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SUMMARY OF AVAILABLE SCIENTIFIC INFORMATION ON THE LINKS BETWEEN BIODIVERSITY AND ITS CONSERVATION AND SUSTAINABLE USE AND CLIMATE CHANGE MITIGATION

Report submitted by the World Conservation Monitoring Centre of the United Nations Environment Programme

Note by the Executive Secretary

1. The terms of reference for the Second Ad Hoc Technical Expert Group (AHTEG) on Biodiversity and Climate Change include: (i) identifying options to ensure that possible actions for reducing emissions from deforestation and forest degradation do not run counter to the objectives of the Convention on Biological Diversity but rather support the conservation and sustainable use of biodiversity; and (ii) identifying opportunities for, and possible negative impacts on, biodiversity and its conservation and sustainable use, as well as livelihoods of indigenous and local communities, that may arise from reducing emissions from deforestation and forest degradation. 1/

2. In order to facilitate the consideration of this item by the AHTEG, the World Conservation Monitoring Centre of the United Nations Environment Programme (UNEP-WCMC) was contracted to prepare a review of the impacts of climate change on biodiversity, including the linkages between biodiversity and climate-change mitigation. This work was completed thanks to the financial support of the Government of the United Kingdom. It should be noted that this work contains a number of examples of impacts but is not an exhaustive list.

3. The report is reproduced in the form and language in which it was received by the Secretariat.

* UNEP/CBD/ AHTEG/BD-CC-2/1/1.

1/ Decision IX/16 B of the Conference of the Parties to the Convention on Biological Diversity, annex III, paragraphs 3 (j) and (k).

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The linkages between biodiversity and climate change mitigation

A review of the recent scientific literature

October 2008

UNEP WCMC

The United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) is the biodiversity assessment and policy implementation arm of the United Nations Environment Programme (UNEP), the world's foremost intergovernmental environmental organization. The centre has been in operation since 1989, combining scientific research with practical policy advice.

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Prepared by

Alison Campbell, Anna Chenery, Lauren Coad, Valerie Kapos, Francine Kershaw, Jörn Scharlemann, Barney Dickson

UNEP WCMC

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The linkages between biodiversity and climate change mitigation

1 Executive summary

The IPCC Fourth Assessment Report (4AR) provided growing evidence of the importance of natural ecosystems in the carbon cycle and in mitigation policies. In addition, it was recognised that climate mitigation polices focussed on reducing CO2 emissions can have impacts on biodiversity; both positive and negative.

Research since IPCC AR4 has served to strengthen the conclusion that biodiversity is important in mitigating climate change. This importance stems from the role of ecosystems in the carbon cycle. Ecosystems sequester carbon dioxide from the atmosphere and then store it. Human-induced changes in those ecosystems can lead either to increased sequestration of carbon dioxide or to increased emissions of carbon dioxide and other greenhouse gases. Promoting the former type of change and reducing the latter type of change can make a very significant contribution to climate change mitigation. The use of ecosystem-based mitigation policies can also contribute to sustaining a variety of ecosystem services including biodiversity conservation.

There is considerable uncertainty about the volume of carbon stored in terrestrial and marine ecosystems. A recent study has estimated that over 2,000 Gt carbon is stored in terrestrial ecosystems, but this figure is likely to be an under-estimate. It has been estimated that terrestrial ecosystems sequester 2.1-3 GtC of atmospheric carbon annually, approximately 30% of all anthropogenic CO2 emissions. Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface, accounting for approximately 50% of the global ecosystem uptake of CO_2.

The IPCC AR4 reported that 20% of anthropogenic GHG emissions come from the loss of terrestrial ecosystem carbon stores through land use change, primarily deforestation. This is equivalent to approximately 1.5 GtC/yr. Uncertainty surrounding estimates of emissions from tropical forest deforestation remains high and the figure of 1.5-1.6 GtC per year remains the default value. It is widely agreed that estimating emissions from forest degradation is more difficult. Some estimate that forest damage from logging in the Amazon results in a 15% reduction in carbon stocks, with increased susceptibility to fire damage releasing an additional 20% of forest carbon.

Loss of carbon from soils due to land use change is also difficult to assess, but is likely to be considerable. It has been estimated that soils lose carbon at the rate of approximately 1.6 Gt C per year, almost identical to that lost through deforestation. Much of these soil- based emissions come from peat degradation. Human disturbances such as drainage for agriculture or forestry have transformed peatland from a sink to a source in large areas. Drainage and drying of peat also facilitates fires. In combination, these processes are estimated to result in the loss of 3GtCO2 to the atmosphere every year, or 10% of global emissions.

The feedbacks from natural ecosystems due to a warming climate highlight the complex relationship between biodiversity and the carbon cycle. New observations on dampening of the carbon sink capacity are challenging the hypothesis that the carbon sequestration will be enhanced with climate change induced increases in net primary productivity.

The IPCC 4AR estimated that over the next century, 345-1269 GtCO2e could be abated through land-use based mitigation policies. This is about 15-40% of total abatement requirements and could be brought about through a combination of reduced loss of carbon stores, and sequestration policies. Since the emissions from deforestation of amount to 1.5 GtC per year, there appears to be high potential for cost-effective emissions reductions from a mechanism for Reduced Emissions from Deforestation and Degradation (REDD). This mechanism is currently in a demonstration phase in the UNFCCC. It has been estimated that a well designed REDD mechanism could reduce deforestation rates by up to 75% in 2030, and in combination with afforestation, reforestation and restoration, could make the forest sector carbon neutral. Economic modeling has suggested that REDD will be a competitive, low-cost abatement option. Moreover, a successful REDD mechanism has the potential to deliver significant additional benefits, contributing to biodiversity conservation at both the species and ecosystem level, whilst also supporting the maintenance of ecosystem services.

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned. Whilst there is significant potential in increasing the capacity of the natural carbon sink, particularly in the tropics, there is a need for more integrated study of how land management changes may affect climate change. Sequestration schemes can require a tradeoff between carbon sequestration and biodiversity benefits; however, in the long term, biodiversity generally underpins ecosystem resistance and resilience, and thereby strengthens the stability of the carbon storage.

The role of improved soil management in climate mitigation should be emphasised as it is the area with the highest potential outside of forest activities. Global soil organic carbon has a sequestration potential 0.6-1.2 GtC with high levels of carbon stocks, much of which is contained under natural ecosystems rather than managed ecosystems. Whilst estimates of carbon storage in peat soil are still uncertain, largely due to lack of information on peat depth and density, advances are being made in this respect. A new estimate of 5GtC stored in Indonesian peat utilises remote sensing technology supported by ground based observations. The reduction in the rate of current peat degradation in Indonesia therefore has the potential to reduce emissions significantly, particularly as deforestation on peat soils is accelerating. Boreal regions have significant areas of peatland, acting as large carbon sink. But there is peat degradation there too. Many peat bogs in Europe have been drained and are being restored and over 55% of peatland area in Finland has been drained. Currently, there is very limited scope for inclusion of wetland or peatland in carbon accounting through the UNFCCC, and no direct mention in the text. The only option for inclusion in carbon accounting is where conversion of wetland areas is captured through management practices of other ecosystems, such as for forested peatland

Geo-engineering techniques for mitigating climate change are not strictly ‘ecosystem-based’, but they do involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity and this may have impacts on biodiversity. The technique with the most promise for mitigation is carbon capture and storage. This may involve the injection of CO2 into the deepwater and this will alter ocean chemistry and could have significant consequences for marine organisms and ecosystems in the deep sea.

Renewable energy projects can also have impacts on biodiversity. Biofuel production has considerable impacts on biodiversity when it results in direct conversion of natural ecosystems and indirect displacement of agricultural land into natural ecosystems. Birds can be affected by wind turbines though collision with turbine blades, displacement from migration routes, and direct habitat loss. Mortality of birds as a result of wind turbines has been documented by a number of recent studies although some have argued that windfarm impact studies lack an evidence base and have minimal impacts on biodiversity. The biodiversity impacts of hydro-electric dams include habitat destruction, barriers to terrestrial migration barriers to fish migration, reduced sedimentation and changes in flow altering downstream ecosystems, and fish mortality in turbines.

It is clear from the literature reviewed that climate change mitigation policy has the potential to impact biodiversity both positively and negatively. Currently, many renewable energy projects are being planned without consideration for biodiversity impacts; as are some land-based mitigation strategies such as monoculture plantations. However, due to the important role of ecosystems in the carbon cycle, it is clear that the potential exists to develop 'win-'win' mitigation policies that are beneficial for both climate change mitigation and biodiversity.

2 Introduction

The overall objective of the United Nations Framework Convention on Climate Change (UNFCCC) is the *'stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system'* (UNFCCC Article 2). In order to achieve this, global average temperatures should not increase above 2ºC relative to pre-industrial levels; requiring a 60-80% reduction in greenhouse gas (GHG) emissions by 2050 to stabilise atmospheric concentrations at 445-490ppm CO_2e (IPCC 2007). Therefore, mitigation efforts are required across all sectors, including through efforts to reduce emissions from land use change and increase the capacity of the natural carbon sink.

An increase in global average temperature of 0.7 ºC has already been observed, with associated impacts on natural ecosystems and the services that they provide. Increasing temperatures are causing rising sea levels, melting sea ice, altered precipitation patterns and fire regimes, and are likely causing the altered frequency and severity of extreme events such as drought, heat waves, and hurricanes. Such impacts will have significant implications for human welfare (Stern 2007).

The Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) (Nakicenovic & Swart 2000) developed a number of narratives on how the world might develop in the 21st century; exploring impacts on global emissions if demographic, social, economic, technological, and environmental developments take specific directions at the global level. These 'storylines', labelled A1, A2, B1 and B2, do not take into account the implementation of mitigation policies. Despite developments in population models since their development, the SRES projections are still considered to be representative of the range of likely outcomes (IPCC, 2007), and include land-based GHG emissions as an important source throughout the century; including continued but reducing land use change, and an increase in intensity of agricultural practices. There is, however, some evidence that emissions have been increasing at higher rates than those projected by SRES storylines (Raupach *et al.* 2007).

In addition, feedbacks from ecosystems as a result of climate change and land use change are significant, but are not incorporated into climate models because uncertainty is high (IPCC

2007). This is turn can lead to uncertainties in projections of future climate change, and therefore formulation of mitigation strategies (Strassmann, Joos & Fischer 2008). Despite this, the IPCC Fourth Assessment Report (4AR; IPCC 2007) provided growing evidence of the importance of natural ecosystems in the carbon cycle and therefore in mitigation policies. In addition, it was recognised that climate mitigation polices focussed on reducing CO_2 emissions can have impacts on biodiversity; both positive and negative.

This report reviews the literature published since the IPCC 4AR on the linkages between biodiversity and climate mitigation policies in order to highlight developments in our understanding of the role of biodiversity in climate mitigation, and the impacts of mitigation policies on biodiversity. Keyword searches in ISI Web of Knowledge, Scopus, and Google Scholar were carried out to obtain a broad coverage of the available literature.

3 Role of ecosystems in the carbon cycle

3.1 Carbon storage

Although it is known that both terrestrial and marine ecosystems constitute a significant carbon store, the exact figures are uncertain. Global estimates range from approximately 1500 - 2500 GtC (Cao & Woodward 1998; IPCC 2001). A recent study combining data for carbon stored in biomass (Ruesch & Gibbs 2008) with that of carbon stored in soil (IGBP-DIS 2000) has estimated that over 2,000 GtC is stored in terrestrial ecosystems (Campbell *et al.* 2008a).

Table 1. Global carbon stocks (IPCC 2001)

Biome	Global Carbon Stocks (GtC)		
	Vegetation	Soil	Total
Tropical forests	212	216	428
Temperate forests	59	100	159
Boreal forests	88	471	559
Tropical savannas	66	264	330
Temperate grasslands	9	295	304
Deserts and semideserts	8	191	199
Tundra	6	121	127
Wetlands	15	225	240
Croplands	3	128	131
Total	466	2 011	2 477

A large amount of the terrestrial carbon is stored in forest (Eliasch 2008), but there are also significant stores in other ecosystems such as grasslands and wetlands (Table 1). Carbon stored in soil accounts for a high percentage of the total terrestrial store.

