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Technological and Economic Potential of Options to Enhance, Maintain, and Manage Biological Carbon Reservoirs and Geo-engineering

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EXECUTIVE SUMMARY

Terrestrial ecosystems offer significant potential to capture and hold carbon at modest social costs. The Intergovernmental Panel on Climate Change (IPCC) Second Assessment Report estimated that about 60 to 87GtC could be conserved or sequestered in forests by the year 2050 and another 23 to 44 GtC could be sequestered in agricultural soils. In this chapter, we describe and assess biological mitigation measures in terrestrial ecosystems, focusing on the physical mitigation potential, ecological and environmental constraints, economics, and social considerations. Also the so-called geo-engineering options are discussed.

The mitigation costs through forestry can be quite modest, US$0.1–US$20/tC in some tropical developing countries, and somewhat higher (US$20–US$100/tC) in developed countries. The costs of biological mitigation, therefore, are low compared to those of many other alternative measures. The costs would be expected to rise, however, if large areas of land were taken from alternative uses. The technologies for preserving existing terrestrial C and enhancing C pools, while using biomass in a sustainable way, already exist and can be further improved.

Increased carbon pools from management of terrestrial ecosystems can only partially offset fossil fuel emissions. Moreover, larger C stocks may pose a risk for higher carbon dioxide (CO_2) emissions in the future, if the C-conserving practices are discontinued. For example, abandoning fire control in forests or reverting to intensive tillage in agriculture may result in rapid loss of at least part of the C accumulated during previous years. However, using biomass as a fuel or wood to displace more energy-intensive materials in products can provide permanent carbon mitigation benefits. It is useful to evaluate terrestrial sequestration opportunities alongside emission reduction strategies as both approaches will likely be required to control atmospheric CO_2 levels.

Carbon reservoirs in most ecosystems eventually approach some maximum level. Thus, an ecosystem depleted of carbon by past events may have a high potential rate of carbon accumulation, while one with a large carbon pool tends to have a low rate of carbon sequestration. As ecosystems eventually approach their maximum carbon pool, the sink (i.e., the rate of change of the pool) will diminish. Although both the sequestration rate and pool of carbon may be relatively high at some stages, they cannot be maximized simultaneously. Thus, management strategies for an ecosystem may depend on whether the goal is to enhance short-term accumulation or to maintain the carbon reservoirs through time. The ecologically achievable balance between the two goals is constrained by disturbance history, site productivity, and target time frame. For example, options to maximize sequestration by 2010 may not maximize sequestration by 2020 or 2050; in some cases, maximizing sequestration by 2010 may lead to higher emissions in later years.

The effectiveness of C mitigation strategies, and the security of expanded C pools, will be affected by future global changes, but the impacts of these changes will vary by geographic region, ecosystem type, and local abilities to adapt. For example, increases in atmospheric CO_2, changes in climate, modified nutrient cycles, and altered disturbance regimes can each have negative or positive effects on C pools in terrestrial ecosystems.

In the past, land management has often resulted in reduced C pools, but in many regions like Western Europe, C pools have now stabilized and are recovering. In most countries in temperate and boreal regions forests are expanding, although current C pools are still smaller than those in pre-industrial or pre-historic times. While complete recovery of pre-historic C pools is unlikely, there is potential for substantial increases in carbon stocks. The Food and Agriculture Organization (FAO) and the UN Economic Commission for Europe (ECE)'s statistics suggest that the average net annual increment has exceeded timber fellings in managed boreal and temperate forests in the early 1990s. For example, C stocks in the live tree biomass has increased by 0.17billion tonnes (gigatonnes = Gt) C/yr in the USA and 0.11GtC/yr in Western Europe, absorbing about 10% of global fossil CO_2 emissions for that time period. Though these estimates do not include changes in litter and soils, they illustrate that land surfaces play a significant and changing role in the atmospheric carbon budget and, hence, provide potentially powerful opportunities for climate mitigation.

In some tropical countries, however, the average net loss of forest carbon stocks continues, though rates of deforestation may have declined slightly in the last decade. In agricultural lands, options are now available to recover partially the C lost during the conversion from forest or grasslands.

Land is a precious and limited resource used for many purposes in every country. The relationship of climate mitigation strategies with other land uses may be competitive, neutral, or symbiotic. An analysis of the literature suggests that C mitigation strategies can be pursued as one element of more comprehensive strategies aimed at sustainable development, where increasing C stocks is but one of many objectives. Often, measures can be adopted within forestry, agriculture, and other land uses to provide C mitigation and, at the same time, also advance other social, economic, and environmental goals. Carbon mitigation can provide additional value and income to land management and rural development. Local solutions and targets can be adapted to priorities of sustainable development at national, regional, and global levels.

A key to making C mitigation activities effective and sustainable is to balance C mitigation with other ecological and/or environmental, economic, and social goals of land use. Many biological mitigation strategies may be neutral or favourable for all three goals and become accepted as "no regrets" or "win–win" solutions. In other cases, compromises may be needed. Important potential environmental impacts include effects on biodiversity, effects on amount and quality of water resources (particularly where they are already scarce), and long-term impacts on ecosystem productivity. Cumulative environmental, economic, and social impacts could be assessed within individual projects and also from broader, national and international perspectives. An important issue is "leakage" – an expanded or conserved C pool in one area leading to increased emissions elsewhere. Social acceptance at the local, national, and global scale may also influence how effectively mitigation policies are implemented.

In tropical regions, there are large opportunities for C mitigation, though they cannot be considered in isolation from broader policies in forestry, agriculture, and other sectors. Additionally, options vary by social and economic conditions: in some regions, slowing or halting deforestation is the major mitigation opportunity; in others, where deforestation rates have declined to marginal levels, improved natural forest management practices and, afforestation and reforestation of degraded forests and wastelands are the most attractive opportunities.

Non-tropical countries also have opportunities to preserve existing C pools, enhance C pools, or use biomass to offset fossil fuel use. Examples of strategies include fire or insect control, forest conservation, establishing fast-growing stands, changing silvicultural practices, planting trees in urban areas, ameliorating waste management practices, managing agricultural lands to store more C in soils, improving management of grazing lands, and re-planting grasses or trees on cultivated lands.

Wood and other biological products play several important roles in carbon mitigation: they act as a carbon reservoir; they can replace construction materials that require more fossil fuel input; and they can be burned in place of fossil fuels for renewable energy. Wood products already contribute somewhat to climate mitigation, but if infrastructures and incentives can be developed, wood and agricultural products may become vital elements of a sustainable economy: they are among the few renewable resources available on a large scale.

A comprehensive analysis of carbon mitigation measures would consider:

- potential contributions to C pools over time;
- sustainability, security, resilience, permanence, and robustness of the C pool maintained or created;
- compatibility with other land-use objectives;
- leakage and additionality issues;
- economic costs;
- environmental impacts other than climate mitigation;
- social, cultural, and cross-cutting issues as well as issues of equity; and
- the system-wide effects on C flows in the energy and materials sector.

