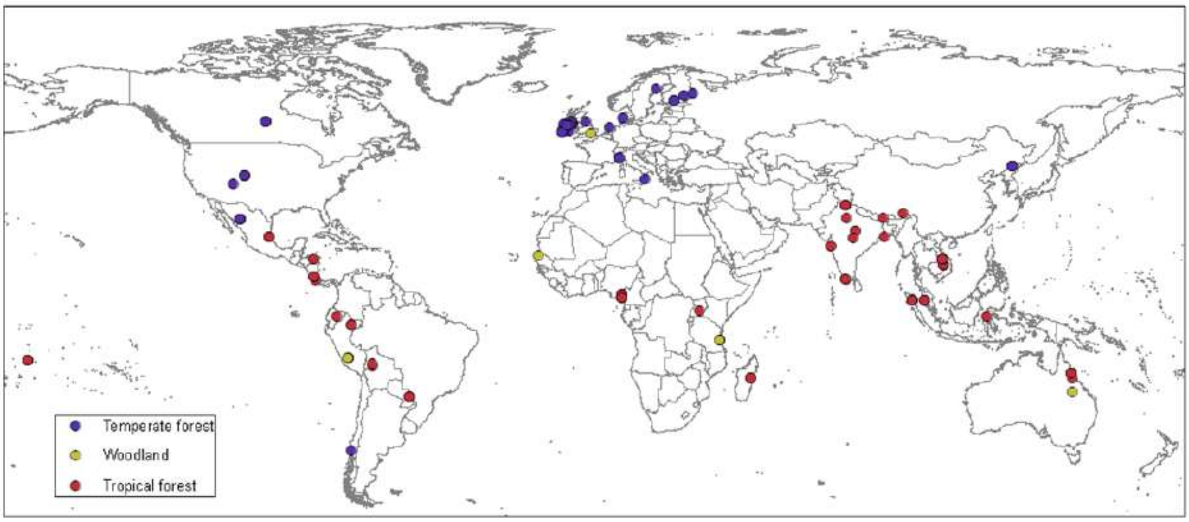

Figure A1 Forest biome site locations and services

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. 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 A3.

Table A4 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 A3 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 A4 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%