Carbon storage estimates to date undoubtedly underestimate the storage of carbon in soil, particularly peat. Recent studies have suggested that there is almost 100 GtC stored in North American Arctic soils alone (Ping *et al.* 2008); 66% more than was recorded for previous estimates (Beer 2008). Indeed, (Schuur *et al.* 2008) have estimated that 1672 Gt carbon is stored in the northern circumpolar permafrost zone; equivalent to twice the atmospheric carbon pool and more than double the previous high-latitude inventory estimates. Such estimates have increased largely due to carbon stored in peat. A recent global assessment of peat has estimated that peatlands alone store 550Gt of carbon, nearly 30% of all global soil carbon, despite covering only 3% of the land area (Parish *et al.* 2008), and are therefore the most important long-term terrestrial carbon store. This could be an underestimate when taking into account the peat store in permafrost (Schuur *et al.* 2008). Although there is much uncertainty over the exact figure, particularly as peat depth estimates are still uncertain, this significantly increases previous estimates of the terrestrial carbon store.

Comparatively, knowledge of carbon storage within marine environments is limited, and no equivalent literature exists. However, total amount of carbon stored in the ocean has been estimated to be 50 times that of the atmosphere (IPCC 2001).

3.2 Carbon sequestration

Natural ecosystems are intrinsically linked to the carbon cycle. In addition to the historical carbon store, ecosystems take carbon out of the atmosphere, a process known as sequestration.

It has been estimated that terrestrial ecosystems sequester 2.1-3GtC of atmospheric carbon annually (Luyssaert *et al.* 2007; Canadell & Raupach 2008), approximately 30% of all anthropogenic CO_2 emissions. Much of this is realized by forest (Luyssaert *et al.* 2007); although over the past 10,000 years peatlands have sequestered an estimated 1.2 trillion tonnes of CO_2 (Parish *et al.* 2008). The Luyssaert *et al.* (2007) estimate for forest is based on a global database of flux observations, updated since the IPCC AR4.

Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface, a process that accounts for approximately 50% of the global ecosystem uptake of CO_2 (Arrigo 2007). Some of this carbon is pumped into the deepwater both through the food chain and through physical processes. The role of coastal margins is less well understood, although it is known that mangroves and seagrass sequester carbon (Yin *et al.* 2006).

3.3 Emissions from deforestation

The IPCC AR4 (IPCC 2007) reported that the loss of terrestrial ecosystem carbon stores through land use change, primarily deforestation, account for 20% of anthropogenic GHG emissions; equivalent to approximately 5.8 $GtCO_2e$/yr (or 1.5 GtC). This figure was gained from estimates for tropical deforestation in the 1990s (DeFries *et al.* 2002; Houghton 2003). Recognition of the importance of emissions from such land use change has led to the commitment to include reduced emissions from deforestation and degradation in developing countries (REDD) in post 2012 commitments under the UNFCCC in the Bali Roadmap (Decision 1/CP.13; Decision 2/CP.13).

Uncertainty surrounding estimates of emissions from tropical forest deforestation remains (Achard *et al.* 2007; Olander *et al.* 2008), and the figure of 1.5-1.6 GtC per year remains the default value (Canadell & Raupach 2008). A third of these emissions come from the Amazon (Ramankutty *et al.* 2007). Recent studies have suggested that actual net tropical emissions were lower than these estimates for the 1990s (Stephens *et al.* 2007), particularly for the Brazilian Amazon, where a lower than average wood density and tree height (and therefore lower carbon stock) in the 'arc of deforestation' is not taken into account (Nogueira *et al.* 2007; Nogueira *et al.* 2008). Although this may be the case, the impact of deforestation on soil carbon is still largely unknown, with emissions dependent upon the land conversion and subsequent management practices (Murty *et al.* 2002). A disproportionate amount of deforestation in SE Asia, for example, takes place on peatland (Hoojier *et al.* 2006), and emissions from deforestation in this region are likely to be underestimates.

Despite the uncertainties over the exact figures, largely due to lack of data and differences in methodologies (Ramankutty *et al.* 2007), it is widely agreed that emissions from deforestation make a significant contribution to climate change (Laurance 2007; Eliasch 2008). In addition to releasing carbon stores to the environment, deforestation removes sequestration capacity of forest, reducing the ability of forest to act as a carbon sink (Stephens *et al.* 2007).

3.4 Emissions from forest degradation

It is widely agreed that estimating emissions from forest degradation will be more of a challenge due to the difficulties in measurement from satellite observations (DeFries *et al.* 2007; Asner *et al.* 2005). In addition, the definition of degradation is open to debate and can include unsustainable timber harvesting for commercial or subsistence use, in addition to other damaging processes such as fire and drought; all of which lead to reductions in carbon stocks (Mollicone *et al.* 2007).

Despite these issues, the need to include degradation in the REDD mechanism is widely accepted, as was established at COP13 of the UNFCCC in Bali. The area of logged and degraded forest is comparable to that deforested (DeFries *et al.* 2007; Putz *et al.* 2008; Asner *et al.* 2005; Barreto *et al.* 2006; Feldpausch *et al.* 2005; Nepstad *et al.* 2008), with significant implications for carbon stocks and biodiversity. Asner *et al.* (2005) estimate that forest damage from logging in the Amazon results in a 15% reduction in carbon stocks, and increased susceptibility to fire damage (Fearnside 2005a; Malhi *et al.* 2008) releases an additional 20% of forest carbon. This estimate of 0.08 GtC lost annually from logging increases emissions estimates from deforestation in the Amazon (Defries *et al.* 2002) by 25%. Indeed, it has recently been reported that clear-cut logging can release 40-60 % of carbon stored in vegetation (Sajwaj, Harley & Parker 2008).

In a 'business as usual' deforestation scenario, it has been estimated that 24% of the Amazon will be damaged by drought and logging (Nepstad *et al.* 2008). Forest degradation can also be a precursor to deforestation (DeFries *et al.* 2007; Putz *et al.* 2008; Asner *et al.* 2005). At present, no Parties to the UNFCCC are required to report on degradation, unless forest management has been selected as an option under Article 3.4 of the Kyoto Protocol.

3.5 Emissions from general land use change

Despite the current focus on emissions from deforestation and degradation, land use changes across all ecosystems can release significant amounts of carbon to the atmosphere. Gross historical emissions from land use change have been estimated at approximately 200 GtC

(Canadell & Raupach 2008), higher than the loss estimate of 572 $GtCO_2$ reported in the IPCC 4AR. Fires also contribute significantly to emissions, with the release of 1.7-4.1 GtC per year (Lavorel *et al.* 2007).

Although often not accurately accounted for in estimates of emissions from land use change, a recent study has suggested that soils have lost 40-90GtC to the atmosphere, and continue to lose carbon at rates of approximately 1.6 GtC per year (Smith 2008); almost identical to that lost through deforestation. Houghton (2005) has estimated that soil carbon accounted for 28% of net loss from land use change in the period 1850-1990.

Such estimates appear higher when taking peat degradation into account. Human disturbances such as drainage for agriculture or forestry have transformed peatland from a sink to a source in large areas (Parish *et al.* 2008). Drainage and drying of peat also facilitates fires. In combination, these processes are estimated to result in the loss of 3 $GtCO_2$ to the atmosphere every year, or 10% of global emissions. In El Nino years, increased fires can raise this figure (Parish *et al.* 2008). In Southeast Asia alone, emissions from peat drainage and fire average 2 $GtCO_2$ per year; equivalent to 8% of global fossil fuel emissions from just 0.2% of the land area (Hoojier *et al.* 2006), and almost twice the emissions from fossil fuel burning in Indonesia. Including emissions from peat in carbon accounting would raise Indonesia to third in the global emissions table, from 21st place (Hoojier *et al.* 2006; Uryu *et al.* 2008).

The knowledge base on carbon storage and emissions from soil and peatland is still small but developing, and is clearly and important area for further study. More generally, there is still a lack of information on carbon emissions from ecosystems such as grasslands and wetlands.

Conversion of natural ecosystems to agriculture can result in significant greenhouse gas emissions, through a combination of loss of stored carbon, and the large amounts of CH4 and N2O emitted from agricultural practices (Berry *et al.* 2008; Lal 2008). These emissions are expected to increase rapidly until the end of the century, as reported in IPCC 4AR.

3.6 Ecosystems as ‘sinks’ or ‘sources’

As the amount of carbon sequestered by ecosystems is larger than that lost, global terrestrial ecosystems are acting as a net sink of approximately 1.5GtC per year (the AR4 reported approximately 0.5-1.5 GtC). Sequestration at these levels would be equivalent to a 40-70ppm reduction of CO_2 in the atmosphere from anthropogenic emissions by 2100 (Canadell & Raupach 2008). Tropical forests account for a large proportion of this sink (Luyssaert *et al.* 2007); the absence of which would have increased the current atmospheric CO_2 concentrations by 10% (Betts *et al.* 2008a)

The exact processes involved with the oceanic carbon cycle are not well understood. However, it is clear that the ocean acts as a considerable sink; The AR4 reported that the size of the marine sink is approximately 1.8-2.6 Gt, and has increased by approximately 22% from the 1980s to the 1990s. Subsequent modeling has supported this estimate of the oceanic carbon sink (Canadell *et al.* 2007b).

A recent study taking into account fluxes of the three major greenhouse gases (CO_2, CH_4 and N_2O) has strengthened these findings, suggesting a significant role of natural and relatively unmanaged ecosystems in slowing climate change through the provision of a net yearly sink of 3.55 GtCO_2; equating to roughly 0.5ppm atmospheric CO_2 per year (Dalal & Allen 2008). Natural ecosystems are acting as a sink for 55% of anthropogenic GHG emissions (Canadell *et al.* 2007b).

Lal (2008) reports that the terrestrial sink is increasing at a net rate of 0.7 GtC per year, and is set to continue increasing due to increased CO_2 fertilisation. In addition, there is evidence that melting sea ice is increasing the sink capacity of the Arctic Ocean (Bates *et al.* 2006), and that increased CO_2 concentrations are increasing the capacity of oceanic sequestration (Riebesell *et al.* 2007); although the impacts on biodiversity of an increased ocean sink have the potential to be significant through ocean acidification (Cao 2008).

However, this is not to say that *all* ecosystems are acting as carbon ‘sinks’. There is some evidence that emissions from land use change are beginning to outweigh sequestration capacity, with the potential to reach a ‘tipping point’ whereby they will become net sources (Nepstad *et al.* 2008). Recent climate models have estimated that past land use change, largely due to cropland and agricultural expansion, has eliminated potential future carbon sinks equivalent to emissions of 80–150 GtC over this century (Strassmann *et al.* 2008). There is evidence, for example, of a reduced sink in the Southern Ocean due to changes in circulation patterns as a result of increased temperate (Le Quere *et al.* 2007), and reductions of sinks in coastal margins through loss of vegetation (Duarte, Middelburg & Caraco 2005), but modelling results are still uncertain (Baker 2007). Over the past 100 years, anthropogenic impacts have turned peatlands from a net store to a source of carbon emissions (Parish *et al.* 2008).

Recent evidence of reduced sinks (Canadell *et al.* 2007a) suggests that on a global scale terrestrial ecosystems will provide a future positive feedback of uncertain magnitude, due to altered land use practices and increasing temperatures (Heimann & Reichstein 2008).

3.7 Feedbacks to the climate system

Recent recognition of the scale of positive feedbacks to the climate system from land use change and climate impacts has further raised the relevance of biodiversity to the UNFCCC objective of limiting climate change to a 2ºC rise. Although such feedbacks are not yet incorporated into global climate change projections and are still uncertain (Baker 2007), advances in this area are being made (Chapin *et al.* 2008).

3.7.1 Feedbacks from climate change

The feedbacks from natural ecosystems due to a warming climate highlight the complex relationship between biodiversity and the carbon cycle. New observations on dampening of the carbon sink capacity are challenging the hypothesis that carbon sequestration will be enhanced with climate change induced increases in NPP (Canadell *et al.* 2007a).

It is generally agreed that one of the main feedbacks to the climate system will be through the increase in soil respiration under increased temperature, particularly in the arctic (Chapin *et al.* 2008), with the potential to add 200ppm CO_2 to the atmosphere by 2100 (Canadell *et al.* 2007a). Although the exact dynamics are still unclear, recent research has suggested that feedbacks from the two major soil carbon stores, permafrost and peatland, could be considerable (Smith et al. 2008). Estimates for emissions from the thawing of permafrost, for example, have ranged from global increases of 100 GtC by 2100, to 40-100 GtC increases from Canada and Alaska by 2100. It has also been suggested that a 10% thawing of the Siberian permafrost will release 40 GtC by 2050; an increase that will not be offset by the predicted advance of the tree line into the tundra (Ise *et al.* 2008; Schuur *et al.* 2008). Emissions on this scale would make reaching the target set of stabilization at a 2º C rise difficult.