Activities undertaken for other reasons may enhance mitigation. An obvious example is reduced rates of tropical deforestation. Furthermore, because wealthy countries generally have a stable forest estate, it could be argued that economic development is associated with activities that build up forest carbon reservoirs in the long run.

Marine ecosystems may also offer possibilities for removing CO_2 from the atmosphere. The standing stock of C in the marine biosphere is very small, however, and efforts could focus not only on increasing biological C stocks, but also on using biospheric processes to remove C from the atmosphere and transport it to the deep ocean. Some initial experiments have been performed, but fundamental questions remain about the permanence and stability of C removals, and about possible unintended consequences of the large-scale manipulations required to have significant impact on the atmosphere. In addition, the economics of such approaches have not yet been determined.

Geo-engineering involves efforts to stabilize the climate system by directly managing the energy balance of the earth, thereby overcoming the enhanced greenhouse effect. Although there appear to be possibilities for engineering the terrestrial energy balance, human understanding of the system is still rudimentary. The likelihood of unanticipated consequences is large, and it may not even be possible to engineer the regional distribution of temperature, precipitation, etc. Geo-engineering raises scientific and technical questions as well as many ethical, legal, and equity issues. And yet, some basic inquiry does seem appropriate.

In practice, by the year 2010 mitigation in land use, land-use change, and forestry activities can lead to significant mitigation of CO_2 emissions. Many of these activities are compatible with, or complement, other objectives in managing land. The overall effects of altering marine ecosystems to act as carbon sinks or of applying geo-engineering technology in climate change mitigation remain unresolved and are not, therefore, ready for near-term application.

4.1 Introduction

Land is used to raise crops, graze animals, harvest timber and fuel, collect and store water, create the by-ways of travel and the foundations of commerce, mine minerals and materials, dispose of our wastes, recreate people's bodies and souls, house the monuments of history and culture, and provide habitat for humans and the other occupants of the earth. Can land, and water, also be managed to retain more carbon, and thereby mitigate the increasing concentration of atmospheric carbon dioxide (CO_2)? This chapter examines the present scientific thinking on this question.

The atmosphere now contains about 760 billion tonnes (gigatonnes = Gt) of carbon as CO_2, an amount that has increased by an average of 3.3 ± 0.2GtC each year throughout the 1990s, mostly from combustion of fossil fuels (IPCC, 2000a). Atmospheric C represents only a fraction (~ 30%) of the C in terrestrial ecosystems; vegetation contains nearly 500GtC, while soils contain another 2000 GtC in organic matter and detritus (Schimel, 1995; WBGU, 1998) as cited in Intergovernmental Panel on Climate Change (IPCC) Special Report on Land Use, Land-Use Change and Forestry (LULUCF) (IPCC, 2000a). *Table 4.1* provides estimates of the carbon stocks in terrestrial ecosystems now.

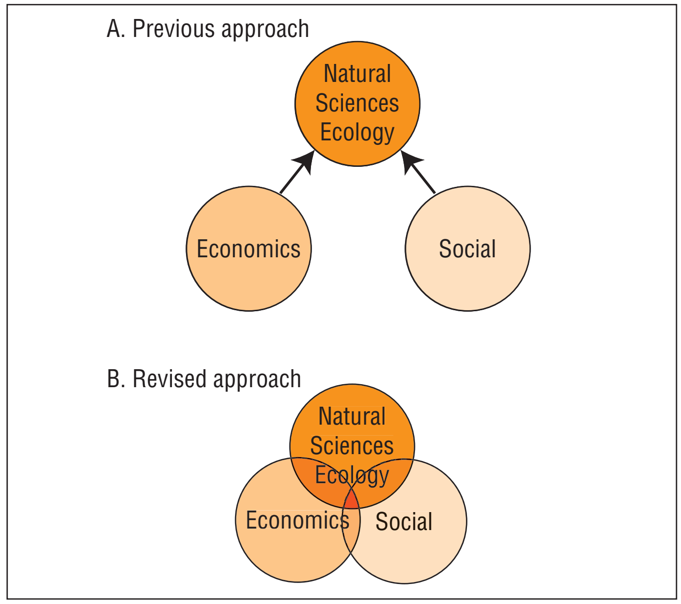

Figure 4.1: *Evolution of approaches to carbon sequestration in terrestrial ecosystems. Previous assessments (e.g., IPCC, 1996) tended to focus on ecological processes and potentials, and treated economic and social factors as constraints (A). A slightly different viewpoint considers the three dimensions as mutually reinforcing and seeks to maximize the overlaps (B).*

The Second Assessment Report (SAR) of the IPCC (1996) suggested that 700Mha of forestland might be available for carbon conservation globally – 138Mha for slowed tropical deforestation, 217Mha for regeneration of tropical forests, and 345Mha for plantations and agroforestry. The IPCC suggested that by 2050 this area could provide a cumulative mitigation impact of 60 to 87GtC, of which 45 to 72GtC in the tropics. Towards the end of this time interval, the mitigation impact could approach a maximum rate of 2.2GtC/yr. The cost of mitigation (excluding land and other transaction costs) was envisioned to be about 2 to 8US$/tC. The SAR (IPCC, 1996) further suggested that, over the next 50 years, an additional 0.4 and 0.8GtC could be sequestered per year in agricultural soils, with the adoption of appropriate management practices.

The current report, while supporting many of these earlier findings, provides a broader evaluation of the potential for management of C stocks in the biosphere (*Figure 4.1*). Recent studies, for example, suggest that costs may often be higher than estimated earlier, particularly when opportunity costs of the

Table 4.1: *Estimates of global carbon stocks in vegetation and soils to 1 m depth* (from Bolin *et al*., 2000; based on WGBU, 1998).

Biome	**Area (million km^2)**	**Carbon stocks (GtC)** Vegetation	Soils	Total
Tropical forests	17.6	212	216	428
Temperate forests	10.4	59	100	159
Boreal forests	13.7	88	471	559
Tropical savannas	22.5	66	264	330
Temperate grasslands	12.5	9	295	304
Deserts and semideserts	45.5	8	191	199
Tundra	9.5	6	121	127
Wetlands	3.5	15	225	240
Croplands	16.0	3	128	131
Total	**151.2**	**466**	**2,011**	**2,477**

land are included. In addition, the issue of "leakage" (where actions at one site influence actions elsewhere, a problem not considered by the SAR) is examined. This report considers forests, grasslands, croplands, and wetlands, and, where possible, examines all C pools within them. Carbon mitigation is evaluated as one of many services provided by ecosystems. The objectives of this chapter are to review progress made since the IPCC-SAR, and to evaluate prospects for storing more carbon in ways that ensure the continued provision of other goods and services from the varied and finite land resources.

The aim of this chapter is not to assess specifically the implications of the Kyoto Protocol (UNFCCC, 1997), a mandate assigned to the IPCC Special Report on LULUCF (IPCC, 2000a). Rather, it seeks to provide a broader scientific view of the prospects and problems of land management for carbon sequestration, unconstrained by the limited scope of the Kyoto Protocol.

This chapter begins by describing the current state of land use, the history of land use, ongoing changes in land use, pressures driving these changes, and potential competition among demands for land (Section 4.3). It then considers opportunities for enhanced C stocks, especially in forestry and agriculture (Section 4.4). Having identified possible C conservation measures, the physical, environmental, social, and economic impacts of these measures are examined; and assessment is made of how they augment or compete with other services provided by land (Sections 4.5 - 4.7). How these options might be evaluated and, where appropriate, encouraged (Sections 4.8 and 4.9) is also considered. Finally, the prospects for managing marine ecosystems to increase carbon sequestration, and the possibility of managing the global ecosystem by 'geo-engineering' of the earth's energy balance (Section 4.10) are considered.

Land-use changes and the pressures that influence them vary widely, especially between tropical and non-tropical regions. Both of these regions are addressed.

4.2 Land Use, Land-Use Change, and Carbon Cycling in Terrestrial Ecosystems

Terrestrial ecosystems provide an active mechanism (photosynthesis) for biological removal of CO_2 from the atmosphere. They act as reservoirs of photosynthetically-fixed C by storing it in various forms in plant tissues, in dead organic material, and in soils. Terrestrial ecosystems also provide a flow of harvestable products that not only contain carbon but also compete in the market place with fossil fuels, and with other materials for construction (such as cement), and for other purposes (such as plastics) that also have implications for the global carbon cycle.

Human activities have changed terrestrial carbon pools. The largest changes occurred with the conversion of natural ecosystems to arable lands. Such disruptions typically result in a large reduction of vegetation biomass and a loss of about 30% of the C in the surface 1 metre of soil (Davidson and Ackermann, 1993; Anderson, 1995; Houghton, 1995a; Kolchugina *et al*., 1995). Globally, conversion to arable agriculture has resulted in soil C losses of about 50GtC (Harrison *et al*., 1993; Scharpenseel and Becker-Heidmann, 1994; Houghton, 1995a; Cole *et al*., 1996; Paustian *et al*., 2000), and total emissions of C from land use change, including that from biomass loss, have amounted to about 122 ± 40GtC (Houghton, 1995b; Schimel, 1995). Most of the soil C losses occur within a few years or decades of conversion, so that in temperate zones, where there is little expansion of agricultural lands now, losses of C have largely abated (Cole *et al*., 1993; Anderson, 1995; Janzen *et al*., 1998; Larionova *et al*., 1998). Tropical areas, however, remain an important source of CO_2 because of widespread clearing of new lands and reduced duration of "fallow" periods in shifting agriculture systems (Paustian *et al*., 1997b; Scholes and van Breemen, 1997; Woomer *et al*., 1997; Mosier, 1998).

The competition for land varies among countries and within a country. Land-use and forestry policies for C management may be most successful when climate mitigation is considered alongside other needs for land, including agriculture, forestry, agroforestry, biodiversity, soil and water conservation, and recreation. Forest fires, for example, are controlled, in many parts of the world, not as a measure for carbon mitigation, but simply because fire threatens areas of human settlement and the habitats of living organisms.

Similarly, biodiversity and landscape considerations have motivated protection of old-growth stands in temperate, boreal, and tropical rain forests from commercial logging. In many cases such decisions have prevented C release into the atmosphere, even though C mitigation was not the initial intent (Harmon *et al*., 1990). The impact of harvest restrictions on C pool in old-growth forests may be affected by "leakage". If one ecosystem is protected but timber demand remains constant, logging may simply be shifted to another, similar ecosystem elsewhere, perhaps to a country where conservation priorities are lower.

4.2.1 Historical Land-Use Change in the Tropics

4.2.1.1 Trends in Land Use and Changes in Carbon Stocks

Tropical forests were largely intact until colonial times, when large tracts were removed to provide raw materials for railroads, ships, etc., in the period following the industrial revolution. The loss of tropical forests escalated in the second half of the 20th century. According to the UN Food and Agriculture Organization (FAO, 1996), about 15.4 million ha of natural tropical forests are lost each year. Of this, 42% occurs in Latin America, 31% in Africa, and 27% in Asia. Brunner *et al*. (1998) estimated tropical deforestation at 19.1 million ha/yr during the period 1990 to 1995. There has, how-

Box 4.1. Stocks and Flows

The global carbon cycle consists of the various stocks of carbon in the earth system and the flows of carbon between these stocks. It is discussed at length in IPCC WG I (Prentice *et al*., 2001) and IPCC Special Report on LULUCF (IPCC, 2000a) and is illustrated in *Figure 4.2*.