In addition to melting permafrost, and soil respiration, peat emissions are linked to lowered water table levels, which are highly vulnerable to climate change (Ise *et al.* 2008), suggesting a need for water table management. One issue that has not received much coverage in the literature is that of potential impacts on sea level rise. It is not just increasing temperatures that can lead to such feedbacks. A study in California has suggested that inundation of the 150,000km^2 of low-lying peatlands may cause substantial emissions (Henman & Poulter 2008).

One area of research that has expanded since the 4AR is that of the projected Amazon drying and dieback. Although there is still considerable uncertainty, most models predict reduced precipitation in areas of the Amazon, which will lead to increased drying (Betts, Sanderson & Woodward 2008). Models have also suggested that CO_2 emissions will be accelerated by up to 66% by feedbacks arising from global soil carbon loss and forest dieback in Amazonia as a consequence of climate change (Betts *et al.* 2006). Again, impacts are not solely down to increasing temperature; Amazon forest dieback may also exert feedbacks through changes in the local water cycle and increases in dust emissions. This is exacerbated by deforestation and degradation, which increases the vulnerability of forest and lowers resilience for adaptation to climate change; therefore lowering the value of the Amazon in mitigation (Malhi *et al.* 2008). Climate- ecosystem feedbacks have also been implicated in droughts in the Sahel and Western Australia (Chapin *et al.* 2008).

On a global scale, climate scenario modelling suggests that the terrestrial biosphere will become a carbon source by 2100, largely due to increased soil respiration and the dieback of the Amazon. Climate models incorporating these feedbacks led to a 0.38ºC or 8% increase in warming compared to a model when feedback was not considered (Betts, Sanderson & Woodward 2008b). Such modeling is, however, still uncertain (Chapin *et al.* 2008). The interaction of the carbon cycle with the nitrogen cycle is also not generally included in climate models (Gruber & Galloway 2008); although it has been estimated that increased carbon sequestration may lead to an increase of N2O emissions in grassland (Kammann *et al.* 2008).

There are growing concerns that impacts of climate change will reduce the mitigation capacity of ecosystems; a possibility made more likely by the influence of land use change and degradation, which could potentially lower resistance to climate change impacts in addition to increasing CO_2 emissions (Malhi *et al.* 2008).

3.7.2 Feedbacks from land use change

As has been discussed previously, emissions from land use change can be significant, and will act in synergy with increasing temperature (Muller *et al.* 2007); particularly on the century time scale (Voldoire *et al.* 2007). This is not just true of the tropics. Mankind is ultimately controlling the carbon balance of temperate and boreal forests; either directly through forest management, or indirectly through nitrogen deposition (Magnani *et al.* 2007). Increased levels of pollution could impact on the carbon sink strength of ecosystems (Canadell *et al.* 2007a). There is evidence that increased nitrogen deposition causes carbon emissions from peat in Europe (Bragazza *et al.* 2006).

Impacts of land use change do not just provide feedbacks through greenhouse gas emissions. Deforestation in Amazonia can exert a large influence on precipitation patterns (Correia, Alvala & Manzi 2008). 25-50% of rainfall is recycled from forest, forming one of the most important regional ecosystem services; and removal of 35-40% of the Amazon could shift the Amazon into a permanently drier climate (Malhi *et al.* 2008). This combines with slash-and burn, logging, and degradation, to increase risk of fire (Aragao *et al.* 2008), and amplifies the climate-induced Amazon dieback described above (Betts, Sanderson & Woodward, 2008). Conversely, deforestation strongly increases precipitation during El Nino years (Da Silva, Werth & Avissar 2008). Current climate models do not incorporate these feedbacks from forest loss (Malhi *et al.* 2008; Betts, Sanderson & Woodward, 2008). Desertification and deforestation also play a large role in the monsoon and rainfall pattern in West Africa - increasing the monsoon flow over the Guinean region and reducing rainfall over the entire West African region (Abiodun *et al.* 2008).

The recent Large Scale Biosphere-Atmosphere program in Amazonia has provided mounting evidence that intact rainforests are more resilient to climate drying than current vegetation models suggest, but that a pattern of logging, degradation and fire could reduce this resilience (Bush *et al.* 2008; Malhi *et al.* 2008), potentially converting forest into 'brush' with low evapotranspiration and high albedo providing more feedback to the climate system (Nepstad *et al.* 2008). In addition, experimental evidence suggests that the forest will reach a drought threshold where resilience is lost, emphasizing the need for combined mitigation and adaptation to climate change (Nepstad *et al.* 2008). This suggests that mitigation strategies aimed at protecting forest and reducing forest degradation could play a significant role in reducing the impacts of climate change on biodiversity and ecosystems services such as water cycling, particularly in the Amazon (Betts, Sanderson & Woodward, 2008).

Our understanding of the scale of feedbacks from land use change is increasing, but still lacking, and it is important to better understand role of natural ecosystems and management practices in the carbon cycle (Potter *et al.* 2008; Bonan 2008; Betts *et al.* 2008b; Chapin *et al.* 2008; Dalal & Allen 2008; Heimann & Reichstein 2008). For example, peatlands (Limpens *et al.* 2008) are not explicitly included in global climate models and therefore predictions of future climate change may be underestimates. This emphasises the need to fully consider the role of biodiversity in mitigation policies.

4 Role of biodiversity in mitigation policies

Terrestrial ecosystems clearly play a major role in the carbon cycle through the net removal of carbon from the atmosphere. The role of the natural biosphere in climate change mitigation is recognised in the UNFCCC through Land Use Land Use Change and Forestry (LULUCF). Given the scale of biospheric carbon stores, losses and sequestration, and the potential to manage these processes, the inclusion of LULUCF in future international climate change agreements is of utmost importance (Schlamadinger *et al.* 2007; Cowie, Schneider & Montanarella 2007a; Henschel *et al.* 2008; Mollicone *et al.* 2007). The IPCC 4AR estimates that over the next century, 345-1269 $GtCO_2e$ could be abated through land-use based mitigation policies; 15-40% of total abatement requirements; due to a combination of carbon store management, and sequestration policies (Rokityanskiy *et al.* 2007). In addition, land-use based mitigation policies have the potential to deliver significant additional benefits for biodiversity.

4.1 Land use activities under the UNFCCC

Annex I Parties, under Article 3.3 of the Kyoto Protocol, can use *"direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990, measured as verifiable changes in carbon stocks,"* to meet emissions reductions targets. In addition, they can elect Forest Management, Grassland Management, Cropland Management, and Revegetation for inclusion in the accounting process (Benndorf *et al.* 2007; Schlamadinger *et al.* 2007). There are calls by some (Cowie, Kirschbaum & Ward 2007b; Mollicone *et al.* 2007) to include all lands and associated processes in the LULUCF, rather than the narrow activities specified above. Whilst the lack of an approach to properly account for this appears to be a barrier in this respect, there is a notable omission of peatlands and wetland (Henschel *et al.* 2008), particularly as Annex I countries have large extents of these areas.

The rules for LULUCF were only set after emission reduction targets had been agreed. This has been viewed as a limitation, as in effect land use activities ‘offset’ emissions in other sectors, rather than acting as an integral part of the mitigation portfolio (Benndorf *et al.* 2007). Issues still remain over the permanence of sequestration activities as management changes or natural disturbances can quickly release any carbon accumulated (Lal 2008).

The opportunities for Non Annex 1 countries to participate in such activities is also limited, and restricted to the Clean Development Mechanism (CDM); where Annex I countries can gain carbon credits through activities in developing countries. CDM activities are restricted to Afforestation, Reforestation and Deforestation activities, and can make up only 1% of the emissions reduction portfolio for Annex I countries (Dutschke 2007; Schlamadinger *et al.* 2007). A detailed discussion of the current structure of LULUCF and the potential for development in post-2012 agreements is beyond the scope of this paper, but can be found in the literature (Mollicone *et al.* 2007; Benndorf *et al.* 2007; Cowie *et al.* 2007a; Cowie *et al.* 2007b; Dutschke 2007; Rokityanskiy *et al.* 2007; Schlamadinger *et al.* 2007).

The Bali Action Plan, adopted by UNFCCC at the thirteenth session of its Conference of the Parties (COP-13) held in Bali in December 2007, mandates Parties to negotiate a post-2012 instrument, including possible financial incentives for forest-based climate change mitigation actions in developing countries (Decision 1/CP.13). The Parties specified that the development of such an instrument should take into consideration ‘the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.’ COP-

13 also adopted a decision on ‘Reducing emissions from deforestation in developing countries: approaches to stimulate action’ (Decision 2/CP.13). This decision recognises both that reducing emissions from deforestation and forest degradation in developing countries (REDD) can promote co-benefits and may complement the aims and objectives of other relevant international conventions and agreements, and that the needs of local and indigenous communities should be addressed when action is taken to reduce emissions from deforestation and forest degradation (Eliasch 2008).

It is generally agreed that a post-2012 LULUCF agreement should aim to reduce emissions from land use change (including REDD) and enhance carbon reservoirs, linked to adaptation strategies (Eliasch 2008; Gibbs & Herold 2007; Schlamadinger & Bird 2007). The following section will examine the potential for REDD and other LULUCF activities contribute to climate change mitigation.

4.2 Issues with including ecosystems in mitigation policy

Despite the role of biodiversity in the carbon cycle, land-use based mitigation policy has been constrained by a number of issues, both methodological and practical. One such issue is that of uncertainties over the exact role of ecosystems in the carbon cycle as detailed previously. Other methodological issues include the lack of accurate carbon accounting, difficulties in estimating emissions ‘saved’, and factoring out natural disturbances from anthropogenic activities (Schlamadinger *et al.* 2007; Cowie, Kirschbaum & Ward 2007b).

More specific concerns surrounding land-use based climate changed mitigation include the practical issues of permanence, leakage, and additionality (Eliasch 2008; Gibbs *et al.* 2007). ‘Permanence’ refers to the issue that carbon locked up in biomass and soils may be released at a later date, either following human disturbance, or natural disturbance such as drought, fire, or pests (Eliasch 2008). ‘Leakage’ occurs when emissions reduced in one area, for example through protection of one section of forest, are simply displaced to deforestation nearby (Benndorf *et al.* 2007); and ‘additionality’ refers to a situation in which the emissions reductions or carbon savings would have occurred anyway in the absence of mitigation policy.

These issues have been a particular concern in project-based activities such as those currently allowed under the CDM, but are less so when emissions are reported through national level accounting, as is likely for a REDD mechanism (Eliasch 2008); which is discussed in section 4.3.1.

4.3 Potential for mitigation through forest activities

Mitigation strategies in the forest sector fall under two main areas; the maintenance of stored carbon through reducing emissions from deforestation and degradation (REDD), and the sequestration of carbon from the atmosphere through afforestation, reforestation, and restoration (ARR). It is not clear how the development of the REDD mechanism will interact with LULUCF, but is possible that REDD will include afforestation and reforestation activities in an all encompassing mechanism (Eliasch 2008). However, for the purposes of this report, afforestation and reforestation are treated separately from REDD in accordance with the current structure under LULUCF.

The IPCC AR4 reported that a combination of forestry activities would have the potential to achieve 0.4GtC emissions reductions per year with a price of $20 per tonne, and 1.3-4.2GtCO_2/yr reductions in 2030 at costs up to $100 US per tonne CO_2. Mitigation through reduced deforestation was considered to have greater potential than that offered by afforestation (IPCC 2007), and is therefore the focus of this section.

More recent analyses have suggested that including forest in the cap and trade system would reduce emissions by 2.6 GtCO_2 per year by 2030 (Eliasch 2008). Including Afforestation, Reforestation and Restoration (ARR) in this scenario adds another 0.9 GtCO_2 per year of emissions savings; with a total potential for 3.5 GtCO_2 emissions savings by 2030.

4.3.1 Reduced emissions from deforestation and degradation (REDD)

In recognition of the importance of tropical forest in the global carbon cycle, and in provision of biodiversity and ecosystem services, proposals for the development of a REDD mechanism are being rapidly developed (Canadell & Raupach 2008; Olander *et al.* 2008). As emissions from deforestation are currently approximately 1.5 GtC per year, there appears to be high potential for cost-effective emissions reductions from REDD (Canadell & Raupach 2008). Currently, the UNFCCC has no mechanism for reducing deforestation in developing countries (Gullison *et al.* 2007).

The exact form of REDD is still to be determined, but is likely to involve national-level accounting whereby reductions in emissions from deforestation are measured relative to a baseline, determined according to the circumstances and historical emissions of the country (Eliasch 2008). A national-level approach would reduced risk of leakage. It is still unclear how Parties with low deforestation rates will be compensated, but there are various proposals considering how this might be achieved (Mollicone *et al.* 2007; Strassburg 2007; Strassburg *et al.* 2008; TCG 2008). Such proposals are detailed in the Eliasch Review (Eliasch 2008), which outlines a number of options including linking baselines to a global ‘business as usual’ emissions scenario in order to ensure that all forest stocks are incorporated. It is also unclear whether REDD will be financed through taxation, an international fund, or through the carbon market (Skutsch *et al.* 2007). The scale of emissions reduced, and particularly biodiversity benefits, will be determined by the design of the mechanism.

It is widely accepted that a successful mechanism for REDD will have to address the drivers of deforestation and will require effective targets, robust monitoring and measuring, appropriate financial mechanisms, and good governance (Eliasch 2008).

4.3.1.1 Mitigation potential

There remains significant deforestation pressure in the tropics. A recent study of the tropical humid biome has estimated that 27.2 Mha of forest, or 2.36% of the total stock, was cleared between 2000 and 2005 (Hansen *et al.* 2008), with deforestation ‘hotspots’ in Brazil and

Indonesia. It has further been estimated that current plans for infrastructure development in the Amazon will result in the release of approximately 32 GtC (Malhi *et al.* 2008), and 15-26 GtC in the next three decades in combination with fire, degradation and drought (Nepstad *et al.* 2008). However, it has also been suggested that effective enforcement of protected areas could avoid 17 GtC emissions by 2050 (Soares-Filho *et al.* 2006). Clearly, there is significant scope for reduction of deforestation, providing that the financial incentives are sufficient to cover the opportunity costs of land conversion (Nepstad *et al.* 2008). In addition to the scope for reducing deforestation, there appears to be considerable need. It has been estimated that the global economic cost of the climate change impacts of deforestation will rise to around $1 trillion a year by 2100 in the absence of mitigation (Hope & Castilla-Rubio 2008).

Indeed, it has been estimated that an additional 87-130 GtC will be released by 2100 in the absence of policy measures (Houghton 2005), whereas a 50% reduction in deforestation rates by 2050 (and maintained until 2100, with a cessation in deforestation when only 50% of the forest area remains) would avoid the direct release of up to 50 GtC, or nearly 12% of total required reductions for stabilisation at 450ppm (Gullison *et al.* 2007). Eliasch (2008) suggests that a *well designed* REDD mechanism could reduce deforestation rates by up to 75% in 2030; and in combination with ARR could make the forest sector carbon neutral. Absence of mitigation efforts through reducing emissions from the forest sector would increase atmospheric CO_2 levels by approximately 30ppm (Hope & Castilla-Rubio 2008). As the current levels stand at 433ppm, and the stabilisation target is at 445-490ppm, forests are critical for achieving reduction targets (Eliasch 2008).

There remains a scarcity of literature on the potential for reducing emissions from forest degradation specifically, although estimates of the scale of degradation suggest that the potential is high (section 3.4). A recent study has demonstrated that improved management of forest, e.g. through reduced impact logging, can reduce carbon emissions by approximately 30% (Putz *et al.* 2008). Therefore, improved practices in tropical forest designated for logging would retain at least 0.16 GtC per year, (particularly in Asia), or 10% of that obtainable through completely halting tropical deforestation (Putz *et al.* 2008). Recent evidence that many tree species with high C storage are preferred timber species (Kirby & Potvin 2007), suggests that species-level management will be important in reducing emissions through degradation.

The potential for REDD to contribute to emissions reductions through protecting carbon stores is clear, but there have been questions since the AR4 over the potential of old-growth forest to act as both a carbon store and a sink. Recent evidence suggests that old- growth and established forests can continue to accumulate carbon, contrary to the long-standing view that they are carbon neutral (Luyssaert *et al.* 2008; Desai *et al.* 2005). In addition, old-growth forests can accumulate carbon in soils (Zhou *et al.* 2006); suggesting that REDD will contribute to emissions reductions through carbon sequestration, in addition to maintenance of carbon stocks. Further, the carbon sink in old growth Amazonian forest is comparable to the emissions from deforestation (Phillips *et al.* 2008), and it has been estimated that the atmospheric CO_2 concentration would be 10% higher in the absence of the tropical forest sink (Betts *et al.* 2008a); although the potential future impact of climate change on this sink is uncertain (Heimann & Reichstein 2008).

4.3.1.1.1 Mitigation capacity in the face of climate change

Although it was noted in the IPCC 4AR that global change will impact upon carbon mitigation in the forest sector, the magnitude and direction of the change could not be predicted with confidence. This remains the case (Heimann & Reichstein 2008), although there is a growing body of literature in this area. Recently, it has been reported that 10 of 11 climate models project that tropical forests will continue to act as a net sink even in the face of climate change (Gullison *et al.* 2007), and evidence suggests that reduced deforestation and degradation can increase resilience of ecosystems to climate change impacts (Malhi *et al.* 2008; Nepstad *et al.* 2008; Betts 2007). There is therefore some evidence to support the claim that REDD could maintain the capacity of forests to resist climate change (below a certain threshold), and provide and assist with local adaptation to climate change, (Betts, Malhi & Roberts 2008).

4.3.1.1.2 REDD and tropical peatland

As highlighted previously, peatland is not eligible for inclusion under any of the current carbon accounting mechanisms within the LULUCF. Whilst this could potentially remain the case for boreal peatland, tropical peatland has the potential to be captured by REDD; particularly if emissions are measured as the difference between the carbon stock of the original forest and the altered land use (Eliasch 2008). 46% of deforestation in South East Asia occurs on peat (Hoojier *et al.* 2006), and accounts for substantial emissions of CO_2 to the atmosphere. However, to fully capture the carbon emissions from peatland it would be necessary to report carbon loss from soil below the current depth of 30cm specified by the IPCC (Miles & Kapos 2008).

4.3.1.1.3 Forest in Annex 1 countries

Although REDD has dominated recent forest discussions, it has been suggested that land use change and degradation in all areas (not just tropical) should be included in a future climate change agreement (Mollicone *et al.* 2007; hne *et al.* 2007; Eliasch 2008). Forests in temperate regions, particularly boreal forests, store large amounts of carbon; particularly in the soil (Nabuurs et al. 2008; Ciais et al. 2008), but current climate mitigation policies do not incentivise their conservation (Nabuurs *et al.* 2008).

4.3.1.1.4 Economic feasibility of REDD

Economic modeling has suggested that REDD will be a competitive, low-cost abatement option (Ebeling & Yasue 2008; Kindermann *et al.* 2008; Neeff 2008), as had been suggested in the IPCC AR4 (Fig 1.).

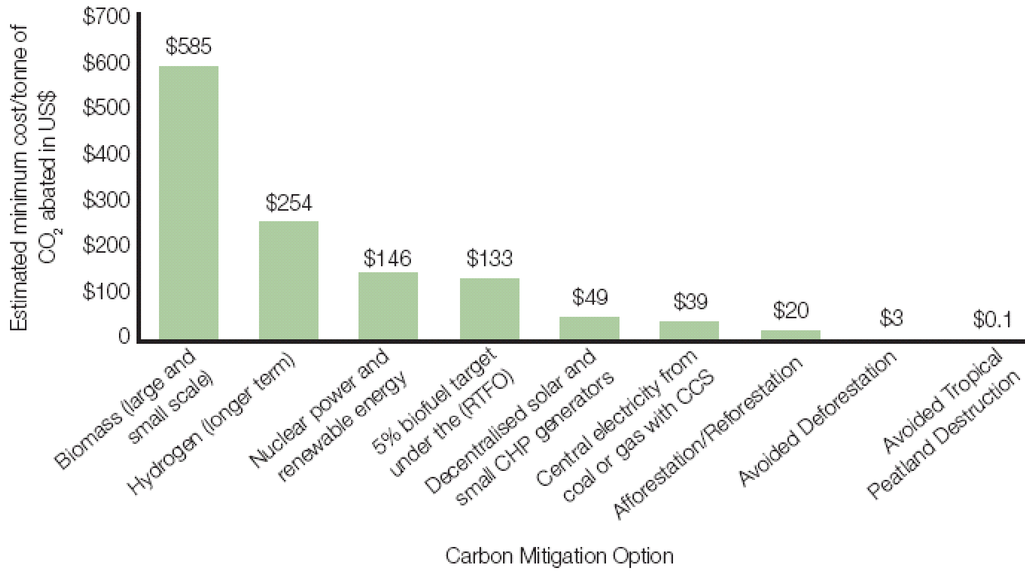

Figure 1. Cost comparison of carbon mitigation options (Spracklen *et al.* 2008). Original source IPCC 4AR

A 10% reduction in deforestation from 2005 to 2030 could provide 0.3-0.6 GtC in emission reductions annually and would require $0.4 billion to $1.7 billion (Kindermann *et al.* 2008). Mollicone *et al.* (2007) suggest that a 10% annual reduction in deforestation would reduce deforestation emissions 75% by 2020; and that a 50% reduction of total deforestation could provide 1.5-2.7 GtC in emission reductions and would require $17.2 billion to $28.0 billion.

Indeed, it has been suggested that an investment in reducing deforestation on the same scale as that put into the Renewable Transport Fuel Obligation in the UK would result in avoided emissions 50 times greater than those currently achieved (Spracklen *et al.* 2008). The Eliasch Review (2008) similarly concluded that the cost of halving global emissions could be reduced by 50% in 2030 through inclusion of the forest sector in a trading system. This would require finance of approximately $17-33 billion per year, of which $7 billion could be supplied by the carbon market, and $11-19 billion would need to come from other funding sources. Opportunity costs of forest conservation have risen since estimates by (Stern 2007) due to the rise in agricultural commodity prices, and now stand at $7 billion (Eliasch 2008).

These levels of finance can be put in context when considering the costs of *not* reducing emissions from deforestation. Modelling for the Eliasch review has suggested that the net benefits (including ecosystem services) of a 50% reduction in deforestation could amount to $3.7 trillion over the long term; rising to $6.3 trillion if 90% deforestation is reduced (Braat & ten Brink 2008).

Whilst these global figures highlight the potential for REDD, it is at the national level that such finance will need to be realized. Nepstad *et al.* (2008) suggest that a 30 year programme costing $8 billion (less than $2 per tonne C) could result in the cessation of deforestation in the Amazon within 10 years. A study in Panama, has estimated the total yearly cost of REDD at US$3.5 million (Potvin, Guay & Pedroni 2008). Although clearly ‘cost effective’, these studies emphasize the need for significant financial investment and capacity building. The Eliasch

Review (2008) estimates that such support for 40 forest nations could cost $4 billion over five years. A number of countries are already receiving support through the World Bank's Forest Carbon Partnership Facility and the UN REDD programme in a demonstration phase.

4.3.1.1.5 Methodological capabilities

The uncertainty in estimates of emissions from deforestation is largely due to data availability and methodological issues, and a significant body of research has gone into developing methods to resolve this (Gibbs *et al.* 2007; Herold & Johns 2007b; Olander *et al.* 2008; Ramankutty *et al.* 2007). Although the exact form of a REDD mechanism is yet to be determined, Parties will be required to monitor carbon emissions from deforestation and degradation which will require monitoring of forest area loss and proportion of biomass lost in degradation, in addition to knowledge of the biomass and the carbon content of each type of forest lost or degraded (Olander *et al.* 2008). Clearly, improvements in monitoring are required at a pan-tropical and national scale, and require commitments of capacity building and standardised protocols (Achard *et al.* 2007).

It is likely that reduced emissions will be measured against a baseline, probably established through historical rates. Establishment of baselines and monitoring of deforestation is likely to see a significant role for remote sensing data (Olander *et al.* 2008). Details of the available remote sensing options have been extensively reviewed (Olander *et al.* 2008; Herold & Johns 2007b), with significant progress being made.

These estimates of deforestation need to be combined with carbon stock estimates. Again, there is no perfect option for estimating carbon stock, but a range of options do exist, and it is generally agreed that technological constraints should not act as a barrier to the development of a REDD mechanism (Herold & Johns 2007a). It is likely that methodologies for assessing and monitoring carbon stocks will be based on the current IPCC good practice guidelines (Olander *et al.* 2008).