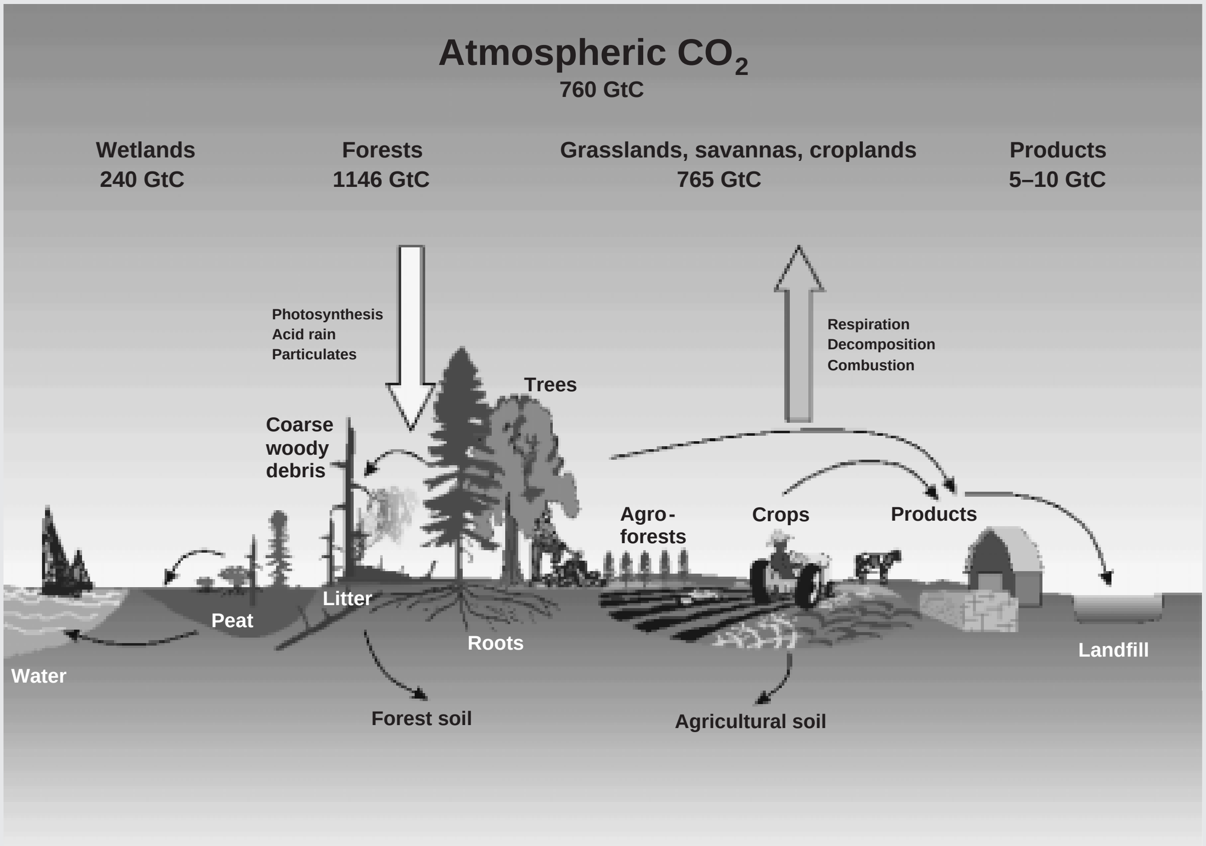

Figure 4.2: *Different ecosystems, their components, and human activities. The carbon stocks associated with the different ecosystems are stored in aboveground and belowground biomass, detrital material (dead organic matter), and soils. Carbon is withdrawn from the atmosphere through photosynthesis (vertical down arrow), and returned by oxidation processes that include plant respiration, decomposition, and combustion (vertical up arrow). Carbon is also transferred within ecosystems and to other locations (horizontal arrows). Both natural processes and human activities affect carbon flows. Mitigation activities directed at one ecosystem component generally have additional effects influencing carbon accumulation in, or loss from, other components. Estimates of ecosystem and atmospheric C stocks are adapted from Bolin et al. (2000). Values for C stocks in some ecosystems are still very uncertain. Not shown are estimates of C stocks in tundra (127GtC), deserts and semi-deserts (199GtC), and oceans (approx. 39,000GtC) (numbers are taken from Special Report on LULUCF, Fig 1-1, page 30; IPCC, 2000a).*

A consequence of the conservation of mass is that the net of all of the flows (measured as a rate variable in units such as tC/yr) into and out of a given reservoir or stock (measured in units such as tC) during a period of time must equal the change in the stock (tC) in that period. Conversely, a change in stock of a reservoir during a given period must exactly equal the integrated net difference in C flows into and out of that reservoir during that period. Elsewhere in this text the word “pool” is sometimes used to represent the various reservoirs of carbon in the global carbon cycle. The word “sink” is used to indicate the net positive flow of carbon into a terrestrial carbon pool.

The maximum rate of net ecosystem carbon uptake cannot occur at the same time as the maximum ecosystem carbon stock (see *Figure 4.3*). An ecosystem depleted of carbon by past events may have much higher rates of carbon accumulation than a comparable one in which carbon stocks have been maintained. Ecosystems eventually approach some maximum carbon stock – a carrying capacity – at which time the flows into the carbon pool are balanced by flows out of the carbon pool. Because C sink and C stock in ecosystems cannot be maximized simultaneously, mitigation activities aimed at enhancing the sink and maintaining the biological carbon stock coincide only partially (IGBP, 1998). (*continued*)

Box 4.1. continued

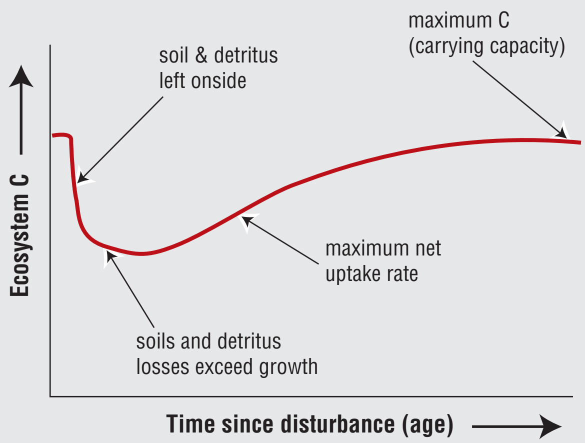

Figure 4.3: *An example of net changes in ecosystem carbon stocks over time. Changes in individual ecosystem components take place at different rates, but it is the net of the changes in all interconnected pools that determines the net flow to or from the atmosphere. In the example, the accumulation of biomass initially is at a lower rate than the decomposition of the dead organic matter stock so the stock of ecosystem C declines. Later in the cycle, dead organic matter stocks may increase, although other components have reached a steady state. Maximum ecosystem stocks (highest value of ecosystem C) occur at a later time than the maximum rate of net carbon uptake (steepest slope of the ecosystem C line).*

Similarly, the maximum rate of C substitution cannot occur at the same time as maximum C conservation. High rates of carbon substitution, through use of forest products or biofuels, generally require high productivity and efficient manufacture and use of derived products.

Carbon taken up by the biosphere may also accumulate in offsite pools – as products or in landfills – but it continues to oxidize at rates that depend on the conditions of those pools. It is the net of many flows that defines the changes in carbon stocks of off-site pools as well as of on-site pools. Carbon accumulation in off-site pools is an often overlooked, but a potentially important, form of sequestration.

ever, been a large increase in area devoted to forest plantations. By 1990, there were 61.3 million ha under plantations and the rate of establishment is now about 3.2 million ha/yr (FAO, 1996).

As pointed out by the IPCC (IPCC, 1996) global estimates of C emissions from deforestation have remained highly uncertain and show high geographical variability. The magnitude of forest regeneration (particularly secondary forest regrowth and regrowth of abandoned lands) and forest degradation processes is not well documented. Improving the accuracy of these estimates remains an urgent and challenging task (Houghton *et al*., 2000).

Estimates of C emissions from land-use change and forestry activities in the tropics during the1990s range from 1.1 to 1.7GtC/yr, with a best estimate of 1.6GtC/yr (Brown *et al*., 1996b; Melillo *et al*., 1993; Bolin *et al*., 2000). These estimates may change with improved information on biomass densities and land-use conversion. Detailed studies for major tropical countries in the early 1990s, studies that include forest regeneration and afforestation, show lower net emissions for most countries than those from aggregate estimates (Makundi *et al*., 1998).

A review of scenarios of future land-use changes in the tropics, and their implications for greenhouse gas (GHG) emissions, shows a wide range of estimates, particularly for the first part of the 21st century, where estimates differ by a factor of 14 (Alcamo and Swart, 1998). These disparities reflect a lack of agreement on the definition of deforestation, and a lack of knowledge and agreement on the estimation of C emissions (Alcamo and Swart, 1998). These scenarios can be divided into two groups: in one group emissions decline smoothly after 1990; in the other group emissions increase for a few decades after 1990.