Although there are clearly many technical and political issues to be resolved before REDD could be put in place, there is growing consensus that such issues can be overcome. General scientific opinion appears to suggest that tools in development for assessing and monitoring carbon stocks are accurate and feasible for use in a REDD mechanism (Eliasch 2008).

4.3.1.2 Biodiversity impacts

A successful REDD mechanism has the potential to deliver significant benefits; contributing to biodiversity conservation at both the species and ecosystem level, whilst contributing to the maintenance of ecosystem services (Eliasch 2008). However, the design of REDD is still under discussion and will be the subject of negotiation (Skutsch *et al.* 2007). The different proposed versions of REDD are likely to have differing impacts on biodiversity (Strassburg 2007; Strassburg *et al.* 2008; TCG 2008); which will be influenced by the baselines adopted, and the financial mechanism employed (TCG 2008; Mollicone *et al.* 2007; Eliasch, 2008). For example, whether or not incentives for REDD are directly connected to forest area (regardless of deforestation rates) will impact upon tropical forest conservation (Mollicone *et al.* 2007; Strassburg 2007; TCG 2008). It has been suggested that REDD should include an explicit means of rewarding actions that reduce emissions from deforestation and degradation in ways that deliver benefits for biodiversity and ecosystems, e.g through premium credits for project providing co-benefits (Eliasch 2008).

Although the biodiversity impacts and benefits of REDD will depend upon the exact mechanism decided upon, REDD is likely to have net biodiversity benefits through tropical forest conservation at the scale detailed above; as habitat conversion is the major cause of biodiversity loss (Ravindranath 2007). The potential for generation of finance at levels of $1-10 billion is also at a scale not seen previously for forest conservation (Miles & Kapos 2008).

4.3.1.2.1 Biodiversity value of natural forest

Tropical forests have extremely high levels of biodiversity. The Amazon rainforest alone hosts about a quarter of the world's terrestrial species (Malhi *et al.* 2008). Deforestation continuing at projected rates in SE Asia could results in the loss of 79% SE Asian vertebrates by 2100, half of which are endemics (Brook, Sodhi & Bradshaw 2008), and mean total extinction rates of 20% and 33% of tree species in the Brazilian Amazon are projected under the optimistic and non-optimistic scenarios of habitat loss, respectively (Hubbell *et al.* 2008). Forest degradation and logging increases access to bushmeat, further threatening many tropical forest vertebrates (Brook *et al.* 2008), and habitat fragmentation reduces the adaptive capacity of species to climate change (Brook *et al.* 2008), and influences species distributions (Escalante *et al.* 2007).

4.3.1.2.2 Possible biodiversity impacts of REDD

Despite the obvious biodiversity benefits of conserving tropical forest, an international REDD mechanism under UNFCCC will be focused on carbon storage, and may not explicitly support biodiversity and other forest ecosystem services. There may be risks of cross-ecosystem leakage under REDD, whereby protection of forest leads to additional pressure to convert or degrade other ecosystem types (Miles & Kapos 2008). This could have negative effects on the biodiversity of these other ecosystems, and should be considered in conservation planning through, for example, focus of funds on non forest ecosystems and low-carbon forest (Miles & Kapos 2008).

Management practices such as suppression of fires may also impact biodiversity in the long term, as many forest processes rely on natural fire regimes (Berry *et al.* 2008). There is also no guarantee that representative forest types will be protected, with the representation of forests across environmental gradients beneficial for biodiversity conservation but not necessary for reducing carbon emissions (Berry *et al.* 2008).

One aspect that may have an impact on biodiversity conservation is the definition of forest and forest degradation. It is difficult to agree appropriate universal definitions and national definitions may be more applicable to the development of a REDD mechanism, but would have to be developed during the REDD preparation phase. Conversely, a definition clearly distinguishing between natural forest and plantations appears essential if afforestation and reforestation are to be included in the REDD mechanism. Under such a scenario, it is conceivable that deforestation could continue at present rates, provided the emissions were offset by the establishment of new plantations (Eliasch 2008). Afforestation does not always have positive biodiversity benefits, and can in fact have negative impacts when replacing natural ecosystems (section 4.3.2.2).

If REDD is to deliver benefits for biodiversity, it is important that feasibility studies and demonstration phases for REDD take into account the national pressures affecting biodiversity conservation, and assist in the development of tools to quantify and report methods for assessing and prioritising these benefits (Miles & Kapos, 2008).

With regard to the impacts of REDD on local and indigenous communities, there are risks as well as opportunities, and there are issues of governance and tenure to be resolved (Peskett *et al.* 2008). It has been suggested that involving local communities in REDD is essential if it is to provide both biodiversity and carbon benefits (Singh 2008) and appears to be an aspect that requires further research. However, with the scale of finance that could be made available, REDD has a significant opportunity to provide financial benefits to local communities as well as maintained ecosystem services (Eliasch 2008).

4.3.2 Afforestation, Reforestation and Restoration

4.3.2.1 Mitigation potential

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned (Canadell *et al.* 2007a). Whilst there is significant potential for increasing the capacity of the natural carbon sink, particularly in the tropics, it has been suggested that there is a need for more integrated study of how land management changes may affect climate change (Betts 2007; Chapin *et al.* 2008).

According to a range of cost estimates from $20 to $100, reforestation could sequester 0.16-1.1 GtC per year to 2100, with land requirements of up to 231 Mha (Canadell & Raupach 2008). Modelling for the Eliasch review has supported these figures, with an estimated mitigation potential of 0.9 GtC per year. A global analysis of land suitability for CDM-AR carbon 'sink' projects identified large amounts of land (749 Mha) as biophysically suitable and meeting the CDM-AR eligibility criteria, but much was on productive lands, grassland, or savanna. The implications of this would require consideration if the cap on CDM-AR were to be raised (Zorner *et al.* 2008).

Whilst in some regions afforestation has clearly had an impact (China has established 24Mha of plantations to transform the forestry sector from a source to a sink, offsetting 21% of their fossil fuel emissions), there is debate over the climate mitigation benefits provided by afforestation thus far (Canadell *et al.* 2007a). Indeed, evidence questions the mitigation benefits of afforestation and reforestation; suggesting that although such activities are cost effective, the relative contribution of plantations to emission reductions is relatively low (Strengers, Van Minnen & Eickhout 2008). Clearly, the previous land use goes a long way to determining the carbon benefits of afforestation. Expanding agroforests into areas currently under pasture could sequester significant amounts of carbon while providing biodiversity and livelihood benefits (Kirby & Potvin 2007; Jindal, Swallow & Kerr 2008), whereas expanding into natural grassland or wetland can have both negative impacts for both carbon and biodiversity (Berry *et al.* 2008).

The impact of afforestation on soil is an area identified by IPCC 4AR as requiring further research, and this appears to remain a priority. For example, a recent study in Africa has found that afforestation projects in savannah ecosystems had negative impacts on the carbon budget one year after plantation due to soil disturbances (Nouvellon *et al.* 2008), whereas afforestation of grassland has had a net positive impact in one region of China (Hu *et al.* 2008). Peatlands are used extensively for forestry in Canada and Scandinavia, in which carbon emissions from the draining of peat are likely to outweigh carbon sequestration (Parish *et al.* 2008). Effective forest C sequestration requires the management of all C pools, including traditionally managed pools such as bole wood and also harvest residues and soils (Gough *et al.* 2008). It is also important to consider local factors in reforestation policies (Clement & Amezaga 2008).

All evidence suggests that the greatest carbon benefits from afforestation and reforestation can be gained from the tropics (Bala *et al.* 2007). The climate mitigation benefits of reforestation in boreal regions are less certain when taking albedo and evaporation into account (Bonan 2008; Bala *et al.* 2007); which has led to the conclusion that the best strategy for forest carbon management in temperate regions is to discourage land use change and avoid large albedo changes (Bala *et al.* 2007; Canadell & Raupach 2008).

In addition, there are concerns over the response of plantation forest to climate change. It is thought that plantations have less natural resilience than natural forest to climatic perturbations, and climate induced changes in fire and insect outbreaks (Stephens *et al.* 2007), and it will be necessary to consider inter and intra species responses to climate change to optimize mitigation potential (O'Neill, Hamann & Wang 2008).

It should be emphasized that forests are valuable resources for many reasons unrelated to climate, but that this depends on the type of afforestation. Species selected for high carbon sequestration may have the greatest carbon mitigation benefits, but low biodiversity benefits. Mitigation strategies should not reduce the resilience of forest to climate change (Berry *et al.* 2008), and need to be planned with reference to potential future climatic conditions.

4.3.2.2 Biodiversity impacts

It is well publicised that sequestration schemes often require a tradeoff; production forest results in higher carbon benefits but fewer biodiversity benefits, whereas multifunctional forest can have biodiversity benefits but is of lower sequestration value (Garcia-Quijano *et al.* 2007b; Garcia-Quijano *et al.* 2007a). Policies aimed at providing carbon benefits through sequestration do not necessarily provide biodiversity benefits (Nelson *et al.* 2008). Plantations support lower levels of species diversity than natural ecosystems, and afforestation of natural ecosystems can have significant negative impacts on biodiversity (Berry *et al.* 2008; Cowie *et al.* 2007a).

In addition, evidence for negative impacts of afforestation/reforestation CDM projects on the hydrological cycle have been well publicised, with evidence of reduced water flow following afforestation schemes (Jackson *et al.* 2005). Such impacts have been noted as a result of large scale afforestation in China (McVicar *et al.* 2007), largely in previously unforested and water stressed areas. However, afforestation of agricultural land in the tropics can increase the water infiltration capacity of soil (Ilstedt *et al.* 2007), and such impacts are largely dependent on the previous natural land cover. Although the current limit on afforestation projects through the

CDM limits the scale of expansion, land use impacts and potential for conversion of natural ecosystems would need to be taken into account if afforestation efforts were to be increased (Trabucco *et al.* 2008; Jackson *et al.* 2005; Farley, Jobbagy & Jackson 2005) through, for example, the REDD mechanism.

Recent reviews of plantation forest and biodiversity have suggested that plantations can be beneficial, but only when planted on degraded land or on agricultural land where it can buffer edge effects and increase connectivity; whereas conversion of natural forest, and afforestation of non-natural forest land is detrimental (Brockerhoff *et al.* 2008; Berry *et al.* 2008). The conversion of grassland to plantation has been found to have negative impacts on ecosystem services in Ecuador (Farley 2007). Monoculture plantations are likely to have negative biodiversity impacts, and be less resilient to climate change, whereas promotion of heterergeneous plantations with native species and diverse gene pools reduces the biodiversity impact of plantations (Berry *et al.* 2008), and can help to stabilize the carbon storage against natural disturbances.

It has been suggested that plantation forestry can be the ‘lesser of two evils’ where land was earmarked for conversion (Brockerhoff *et al.* 2008). In addition, plantations can reduce degradation pressures on natural forest, but this requires landscape level land use planning. In particular, the impacts of monocultures and biodiversity and ecosystem services should be assessed at a site level (Brockerhoff *et al.* 2008). The use of fast growing genetically modified or non-native trees could also have significant implications for biodiversity, particularly where the species have the potential to be invasive.

The biodiversity benefits of reforestation should be higher than that of afforestation because it is on naturally forested land. However, research has suggested that species utilisation of regrowth forest is variable, and landscape scale management is required to maximise biodiversity benefits through restoration (Bowen *et al.* 2007).

4.3.3 Forest management

4.3.3.1 Potential

In addition to reducing deforestation and improving sequestration capacity of forest, there are carbon benefits to be gained from managing existing forests to increase sequestration capacity. Although Annex 1 countries had the opportunity to include forest management in their carbon accounting, it is generally agreed that the system is limited and does not optimise the potential for sustainable use of forest in climate change mitigation (Nabuurs *et al.* 2008), particularly with reference to removal of forest products and substitution effects (Bottcher *et al.* 2008). Clear-cut harvesting and fire disturbance results in a lasting decrease in annual forest C storage in temperate forest (Gough *et al.* 2008).

Given recent questions surrounding the actual mitigation potential of temperate forest plantations, due to albedo effects (Bala, 2007), there has been some agreement that reducing deforestation and sustainable forest management are the best options in these regions, and recent studies have identified ‘hotspots’ of European forest, where carbon storage and accumulation is high (Nabuurs *et al.* 2008; Ciais *et al.* 2008).

Improving forest management can significantly reduce carbon emissions (Putz *et al.* 2008), and most forest management strategies, such as control of fire by thinning and removal of undergrowth have mitigation benefits (Berry *et al.* 2008). However, capacity for reduced emissions through forest management requires consideration of natural disturbances. In Canada, for example, forest has turned from sink to source following large scale insect outbreaks, resulting in the decision not to elect forest management as an accounting option (Kurz *et al.* 2008).

4.3.3.2 Biodiversity impacts

Whilst improved forest management practices can have significant biodiversity benefits, forest management specifically for climate change mitigation can have some negative impacts. Removal of woody debris for biomass, and removal of undergrowth can have negative impacts on undergrowth-dwelling species and can alter ecosystem dynamics, as can control of fire regimes (Berry *et al.* 2008).

4.4 Potential for mitigation of other activities under the LULUCF

4.4.1 Improved cropland management

4.4.1.1 Mitigation potential

The IPCC 4AR estimated that agriculture accounted for 5.1- 6.1 $GtCO_2e$ per year in 2005, or 10-12% of global emissions, mostly through release of N_2O and CH_4, and is a global source of emissions. There is therefore significant potential for emissions reductions through agricultural management, although this is mostly through reductions in loss of soil organic carbon (SOC). Agriculture is likely to remain a net source (Canadell et al. 2007a); particularly where cropland replaces natural ecosystems.

An in depth review of the full breadth of mitigation strategies within agriculture is beyond the scope of this report, but has been produced for Europe (Berry *et al.* 2008). Such mitigation strategies include improvements in; livestock management, animal breeding and husbandry, grassland and grazing management, crop production, water management, reduced tillage, use of breed cultivars, use of N- fixing crops, and fertilizer management.

Enhanced carbon sequestration in soil is seen as the most important agricultural mitigation technique in Europe (Berry *et al.* 2008). Changing agricultural land use, in particular through agro-forestry schemes, is one strategy to achieve this; as is the use of no-till agriculture. Agroforestry involves the planting of trees intermingled with crops and increases both standing biomass and soil sequestration, and has a high mitigation potential in the tropics (Verchot, V *et al.* 2007). Employing no-till agriculture minimizes disturbance to soil carbon that can result in high levels of emissions to the atmosphere (Canadell *et al.* 2007a). Cultivated soils generally contain 50-75% less carbon than those in natural ecosystems (Lal 2008). Crop genetic diversity also aids the ecosystem to sequester carbon, and helps in preventing soil erosion (Hajjar, Jarvis & Gemmill-Herren 2008), with a higher SOC content than soils under monoculture (Lal 2008).

It has been suggested that, in Europe at least, reductions in agricultural emissions do not occur through climate policy, but through improved management practices, which can provide both carbon and biodiversity benefits (Lal, 2008). Agricultural management was specifically identified by the IPCC 4AR as a mitigation option with considerable potential co-benefits (IPCC 2007).

4.4.1.2 Biodiversity impacts

A review of the impacts of all potential agriculture mitigation techniques in Europe (Berry *et al.* 2008) has suggested that when 'worst practice' techniques are employed, no agricultural mitigation practice is beneficial in biodiversity terms. Those identified as detrimental to biodiversity, even under 'best practice' techniques included the improvement of species-rich grassland with specific varieties, the use of high sugar grasses, and displacement of food crops for biofuel.

Most 'best practice' techniques were considered to be either moderately or highly beneficial to biodiversity (Berry *et al.* 2008; Lal 2008), showing potential for considerable 'multiple benefits'. Although the IPCC suggests that agricultural improvements would in general be positive for biodiversity, it is acknowledged that this would require trade-offs. It is clear that the impacts will vary according to location, current biodiversity, and management techniques, making it difficult to apply top-down rules, and raising the question of what the impacts may be if agricultural management is dictated by climate policy (Berry et al, 2008). This issue was raised by the IPCC 4AR, which noted the potential for reduced productivity of cropland to displace land use change elsewhere, whereas increased productivity can 'spare' further land-use change.

In the tropics, it has been suggested that agroforestry can be beneficial for biodiversity, and might increase resilience of agriculture to climate change impacts (Bhagwat *et al.* 2008), (Verchot *et al.* 2007; Kirby and Potvin 2007). The literature reviewed suggests that agricultural mitigation strategies in particular can have considerable overlap with adaptation strategies.

4.4.2 Grassland management

4.4.2.1 Mitigation potential

Grasslands can sequester large amounts of carbon, primarily in the soil. Degradation of grasslands can therefore be a large source of carbon loss. A large body of literature in this area has focused on China, which has large areas of grassland with high stores of soil organic carbon (Yang *et al.* 2008). Degradation of these grasslands accounts for the biggest loss of carbon in China (Xie *et al.* 2007). Although the contribution to the carbon flux remains uncertain, land management practices have a large impact on uptake and release of CO_2 in grasslands (Cernusca *et al.* 2008).

Much of the literature focuses on management of grazing lands rather than unmanaged grasslands. Strategies for grassland management include enhancement of biomass production, the humification of biomass returned to the soil, facilitation of transfer of carbon deep into the subsoil by deep root system development, and the formation of organo-mineral complexes (Lal 2004). Good practice grassland management, including rotational grazing, nutrient management (Khan *et al.* 2007), and reduced burning, can increase soil carbon and reduce the loss of soil carbon through leaching (Lal 2004). Increased fungal biodiversity is also related to higher soil carbon storage in grasslands (Persiani *et al.* 2008). More recently, management of grasslands for species that are likely to increase NPP under conditions of increased CO_2 has been employed.

4.4.2.2 Biodiversity impacts

It has been suggested that improving degraded grassland would be a win-win for climate change and human development (Neely & Bunning 2008), as better grasslands for livestock would provide better food security. Improvement of degraded grassland with native species can have positive biodiversity impacts. However, grassland management can also have negative biodiversity impacts on plant, vertebrate and invertebrate species; particularly where diverse grasslands are replaced by a limited number of specific varieties and high sugar grasses (Berry *et al.* 2008). Introduction of nitrogen fixing species with the potential to become invasive can also have biodiversity impacts, as can increased use of fertiliser.

4.4.3 Revegetation

Re-vegetation is defined in the Kyoto Protocol as a direct human-induced activity to increase on-site carbon stocks through establishment of vegetation that does not meet the definitions of afforestation and reforestation (FCCC/CP/2001/13/Add. 1, page 58). Generally, the purpose is for erosion control on degraded lands (SCBD 2003). There is limited information available about the potential for revegetation to contribute to climate change mitigation, with the more recent studies quoting the figure reported in IPCC 4AR that vegetation regrowth and thickening in semi arid regions and savannahs accounts for 22-40% of the carbon sink in the US (Canadell *et al.* 2007a).

As revegetation tends to be on degraded land, the effects are generally positive but will vary according to the methods used, and whether native or exotic species are utilised (SCBD 2003).

4.4.4 Improved soil management

Improved soil management is not an activity explicitly specified under LULUCF, as it is included under all of the activities described above. However, the role in climate mitigation should be emphasised as it is often considered the area with the highest potential outside of forest activities. Global soil organic carbon has a sequestration potential 0.6-1.2 GtC (Lal *et al.* 2007) with high levels of carbon stocks, much of which is contained under natural ecosystems rather than managed ecosystems (Lal 2008). Emissions of approximately 78 $GtCO_2$ have been estimated from loss of soil carbon (Lal *et al.* 2007).

Degradation of soil also has biodiversity impacts through loss of biomass productivity and reduction in water quality. It has therefore been suggested that improvement in soil management under LULUCF is a 'win-win' strategy for biodiversity (Lal 2008).Wetland and peatland soils in particular are high in carbon and currently being heavily degraded.

4.4.5 Wetland and peatland – options for inclusion in the LULUCF

As reported in the IPCC AR4, wetlands account for approximately 37% of the terrestrial carbon pool, and therefore have a high potential to help mitigate climate change (IPCC, 2007). More recent estimates of a 550Gt carbon store in peatland alone appear to have raised this potential (Parish *et al.* 2008), as this accounts for over 25% of the currently estimated terrestrial carbon store. The susceptibility of peatland to climate and land use change has been emphasised through recent study (Ise *et al.* 2008). Natural peatlands are a vital component of the carbon cycle; emitting N_2O and CH_4 in addition to storing large amounts of carbon. They are currently acting as a carbon sink, but are likely to turn into a carbon source if current management strategies persist (Cagampan & Waddington 2008; Hoojier *et al.* 2006; Jaenicke *et al.* 2008; Neely & Bunning 2008; Parish *et al.* 2008; Uryu *et al.* 2008). Peatlands also support many specialized species and unique ecosystem types, and can provide a refuge for species that are expelled from non-peatland areas due to degradation and climate change (Parish *et al.* 2008).

Whilst estimates of carbon storage in peat soil are still uncertain, largely due to lack of information on peat depth and density, advances are being made in this respect. A new estimate of 5GtC stored in Indonesian peat utilises remote sensing technology supported by ground based observations (Jaenicke *et al.* 2008). The reduction in rate of current peat degradation in Indonesia therefore has the potential to reduce emissions significantly, particularly as deforestation on peat soils is accelerating (Uryu *et al.* 2008).

Tropical
As discussed previously, reduced deforestation and degradation in tropical peatland may capture some of the tropical peat emissions as deforestation largely occurs in peatland areas (Hoojier et al. 2006). The Indonesian peatlands have been identified as one of the largest stores of carbon in the terrestrial biosphere, but currently other climate change mitigation policies such as biofuel production are threatening these ecosystems (Jaenicke et al. 2008), and it has been suggested that they should be explicitly included in climate mitigation policies (Parish et al. 2008).

Temperate
Boreal regions have significant areas of peatland, acting as large carbon sinks (Nilsson *et al.* 2008). However, peat degradation is not limited to the tropics. Many peat bogs in Europe have been drained and are being restored (Glatzel *et al.* 2008), and over 55% of peatland area in Finland has been drained (Turunen 2008b) for agriculture and forestry accounting for approximately one third of greenhouse gas emissions (Turunen 2008a). Low carbon accumulation has been reported in peat bogs in Sweden as a result of high levels of nitrogen deposition, which alters the dominance of peat-forming vegetation (Gunnarsson *et al.* 2008).

Currently, there is very limited scope for inclusion of wetland or peatland in carbon accounting through the UNFCCC, and no direct mention in the text. The only option for inclusion in carbon accounting is where conversion of wetland areas is captured through management practices of other ecosystems, such as for forested peatland (Henschel *et al.* 2008). Recent increased estimates of the global peatland store have emphasised the need to manage peat, both in tropical and temperate regions, for resilience to increasing temperatures (Hoojier *et al.* 2006; Jaenicke *et al.* 2008; Parish *et al.* 2008; Uryu *et al.* 2008); particularly to maintain the water table (Ise *et al.* 2008).

As current emissions from peatlands are largely down to anthropogenic degradation, and degradation increases susceptibility to climate change, it has been suggested that conservation and restoration of peatlands can be cost effective mitigation measures (Parish *et al.* 2008; Spracklen *et al.* 2008). Restoration techniques generally involve raising the water table through water management and reintroduction of peat forming vegetation (Cagampan & Waddington 2008; Limpens *et al.* 2008), which can rapidly reduce carbon loss whilst also reducing the vulnerability of peat to climate induced lowering of water tables (Kechavarzi *et al.* 2007; Limpens *et al.* 2008). However, restoration can be expensive and does not necessarily restore the carbon dynamics to the previous state (Cagampan & Waddington 2008). Conservation of peat through reduction of drainage and fires are therefore the highest priorities (Parish et al, 2008). Modification of agricultural practices in peatland is also important, as is the management of natural forest in peatland areas (Hoojier *et al.* 2006; Parish *et al.* 2008).

Although there is growing literature in this area, the knowledge of the role of peatlands in the carbon cycle is still constrained and would appear to require further research.

4.5 Geo-engineering techniques

All of the mitigation policies discussed thus far have been biodiversity based. This is not strictly the case for geo-engineering techniques, but they are included here as they involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity.

The IPCC 4AR reports that little is known of effectiveness, costs, or side effects of geo-engineering techniques such as carbon capture and storage and iron fertilisation (IPCC 2007).

4.5.1 Carbon Capture and Storage

4.5.1.1 Mitigation potential

Carbon capture and storage is thought by many to be the best option for large scale reductions in emissions from fossil fuel consumption, and involves the capture, liquefaction, and injection of CO_2 into geological formations or the ocean (Berry *et al.* 2008; Lal 2008). Technology for this process is available, and research and development in this field is increasing (Figueroa *et al.* 2008). Geological CCS can take the form of injection into coal seams, oil wells, stable rock strata, or saline aquifers (Lal 2008); whereas oceanic CCS involves injection of CO_2 into the deep sea, or into the seafloor of shallow seas (Huesemann 2006; Yamada *et al.* 2008).