4.2.1.2 Driving Forces for Land-use Change

The rates and causes of land-use change vary by region and scale (Kaimowitz and Angelsen, 1998). Deforestation is often considered a one way process, but the landscape is a dynamic mosaic of land uses and vegetation types, with transitions both to and away from forest (Houghton *et al*., 2000). Natural factors, such as forest fires and pests, as well as socio-economic processes, many of which are not seen at the local level, interact in complex ways, complicating analysis. Understanding the causes of this mosaic of land-use and/or land-cover transitions in order to understand and predict the net effect on deforestation rates and C emissions remains a key research challenge.

Conversion of forests to pasture and cropland has been the most important proximal cause of tropical deforestation. Non-sustainable logging has been the leading factor in parts of Southeast Asia, whereas excessive harvest of wood fuel has been important only in specific sub-country regions and in some African countries (Kaimowitz and Angelsen, 1998). According to Bawa and Dayanandan (1997), the causes (correlates) of deforestation are many and varied, with complex interactions. Overall, Bawa and Dayanandan found that population density, cattle density, and external debt were the key factors. In Africa, the most important factors were extraction of fuelwood and charcoal and demand for cropland; in Asia, it was cropland; and in Latin America, it was cattle density.

Most analyses of land-use change and forestry have concentrated on proximal reasons for land-use and/or land-cover change; that is, on land uses such as agriculture, pasture, and timber extraction that replace forests. But Meyer and Turner (1992) have identified six "underlying" forces: (1) population, (2) level of affluence, (3) technology, (4) political economy, (5) political structure, and 6) attitudes and values. The influence of each varies by region and country.

The rate of population growth is now apparently declining, but the population, and hence the demand for food and other land services, is still growing (Roberts, 1999). Population growth has been widely cited as a major cause of deforestation (Myers, 1989), but the relationship between population and deforestation is not simple. Population growth exerts increasing pressure on resources, but whether these pressures lead to forest degradation or to positive changes (e.g., afforestation, improved forest management, and better technology) depends largely on social structure. Extensive migration may also lead to deforestation and soil erosion. Simplistic assumptions about population and deforestation also do not apply where high population densities and/or growth rates are accompanied by forest conservation and reforestation programmes. In India, for example, deforestation rates have declined since 1980, despite population growth, owing to effective forest conservation legislation (Ravindranath and Hall, 1994).

Patterns that affect land-use are changed by economic development. Affluence usually increases consumption, but it does not necessarily decrease terrestrial C stocks. The maintenance of ecosystems tends to improve with increasing and better distribution of wealth, as well as with proper institutional structures and sound development strategies. The demand for and interest in forests and their services is the driving force for the technological and economic capacity to maintain forests. Also, wealthy societies tend to be urbanized and this may reduce destructive pressures on forests. Technological development provides efficient tools for land-use change and for high-value, alternative uses. Technology can also limit encroachment. As seen by the "green revolution" in agriculture, technological development can increase productivity on intensively managed land, thereby releasing other land areas from agriculture (Waggoner, 1994). Nevertheless, there is always the risk of leakage (*i.e*., tendencies to transfer destructive operations from the developed to less developed areas and countries), or the possibility that technology development and transfer will have positive spillover effects (Brown *et al*., 2000; Noble *et al*., 2000)

In many countries, especially those seeking development of frontier areas, subsidies are provided for activities promoting economic development. Land clearing may be subsidized directly or by providing property rights to cleared land. Frontier development is often considered desirable for security or where there is a disputed area.

Land-use change is driven largely by efforts perceived as "best and highest" use of the land. But benefits of the land that are non-market and/or external to the direct user (e.g., watershed protection, biodiversity, and carbon mitigation) may be ignored by land managers. For example, the decision to convert forestland to agriculture may ignore the many external and non-market benefits lost. Moreover, where long-term land rights are insecure, lands may be used to generate short-term benefits, with disregard for long-term benefits.

Factors related to social structure and political economy have not been studied widely, but studies at the country and regional levels suggest that deforestation is favoured by the following factors: growing landlessness and persistent inequalities in access to land, insecure land tenure, land speculation, rising external debt, large-scale expansion in commercial agriculture, erosion of traditional systems of resource management and community control, and widespread migration of impoverished people to ecologically fragile areas (Hecht, 1985; Palo and Uusivuori, 1999; Tole, 1998).

4.2.2 Land Use in the Temperate and Boreal Zones

4.2.2.1 Historical and Present Land Use in the Temperate and Boreal Zones

The temperate zone is the most populated zone of the world, while the boreal zone is quite sparsely populated. For thousands of years forest area has diminished, particularly in the temperate zone, as forests were cleared for agriculture and pasture. Clearing of the European Mediterranean region began *ca* 5000 years ago; in Central Europe and in China deforestation occurred in early Medieval times; in parts of Russia and Mongolia forest clearing occurred in late Medieval times; and in North America clearing occurred mainly in the 19^{th} century (Mather, 1990, see *Figure 4.4*). Since the mid 20^{th} century the net forest area of the temperate zone has no longer decreased but has instead increased (Kauppi *et al*., 1992). The inner parts of the boreal zone in Siberia, Alaska, and Canada have not been subject to significant land-use management. The opportunities present to store carbon in terrestrial ecosystems in the boreal and temperate zones are thus very much determined by historical land-use change and the associated losses of carbon (Kurz and Apps, 1999).

Understanding the historic and current net sink of C in the temperate and boreal zones is important to assessing the potential of present and future management options. In general, estimates of C flows have been based on a variety of methods and data, resulting in a wide range of reported values for C flows per region. The confidence level in each separate value is therefore low. For example, for European forests the estimates of the present C sink vary from almost 0 to 0.5 GtC/yr (Nabuurs *et al*., 1997; Martin *et al*., 1998; Valentini *et al*., 2000; Schulze, 2000). For Canada, early estimates, based on a static assessment, indicated a net sink of 0.08GtC/yr for the mid-1970s (Kurz *et al*., 1992); whereas subsequent analyses, accounting for changes in forest disturbances over time (see section 4.2.3), indicated that Canadian forests became a small net source of C (–0.068GtC/yr) by the early 1990s (Kurz and Apps, 1999). Estimates of carbon accumulation in woody biomass for the USA also show a large uncertainty. While the average rate for the USA C sink ranges from 0.020 to 0.098GtC/yr for the 1980s and 1990s (Birdsey and Heath, 1995; Turner *et al*., 1995; Houghton *et al*., 1999), atmospheric inversion models applied to the North American continent suggest a sink of 1.7 ± 0.5GtC/yr, largely south of 51ºN (Fan *et al*., 1998), but with very low levels of confidence (Bolin *et al*., 2000).