The rationale behind oceanic CCS is that although oceans have the capacity to store several thousand GtC (Lal 2008), CO_2 is transferred into the deep ocean only at rates of 2 GtC per year

(Huesemann 2008). Geological formations are also considered to have significant storage capacity, with the potential for 2,000 $GtCO_2$ reductions in emissions.

4.5.1.2 Biodiversity impacts

There has been more concern raised over the potential environmental impacts of ocean CCS than that of storage in geological structures, although there is the potential for leakage into aquifers and degradation of subsurface biodiversity, and issues of land use change (Berry *et al.* 2008; Koornneef, Faaij & Turkenburg 2008).

The risks of carbon capture and storage are not well known (Damen, Faaij & Turkenburg 2006; Shepherd, Iglesias-Rodriguez & Yool 2007a). Injection of CO_2 into the deepwater will alter ocean chemistry and could have significant consequences for marine organisms and ecosystems in the deep sea (Lal 2008; Thistle *et al.* 2007), with varying regional impacts (Watanabe *et al.* 2006). Deep-water fish have been shown experimentally to be more sensitive to environmental perturbations than shallow water species (Ishimatsu *et al.* 2006), and deep-sea injection into the seafloor could result in high rates of mortality for sediment dwelling organisms such as flagellates, amoebae, and nematodes (Barry *et al.* 2004; Fleeger *et al.* 2006). Potential impacts on bacteria have also been noted (Yamada *et al.* 2008). Leakage from carbon storage on the sea bed could also have significant impacts on communities in coastal and shelf seas (Widdicombe & Needham 2007). Increased acidification of oceans through leakage from volcanic vents has been shown to have large scale impacts on marine ecosystems (Hall-Spencer *et al.* 2008).

Conversely, if CCS has the potential to contribute to mitigation of climate change, it could have a positive overall impact for marine ecosystems through reduction of larger scale acidification impacts of global climate change (Magi 2008).

4.5.2 Ocean Iron fertilisation

4.5.2.1 Mitigation potential

The option to increase the sequestration capacity of the oceans through iron fertilisation is based on the premise that adding trace amounts of iron will lead to phytoplankton blooms, higher productivity, and therefore increased sequestration (Smetacek & Naqvi 2008). This is receiving increasing attention, particularly through the private sector (Leinen 2008), but requires more extensive fieldwork and modelling before the mitigation potential could be adequately assessed (Lampitt *et al.* 2008; Buesseler *et al.* 2008).

Current, the mitigation potential is uncertain (Buesseler & Boyd 2003; Gnanadesikan 2003), and it has been estimated that it would require fertilization of an area the size of the entire Southern Ocean to sequester 3% of current carbon emissions (Buesseler & Boyd 2003). It has been suggested that the high sequestration efficiency determined in some pilot studies should not be taken as an indication that iron fertilization will be efficient (Tollefson 2008). Indeed, as models have become more developed, the projected mitigation potential of iron fertilization has dropped, and the likelihood that fertilization will lead to the release of other GHGs such as N_2O has increased (Buesseler *et al.* 2008; Denman 2008; Jin *et al.* 2008; Krishnamurthy, Moore & Doney 2008; Law 2008; Upson 2008) .There is growing consensus that even the best case scenario for sustained fertilization would have only a minor impact in sequestration terms (Zahariev, Christian & Denman 2008). One study has found that approximately 75% of Fe added in fertilization experiments is lost very rapidly, and each large-scale fertilisation has a persistence of only one year (De Baar *et al.* 2008). There is also a lack of an appropriate regulatory regime to monitor ocean fertilization activities (Rayfuse, Lawrence & Gjerde 2008).

It has also been suggested that although fertilisation schemes enhance uptake of CO_2 by phytoplankton, they do not facilitate the sinking of organic carbon into the deep ocean that is required for effective sequestration (Shepherd *et al.* 2007a).

4.5.2.2 Biodiversity impacts

The potential environmental consequences of iron fertilisation are largely unknown but could be significant; partly because the scale would involve interference in natural productivity across immense expanses of ocean, which could have profound implications for marine ecosystems (Cullen & Boyd 2008; De Baar *et al.* 2008). Large scale eutrophication from disruptions to nutrient cycling could cause deep ocean anoxia, shifting microbial community structure (Huesemann 2008). In addition, there could be serious implications on the food web structure and dynamics (Shepherd, Iglesias-Rodriguez & Yool 2007b; Cullen & Boyd 2008). This could also have implications for fisheries (Parks 2008a; Parks 2008b). Some initial iron fertilization studies have resulted in a trophic shift in the phytoplankton assemblage, favouring large diatoms (Denman *et al.* 2006; Henjes *et al.* 2007). In addition, carbon sequestered in the ocean has the potential to mineralize and increase ocean acidification (Matsumoto 2006), which has profound impacts on marine ecosystems (Cao 2008). This has led the International Maritime Organization to conclude that ‘knowledge about the effectiveness and potential environmental impacts of ocean fertilization is currently insufficient to justify large-scale operations’ (Huesemann 2008).

Although most studies report negative impacts, it has been suggested that they have been based on worst case scenarios, and that iron fertilization could boost krill populations and therefore the food supply of marine mammals such as whales (Smetacek & Naqvi 2008).

The IPCC 4AR reported that the potential for ocean fertilisation was largely unknown, but was not promising. Information published since the 4AR supports this claim by suggesting limited mitigation potential and likely large scale impacts on oceanic food webs.

4.5.3 Nitrogen deposition

4.5.3.1 Mitigation potential

Increases in nitrogen deposition have been predicted to increase the size of terrestrial and marine carbon sinks (Karl & Letelier 2008), enhancing carbon uptake in forest ecosystems, with a lower impact on ocean sink strength. Combined, the land and ocean sinks may sequester an additional 10% of anthropogenic carbon emissions by 2030 owing to increased nitrogen inputs, but a more conservative estimate of 1 to 2% is more likely (Reay *et al.* 2008). Nitrogen fertilization of the ocean suffers from the same issues as that of iron fertilization, with limited potential for mitigation and high potential for biodiversity impacts (IPCC 2007). In addition, production of fertilizer requires fossil fuel use, further reducing emission reduction capacity (Glibert *et al.* 2008).

4.5.3.2 Biodiversity impacts

The biodiversity impacts of urea fertilisation are similar to those outlined for iron fertilisation. It is likely to change species composition, favouring those species that preferentially use urea as a nitrogen source, and could stimulate growth of toxic dinoflagellates, which could contribute to hypoxia (Glibert *et al.* 2008). Indeed, the impact of increasing N deposition on natural ecosystems is poorly understood (Dalal & Allen 2008), and the literature suggests that such impacts should be thoroughly assessed before employing these techniques for carbon management (Glibert *et al.* 2008).

5 Renewable energy

The renewable energy sector has been developed considerably in recent years in an effort to reduce reliance on fossil fuels. Although renewable energy projects are intended to reduce the impacts of climate change, they can also have impacts upon biodiversity (Paterson *et al.* 2008). Biofuels are included in this section as they are included in governmental renewable energy policies, although it is recognised that they are also a form of land management under the LULUCF.

5.1.1 Biofuel production

The production of liquid biofuels, namely ethanol and biodiesel, has been increasing rapidly in recent years (Gallagher 2008). The main crops used in ethanol production are maize and sugar cane, with rapeseed and oilpalm used to produce biodiesel. Biofuels currently make up less than 1% of the global energy supply (FAO 2008).

5.1.1.1 Mitigation potential

The production of liquid biofuels has been greatly incentivised over recent years as a tool for climate change mitigation and energy production. Biofuel plantations are intended to provide renewable energy for transport; reducing reliance on fossil fuels whilst sequestering carbon from the atmosphere. However, recent research has suggested that production of energy crops may do little to mitigate climate change; even increasing emissions by as much as 17-420x that of fossil fuels (Fargione *et al.* 2008; Righelato & Spracklen 2007; Searchinger *et al.* 2008). This is largely due to the direct conversion of carbon rich natural ecosystems, such as tropical forest, into biofuel plantations; or the indirect conversion through displacement of agricultural activity into such lands (Gallagher 2008). In general, the carbon that can be sequestered through restoring forests is thought to be greater than the emissions avoided through production of liquid biofuel

(Righelato & Spracklen 2007), especially when full life cycle analysis is undertaken (Scharlemann & Laurance 2008).

Clearly, the biofuel feedstock used and the location of plantations will determine the potential for climate change mitigation (Scharlemann & Laurance 2008). Gibbs *et al.* (2008) have shown that the replacement of carbon rich ecosystems by biofuel plantations results in carbon emissions over decades and centuries, whereas plantations on degraded land can have immediate carbon savings. However there is still considerable disagreement on definitions for ‘degraded land’ (RSC 2008) and recent research has suggested that the global bioenergy potential for such land is less than 8% (Campbell *et al.* 2008b) and 5% (Field, Campbell & Lobell 2008) of current energy demand globally.

The IPCC 4AR (IPCC 2007) identified second generation biofuels as one of the key future technologies for mitigation, where non-feedstock crops are used for energy production in combination with processing technologies. However, the potential of ‘second generation’ biofuel for climate change mitigation is also doubtful (Gallagher 2008), particularly if it involves the large areas of land use change projected (FAO 2008); and the technology is not yet available. The climate change mitigation potential of production of biofuel from microalgae remains to be seen, but there is some optimism (Jenner 2008; Wang *et al.* 2008).

5.1.1.2 Biodiversity impacts

Biofuel production has considerable impacts on biodiversity when it results in direct conversion of natural ecosystems (RSC 2008) and indirect displacement of agricultural land into natural ecosystems (Gallagher 2008). Biofuel production is largely driven by government targets and subsidies, and future production is expected to increase by 10-15% (Eickhout *et al.* 2008). Land availability depends in part upon future technological advances, but pessimistic scenarios predict a ‘land deficit’ of approximately 200 ha (Gallagher 2008), even when not taking into account the land requirements of other climate mitigation policies such as afforestation. A global land availability analysis has estimated that a land deficit of 215 million ha by 2030 is likely (Roberts & Nilsson 2007)

Biofuels can be produced with the greatest efficiency in the tropics, and the lack of economic incentives for the conservation of natural ecosystems leaves them vulnerable to replacement with biofuel crops (Doornbosch & Steenblik 2007). The expansion of oilpalm in Indonesia and Malaysia, which account for 86% of global oil palm production, is the most cited example of this. Although it is difficult to directly attribute forest loss to biofuel plantation, recent estimates have calculated that 55-59% of oil palm expansion in Malaysia, and 56% in Indonesia occurred at the expense of forest (Koh & Wilcove 2007b). Other studies have estimated that 27% of forest loss has occurred as a result of oilpalm plantations since 1982 (Uryu *et al.* 2008). Because Indonesia contains some three-quarters of southeast Asia's remaining primary forests, the continuing loss of its primary forests would be disastrous for the region's biodiversity (Koh & Wilcove 2007a). The scale of conversion of natural ecosystems to biofuel plantations is unknown due to the difficulty of accounting for displaced agricultural conversion (Gallagher 2008), such as that of soybean into the Amazon (Martinelli & Filoso 2008).

Issues of land conversion aside, the biodiversity impacts of biofuels are similar to those for plantation forest discussed previously. Plantation forest supports significantly lower levels of biodiversity than natural forest (Koh & Wilcove 2007b), and oil palm plantation support lower levels of biodiversity than other tree crops (Fitzherbert *et al.* 2008). Only 15% of species recorded in primary forest are also found in oil plantations, and forest fragments between biofuel plantations supported less than half the species of continuous forest (Fitzherbert *et al.* 2008). Although oilpalm is cited as one of the major threats to biodiversity, there is very little published research on this topic (Turner *et al.* 2008). Again, the biodiversity impacts of biofuel production will depend upon the previous land use and the crop used (RSC 2008; Scharlemann & Laurance 2008).

The potential biodiversity impacts of second generation biofuels are largely unknown, but recent reports have questioned their capacity to reduce biodiversity impacts as they do not produce beneficial co-products; and where all 'waste' biomass is removed for fuel this can impact on soil fertility (Eikhout et al 2008, Gallagher 2008). However, recent research has suggested that native prairie species grown on degraded land can produce 238% more energy than monocultures, whilst providing biodiversity benefits (Tilman, Hill & Lehman 2006); whilst woody biofuels on degraded land using native tree crops can provide ecosystem services (RSC 2008). However, monoculture non-food crops such as switchgrass have invasive traits and could have significant biodiversity impacts (RSC 2008).

5.1.2 Windfarms

The global rate in growth of wind power in 2005 was 24%, up 4% from 2004 (Kikuchi 2008), and if this trend continues, 120,000 MW is projected to be installed worldwide by 2010 (WWEA 2006). Germany is currently the highest user of wind energy (Huppop *et al.* 2006). In the UK the government target is to have 7 – 8% of its energy derived from wind power by 2010, and this would require the installation of 2000 turbines onshore and 1500 offshore (Drewitt & Langston 2006).

Wind farms must be sited on open, exposed areas with high average wind speeds to be effective, which means that proposed sites are often upland, coastal and offshore areas (Huppop *et al.* 2006). Birds can be affected by wind turbines though collision with turbine blades, displacement from migration routes, and direct habitat loss (Allison, Jedrey & Perkins 2008; Fielding, Whitfield & Mcleod 2006).

Mortality of birds as a result of wind turbines has been documented by a number of recent studies (Allison *et al.* 2008; Barclay, Baerwald & Gruver 2007; Barrios & Rodriguez 2004; Drewitt & Langston 2006; Everaert & Stienen 2007; Huppop *et al.* 2006; Kikuchi 2008). However, rates vary greatly between studies. Some of the highest levels of raptor mortality were recorded at Altamont pass in California (Orloff & Flannery 1992), where annual rates of 75 golden eagles and 400 griffon vultures for the wind farm were recorded, and in Navarre, Spain, wind farms killed 7150 birds in one year. Studies at other sites have not recorded such high rates (Fielding *et al.* 2006), and these high rates at Altamont and Navarre are thought to reflect site selection across known migration routes.

Fielding *et al.* (2006) suggest that site impact assessments at the planning stage could greatly reduce collision risk. In particular, wind farms should be located away from migration routes where possible. Altered bird behaviour to avoid wind turbines can produce a secondary form of habitat loss, with potential displacement away from migration routes, breeding grounds and feeding areas (Fox *et al.* 2006) and bird densities have been shown to decline rapidly with proximity to turbines (Fox *et al.* 2006; Larsen & Guillemette 2007). Bat fatalities from wind turbines have also been recorded (Arnett *et al.* 2008) and there is some evidence that turbine noise can modify anti-predator behaviour (Kikuchi 2008; Rabin, Coss & Owings 2006), although have no impact on density (Lucas, Forseth & Casper 2008). In addition, noise from offshore windfarms can impact upon marine mammals (Carstensen, Henriksen & Teilmann 2006; Madsen *et al.* 2006). Habitat loss through the development of the turbine sites is low, and is estimated at 2 – 5% of the total development area (Fox *et al.* 2006).

Recent evidence suggests that windfarm impact studies lack an evidence base (Stewart, Pullin & Coles 2007), and have minimal impacts on biodiversity (Devereux 2008). Although it is clear that environmental impact assessments should be conducted in land use planning, wind energy appears to have low impacts on biodiversity compared to other renewable energy options (Berry *et al.* 2008).

5.1.3 Nuclear power

Nuclear energy produces greenhouse gasses through mining, enrichment, reactor construction, waste disposal and transport; with emissions higher than for wind and hydro, and about the same as solar (Lenzen 2008).

Environmental impacts of nuclear power can be extremely high in the event of leakage of nuclear material. Chernobyl reduced species richness, abundance, and population density of wildlife (Clouvas *et al.* 2007; Moller & Mousseau 2007; Moller *et al.* 2007), with mutations from radiation spread amongst the wider population. In addition, it has been reported that forest are acting as ‘sink’ for radioactive isotopes from Chernobyl (Clouvas *et al.* 2007).

In the course of normal operations, there appears to be only minimal radioactive isotope release (Eyrolle *et al.* 2008; Gauthier-Lafaye *et al.* 2008; Jean-Baptiste *et al.* 2007; Virbickas & Virbickas 2005) although some changes in species diversity have been noted (Balciauskas 2005). However, uranium mine ponds can contaminate groundwater and soil, effecting ecosystems with either radioactivity or high levels of arsenic (Antunes, Pereira & Goncalves 2007; Carvalho & Oliveira 2007). As with other types of mining, degradation of habitat has negative impacts on biodiversity. Levels of radiation seen as safe for man found to be damaging for many other species (Fesenko *et al.* 2005).

Construction of reservoirs for water cooling can lead to changes in fish and bird diversity (Contador 2005), with fish mortaility from cooling water intake costing approximately 0.5 million euros per year (Greenwood 2008). Release of heated water also reduces algal species diversity (Kim, Choi & Nam 2008), alters fish species composition, and enhances water eutrophication (Contador 2005; Virbickas & Virbickas 2005).

5.1.4 Hydro power

Currently, hydro power provides about 20% of the world's electricity supply and more than 40% of the electricity used in developing countries (Bakis 2007). There are two main types of hydropower; large-scale (dam) hydropower, and small-scale ‘run-of-river’ projects. It is considered to be a sector with vast unexploited potential in developing countries. Hydropower does emit CO_2 through dam construction and algal build-up (Kaygusuz 2004; Ponseti & Lopez-Pujol 2006), and it has been suggested that dams in tropical areas actually cause more GHG emissions than savings (Fearnside 2005b).

The environmental issues involved with hydro-electric dams include habitat destruction, barriers to terrestrial migration, barriers to fish migration, reduced sedimentation and changes in flow altering downstream ecosystems, and fish mortality in turbines (Berry *et al.* 2008). Dams cause severe disruption to ecosystems through construction and flooding of large areas, which can completely alter the species composition of the area (New & Xie 2008). Large dams appear to be one of the most damaging renewable energy policies (Berry *et al.* 2008).

5.1.5 Solar power

Currently solar energy provides only 0.2% of the world’s energy, and production costs are still high and efficiency relatively low. However, with advances in technology, solar power is predicted to provide the world with large amounts of energy in the future (Fritsche, Hennenberg & Wiegmann 2008). Despite this, there is little literature available on environmental impacts, which include; risk of water pollution through leaks of heat transfer fluid and coolants, disposal of toxic material, land requirements, and thermal pollution (Huesemann 2006; Mohr, Schermer & Huijbregts 2007; Tsoutsos, Frantzeskaki & Gekas 2005). Although land use requirements are not large, large scale plants can impact natural ecosystems through competition for land-use on degraded or semi-natural lands (Berry *et al.* 2008).

5.1.6 Geothermal energy

Geothermal energy supplies 0.4% of the global energy supply, and has the potential to increase its share as a relatively ‘clean’ and resilient energy source (Berry *et al.* 2008). Potential impacts on biodiversity include land subsidence, chemical pollution of waterways, construction impacts, soil erosion, and noise disturbance, but it is in general considered to have low biodiversity impacts in comparison with other renewable energy sources (Berry *et al.* 2008; Thórhallsdóttir 2007).

5.1.7 Tidal energy

Tidal energy is considered to have potential as a renewable energy source. It consists of either movement of water through turbines, or tidal barrages. It therefore has the potential for a number of impacts on biodiversity through changes in flow, fish mortaility, changes in salinity, altered sediment deposition, underwater noise, impacts on migration corridors, and physical disturbances (Berry *et al.* 2008; Boehlert 2008; Prater 2006). These can have both short and long term implications for biological communities, which is likely to depend upon their resilience to disturbance (Gill 2005b; Cada *et al.* 2007). Modification of water circulation and currents is likely to have significant impacts on the benthic habitat (Boehlert 2008), and can change fish community structure. In addition, there are concerns that changed wave and noise patterns and physical structures will have an impact on cetaceans (Boehlert 2008), and birds (Clark 2006).

Due to potential impacts on coastal communities, tidal energy projects should involve substantial environmental assessments (Gill 2005a), and tidal barrages are likely to have the biggest impact (Berry *et al.* 2008). However, recent research for Scotland has suggested that net impacts for the environment could be positive (Allan *et al.* 2008).

6 Conclusion

Research since IPCC AR4 has served to strengthen the conclusion that biodiversity is an integral part of the carbon cycle, and important in mitigating climate change. A large amount of carbon is stored within ecosystems, although estimates are still uncertain and appear to underestimate the carbon stored in soils, particularly in peat. In addition ecosystems are continually sequestering carbon from the atmosphere; acting as a net sink for anthropogenic greenhouse gas emissions. It has been estimated that terrestrial ecosystems sequester 2.1-3 Gt of atmospheric carbon annually, approximately 30% of all anthropogenic CO_2 emissions. Marine ecosystems sequester large amounts of carbon through phytoplankton at the ocean surface.

Changes in land use, primarily through deforestation, are releasing significant amounts of the terrestrial carbon store to the atmosphere; accounting for 20% of greenhouse gas emissions. Carbon loss from soil could be comparable to that lost from biomass through deforestation; and emissions from peat could account for 10% of global emissions.

Recent evidence suggests that such damaging land use practices, in combination with climate change, could reduce the capacity of the carbon sink over century timescales; providing a positive feedback loop to the climate system. Our understanding of the scale of feedbacks from land use change is increasing, but still weak, and it is important to better understand role of natural ecosystems and management practices in the carbon cycle. Recent research has highlighted the damaging feedback loops between climate change and land degradation in peatlands and the Amazon rainforest. Although the scale of this feedback is still uncertain, research suggests that the inclusion of natural ecosystems in climate policy is vital if we are to achieve the target specified in the UNFCCC objective of limiting climate change to a 2ºC rise in global average temperatures.

Improved land use management practices can reduce the emissions from land use change and increase the sequestration capacity of the biosphere; with the capacity to make a significant contribution to climate change mitigation. The IPCC 4AR estimated that over the next century, 15-40% of total abatement requirements could be met through a combination of reduced loss of carbon stores, and sequestration policies. The use of ecosystem-based mitigation policies can also contribute to sustaining a variety of ecosystem services including biodiversity conservation.

All recent evidence suggests that there appears to be high potential for cost-effective emissions reductions from a mechanism for Reduced Emissions from Deforestation and Degradation

(REDD). The exact mechanisms for REDD have still to be decided, but there is general agreement that a well-designed mechanism could reduce deforestation rates significantly. A halving of deforestation rates could account for up to 12% of emissions reductions required by 2100, and economic modeling has suggested that REDD will be a competitive, low-cost abatement option. Moreover, a successful REDD mechanism has the potential to deliver significant additional benefits, contributing to biodiversity conservation at both the species and ecosystem level. In addition, there is the potential for REDD to reduce vulnerability of forest to climate change impacts, and maintain the capacity of the sink.

There is significant uncertainty attached to the level of carbon sequestration that can be achieved through afforestation and reforestation; and the potential for mitigation in this sector, particularly on decadal time scales, is often questioned. Whilst there is significant potential in increasing the capacity of the natural carbon sink, particularly in the tropics, there is a need for more integrated study of how land management changes may affect climate change and biodiversity. Improved agricultural management has significant potential to be positive for both climate change mitigation and biodiversity if best practice management techniques are employed.

The role of improved soil management in climate mitigation should be emphasised as it can be considered the area with the highest potential outside of forest activities. Currently, there is limited scope for inclusion of wetland or peatland in existing mechanisms, despite new evidence of their high carbon stores and contribution to global emissions. Evidence suggests that improved management of peatlands could substantially reduce emissions and reduce vulnerability to climate change impacts.

Geo-engineering techniques for mitigating climate change are not strictly ‘ecosystem-based’, but they do involve manipulation of the natural environment, particularly the marine environment, to increase the carbon storage and sequestration capacity; and could have substantial impacts on biodiversity. All evidence questions the capacity of ocean iron fertilisation, and highlights significant biodiversity impacts. Carbon capture and storage appears to have mitigation potential, but could have significant consequences for marine organisms and ecosystems in the deep sea. Renewable energy projects can also have impacts on biodiversity; particularly biofuel production and the construction of large dams.

It is clear from the literature reviewed that climate change mitigation policy has the potential to impact biodiversity both positively and negatively. Currently, many renewable energy projects are being planned without consideration for biodiversity impacts; as are some land-based mitigation strategies such as monoculture plantations. In particular, mitigation policies that reduce the capacity for adaptation to climate change should be avoided. However, due to the important role of ecosystems in the carbon cycle, it is clear that the potential exists to develop ‘win-‘win’ mitigation policies that are beneficial for both climate change mitigation and biodiversity; particularly through forest conservation, improved agricultural management, and land use planning.

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