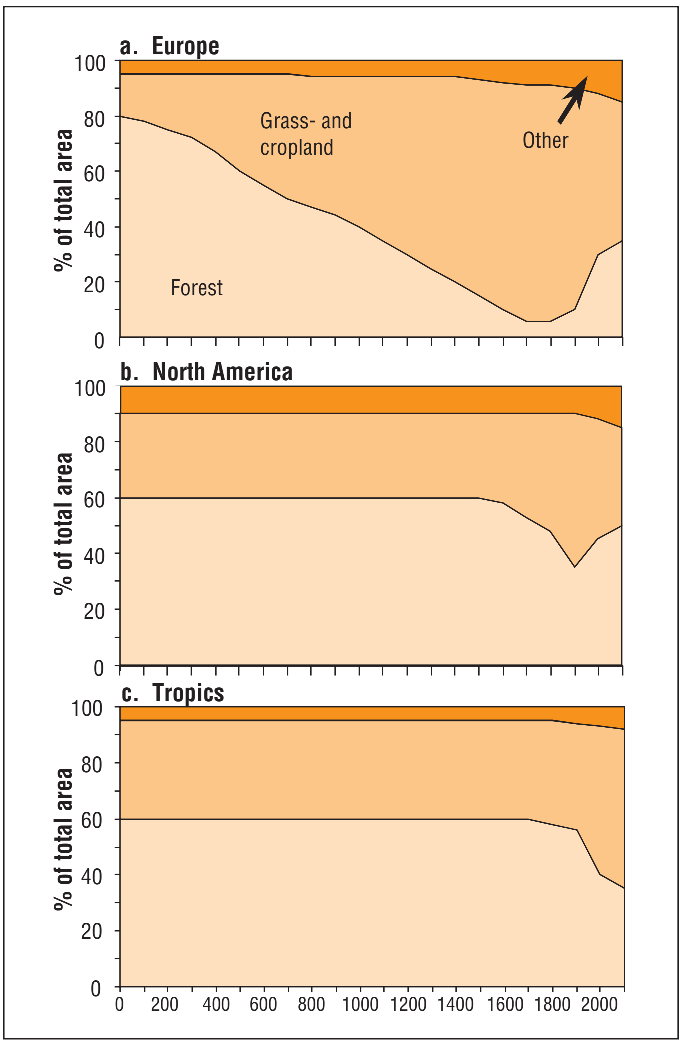

Figure 4.4: *Indicative figure displaying historical changes in land use in three world regions. The presented values should not be taken as absolute, because the historical evidence is often only anecdotal (Mather, 1990; Kauppi et al., 1992; Palo and Uusivuori, 1999; Farrell et al., 2000).*

In the less intensively managed forests of Russia and Canada, changes in mortality associated with natural disturbances appear to dominate over management influences (see Section 4.2.4). In European Russia, managed forest ecosystems were estimated to be a sink of 0.051GtC/yr between 1983 and 1992, but the less actively managed Siberian forest was a net source of 0.081–0.12GtC/yr (Shepashenko *et al*., 1998). The available estimates for Siberia differ even more than for the other regions mentioned above, and their confidence level may be “low” (Schulze *et al*., 1999).

Recent FAO statistics on 55 countries in the temperate and boreal zones indicate a general increase in the forest carbon stock (trees only) of 0.88GtC/yr (UN-ECE/FAO, 2000). Changes in forest management and changes in the environment have contributed to this trend. In Europe, the trend is consistent with the observation of increased growth in individual stands noted by Spiecker *et al*. (1996). The FAO statistics indicate that between the 1980s and 1990s both net annual increment and timber fellings increased, but that the rate of change was lower for fellings than for growth, resulting in a substantial increase in the carbon sink from the 1980s to the 1990s (Kuusela, 1994; Kauppi *et al*., 1992; Sedjo, 1992; Dixon *et al*., 1994; UN-ECE/FAO, 2000). The carbon sink in live woody vegetation was on the order of 10% of the fossil fuel CO_2 emissions in the USA and in western Europe, and higher in the 1990s than in the 1980s (c.f. Kauppi *et al*., 1992).

These relatively high sequestration rates are not a result of active policies aimed at climate mitigation, but less rather appear to be related to general trends in land use and land-use change. In the USA, Schimel *et al*. (2000) and Houghton *et al*. (1999) estimate that the observed sink is a result mainly of changes in land use and land management, rather than a response to changes in the environment. The latest observations, based on forest inventory data (UN-ECE/FAO, 2000), are reflected in the Special Report on LULUCF (IPCC, 2000a). The IPCC (2000a) estimates that the total global terrestrial biopheric sink in the 1990s amounted to 0.7GtC/yr, despite a source from land-use change in the tropics of 0.9GtC/yr.

4.2.2.2 Driving Forces for Land-Use Change

Land management decisions are influenced by many factors. In the temperate zone, and in the European parts of the boreal zone, these are mainly technological and economic. Agricultural production is, for example, heavily influenced by evolving technologies, economic opportunities, subsidies, and restrictions on international trade. Forestry practices are similarly influenced by economic returns, trade, and pressures from society (Clawson, 1979; Waggoner, 1994; Wernick *et al*., 1998). It is within these pressures and opportunities that carbon mitigation possibilities may be found, and preferably they would be region specific. *Table 4.2* gives an overview of some of the specific issues of importance in the temperate and boreal zone of the world.

Competition for land between forestry and agriculture has become less severe. Forest area is increasing in many regions of the boreal and temperate zone, partly because agricultural yields have improved or because the profitability of marginal agriculture has declined. The ability to produce agricultural goods has grown faster than demand, resulting in a downwards trend in prices (Alig *et al*., 1990; Waggoner, 1994). Much abandoned agricultural land has reverted to forest, either naturally or through deliberate planting. Superimposed over these land conversions is a transition in forestry from a foraging and gathering operation, dependent upon primary forest, through a stage of more intensively managed forest, to total forest ecosystem management. The latter occurs when urbanized societies press for nature-oriented forest management. Continuously improving technologies allow low-cost establishment and higher productivity from planted and plantation forests (Sedjo, 1983; 1999a). In agriculture, also, practices are changing towards maintaining site fertility or decreasing the risk of erosion.

Silvicultural practices have increased forest growth in many boreal and temperate regions. The increasing concentration of atmospheric CO_2 may also have contributed to the enhanced growth of forests.

Incentives for planting forests are provided by a combination of market factors and public policy. Remaining wild forests, such as the public forests in the US National Forest System and in British Columbia, are becoming less accessible and have increased harvesting restrictions. Subsidies to harvesting of natural forests are also being withdrawn elsewhere. For example, large subsidies for harvesting Russian forests were prevalent during the Soviet era, largely through subsidized transportation, but have now disappeared. The economic structures are in transition and industrial production has declined. As a result, harvests have fallen dramatically in Russia since the 1990s (Nilsson and Shvidenko 1998).

Market forces, reflecting industrial needs for wood, have provided financial incentives for expansion of commercial forests (Sedjo and Lyon, 1990). This is a trend expected to continue, because of growing demand for industrial wood and low profitability in agriculture (Sohngen *et al*., 1999). Early analyses suggested that economic returns from plantations (in the tropics as well as in the temperate and boreal zone) justify investment in a number of regions (Sedjo, 1983). Recent studies confirm that forest plantations are being established at a rate of 600,000ha/yr (Pandey, 1992; Postel and Heise, 1988; UN-ECE/FAO, 2000). However, industrial plantation forestry is new in many tropical areas and yields vary considerably across ecosystems. In many locations where plantations have only recently been established, little is known about the potential capabilities for increasing productivity as well as the potential problems that may limit yields.

4.2.3 Forest Disturbance Regimes

The concept of "forest disturbance" refers to events such as forest fire, harvesting, wind-throw, insect and disease outbreak (epidemics), and forest flooding that cause large pulses of CO_2 to be released into the atmosphere through combustion or decomposition of resulting dead organic matter. Stand-replacing disturbances, such as crown fires and wind-throw, are associated with the sudden death of large cohorts of trees near one another (Pickett and White, 1985; Kurz *et al*., 1995a, 1995b; Kurz and Apps, 1999, see *Box 4.2*). Some disturbance agents, such as pollution and some insects and disease outbreaks, may result in large areas with productivity decline but only local mortality (Hall and Moody, 1994). Disturbances play an important natural part in the lifecycle and succession dynamics of many forest systems. In boreal systems large-scale, natural, stochastic forces tend to dominate the ecosystem dynamics, even when direct human influences are considered (Kurz *et al*., 1995b). The return interval of these disturbances, their intensity, and their specific impacts are referred to as the disturbance regime (Weber and Flanigan, 1997). Kurz *et al*. (1995b) and Price *et al*. (1998) (having compiled insect, fire, and harvest data) showed that the disturbance regime of Canadian forests changed over the last quarter of the 20th century from about 2.5Mha/yr prior to 1970 to 4Mha/yr between 1970 and 1990. Using these data, Kurz and Apps (1999) showed that these changes in the disturbance regime resulted in a switch of Canadian forests from being a net sink of C to a small net source of C to the atmosphere.

Disturbances, both human-induced and natural, are major driving forces that determine the transition of forest stands, landscapes, and regions from carbon sink to source and back. The current pattern of forest vegetation and its role in carbon cycling reflects the combined effects of anthropogenic and natural disturbances over a range of time scales. For C stocks with very slow turnover rates (such as soils and peat) the effects of past disturbances on carbon cycling may reverberate for centuries and millennia (*Figure 4.5*). For example, carbon continues to accumulate in young soils (such as those associated with the isostatic uplift following deglaciation in Canada and Finland), which appear to be actively accruing carbon (Harden

Table 4.2: *Overview of biological carbon mitigation issues and opportunities in selected countries/regions*
(Based, in part, on Sedjo and Lyon, 1990; Fujimori, 1997; Nilsson and Shvidenko, 1998; De Camino *et al*., 1999; Sohngen *et al*. 1999; Zhang, 1996)

Region	Issues	Options to store carbon arising from the issues
USA/Canada	• Primary forest based forestry and second rotation forestry • High tech forest industry • Fierce environmental debates • Large impacts of natural disturbances • Agriculture under pressure (excess agricultural land)	• Fire management • Afforestation • Efficient use of wood products • Bioenergy • Farming practices (e.g., reduced tillage) that restore soil C
Europe	• Agriculture under pressure, afforestation of agricultural lands • Changing ownership • Forest health problems • Move towards nature-oriented forest management • High tech forest industry • In eastern Europe, privatization of forest ownership	• Nature-oriented forest management • Nature reserves • Afforestation • Efficient use of wood products • Bioenergy • Farming practices (e.g., reduced tillage) that restore soil C
Russia	• Transition to market economy • Bad financial situation of forest service • Large impacts of natural disturbances • Low levels of fellings	• Natural regeneration on abandoned agricultural land • Fire management • Capacity building • Farming practices that restore soil C
Japan	• Plantation-based forestry and managed secondary forestry • High tech forest industry • Forest health problems • Move towards nature-oriented management	• Efficient use of wood • Nature-oriented forests • Reserves • Bioenergy
China	• Transition to market economy • Transition from non-wood fibre sources to using wood fibre • Floods resulting from loss of forest	• Afforestation with plantations • Protecting primary forests • Flood protection • Farming practices (e.g., reduced tillage) that restore soil C
Australia/ New Zealand	• Plantation-based forestry and some primary forest based forestry • High tech forest industry • Afforestation of agricultural lands	• Fire management • Afforestation with plantations • Efficient use of wood products • Bioenergy • Halting deforestation • Farming practices (e.g., more forages) that enhance soil C
Argentina, Chile, Brazil	• Plantation-based forestry and some primary forest based forestry • High tech forest industry developing • Plantations are not able to reduce deforestation because they provide different set of products and services.	• Afforestation with plantations • Efficient use of wood products • Bioenergy • Halting deforestation • Farming practices (e.g., reduced tillage) that enhance soil C
Mexico	• Forestry largely based on native forests • Large deforestation rates • Economic incentives favour agriculture/cattle over forestry • Afforestation of degraded lands mostly for restoration	• Halting deforestation • Sustainable forest management of native forests • Social forestry • Afforestation with local species • Bioenergy

Box 4.2. Disturbance, Age-class Distribution, and their Implications for Forest Carbon Dynamics

At the stand scale, disturbance events (both natural and anthropogenic) have three main impacts on the carbon budget (Apps and Kurz, 1993). First, they redistribute the existing carbon by transferring carbon from living material, above and below ground, to the dead organic matter pools. Second, they transfer some of the carbon out of the ecosystem (e.g., into the atmosphere as combustion products, in the case of fire, and/or into the forest product sector as raw feedstock, in the case of harvest). Third, by opening the forest canopy, the disturbance changes the site micro-environment and restarts the successional cycle for new stand development.

At the scale of forests (typically comprising many stands), the disturbance regime determines the age-class structure (e.g., the even-age structure associated with stand-replacing disturbance regimes or the uneven-age structures associated with individual tree mortality and gap-phase replacement), and age-class structure of stands and trees making up the forest. The C stocks in a forest landscape, and the changes in these stocks over time, are strongly influenced by the age-class distribution (Kurz *et al*., 1995b; Turner *et al*., 1995; MacLaren, 1996; Apps *et al*., 2000; Bhatti *et al*., 2001). In managed plantation forests, the age-class distribution is controlled by the management regime and harvest cycle (Heath and Birdsey, 1993; MacLaren, 1996), while in natural forests other mortality agents play a major role. See Heath *et al*. (1996) and Kurz *et al*. (1995a) for examples.

Box 4.3. The Reduced Impact Logging Project, Carbon Sequestration Through Reduced Impact Logging

The RIL (reduced impact logging) project developed by Innoprise, a Malaysian company with forestry activities, and the New England Power Company, USA, aims to save CO_2 already stored in forest biomass by reducing damage to vegetation and soils during harvesting. The hope is to reduce damage by 50% compared to that of conventional harvesting. The techniques employed are modifications of conventional bulldozer harvesting techniques; including pre-felling climber cutting, directional felling, skid trail design, and post harvest operations such as rehabilitation of log landings. Today the total project area amounts to 2,400 ha. The pilot research project has quantified the carbon implications and costs on 1,415ha. They found that avoided emissions amounted to 65–90MgC/ha and that the associated costs were US$3.55/MgC (Wan Razali and Tay, 2000).

et al., 1992). In these soils, losses from decomposition of accumulated organic matter are exceeded by the inputs of fresh organic debris (Liski *et al*., 1999). Human influences on the disturbance regime include both direct effects, such as harvesting or inducing and/or suppressing natural disturbances (fires, insects, flooding, etc.), and indirect influences from altering the forest environment. Indirect influences include both climate change and atmospheric pollution, and their effects on tree health and survival.

The different types of disturbances are often linked. For example, in some forest types the probability of fire may increase following insect outbreaks because of increases in available fuel (litter). In some cases salvage logging (recovering the usable timber following a disturbance) can reduce the total area of living forest that is disturbed in a given year by all agents combined. It is common to try to replace natural disturbances (such as wildfires) with commercial harvesting, using a combination of protection and scheduled logging. In Sweden and Finland, for example, logging has become the main disturbance type; and large-scale natural disturbances resulting from wildfire, insect outbreaks, or storms have been almost nonexistent for half a century (Lähde *et al*., 1999).

Disturbances affect the carbon stocks of all components of forested ecosystems. During and following a disturbance, carbon is transferred from living material, above and below ground, to the dead organic matter pools (*Figure 4.2*). In the

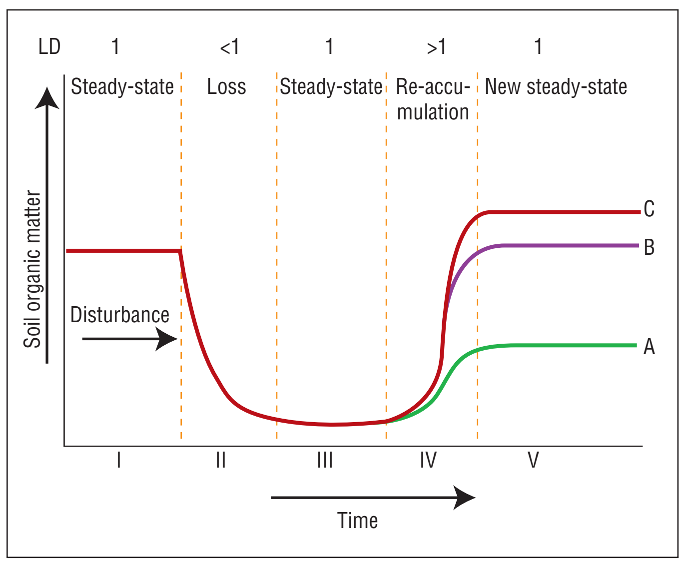

Figure 4.5: *Conceptual model of soil organic matter decomposition and accumulation following disturbance (after Johnson, 1995; IPCC, 2000a). At steady state (I), carbon (C) inputs from litter (L) equal C losses via decomposition (D) (i.e., L/D = 1). After a disturbance, D often exceeds L resulting in loss of C (II), until a new, lower steady state is reached (III). Adoption of new management, where L exceeds D results in a re-accumulation of C (IV) until a new, higher steady state is reached (V). The eventual steady state (A, B, or C) depends on the new management adopted.*

case of a forest fire, part of the ecosystem carbon is released immediately into the atmosphere as combustion products. Disturbed forest stands continue to release carbon into the atmosphere as the enlarged pools of dead organic matter tend towards a new steady-state condition (Bhatti *et al.,* 2001). Regrowth follows, but maximum uptake may not be achieved for some time (decades or more), and during much of this period decomposition of dead organic matter may exceed vegetative uptake. The corresponding re-sequestration of carbon through regrowth can last 50 to 200 years or more.

Management of natural disturbance regimes can provide significant C mitigation opportunities, e.g., through activities to prevent or suppress disturbances. Such measures can significantly enhance the strength of C sinks (Kurz *et al*., 1995a; Apps *et al*., 2000; Bhatti *et al*., 2001) and maintain existing C stocks, but only as long as the programmes are maintained. Other factors being equal, during periods of reduced disturbance (e.g., with increasing suppression effort), C stocks tend to increase as biomass accumulates and litter production (in all forms) increases: forests act as a sink for atmospheric C (Bhatti *et al*., 2001). In contrast, with increasing disturbance (e.g., with reduction in suppression effort), the net losses of C from forest ecosystems can exceed inputs from photosynthesis (*Figures 4.2* and *4.3*) and the forests could become a net source of C. We note that all forms of disturbance, not just highly visible fires, play a role in these dynamics. In a changing climate, the control of new pathogens and immigrant herbivores (especially insects and disease), to which local forest ecosystems may be maladapted, may be critical to avoid emissions and maintain existing forest C stocks.

Disturbances affect the carbon stocks in vegetation, in soil, and in dead organic matter. All these stocks vary over time as a function of the history of disturbances (MacLaren, 1996; Bhatti *et al*., 2001; Kurz and Apps, 1999). With an increase of widespread disturbance events the carbon stocks of living vegetation decrease and the age-class distribution of the forest shifts to younger stands containing less carbon. If forests are disturbed at regular intervals (i.e., an unchanging, disturbance regime), the carbon stock of large tracts of forest can be relatively stable.

4.2.4 Changes in Global Climate and Other Indirect Human Effects

Evaluating the long-term outcome of carbon mitigation activities will require estimating how carbon reservoirs will change in the future. Carbon stocks sequestered through mitigation activities today may be more or less secure, depending on how the environment changes and how society adapts to those changes. Estimating future C stocks in ecosystems is complicated by our inability to predict the magnitude and impact of impending changes in the environment. Some of the possible changes favour larger C stocks; others would lead to smaller stocks. The impact of global climate change on future C stocks is particularly complex. These changes may result in both positive and negative feedbacks on C stocks (Houghton *et al*., 1998). For example, increases in atmospheric CO_2 are known to stimulate plant yields, either directly or via enhanced water-use efficiency, and thereby to enhance the amount of C added to soils (Schimel, 1995; Woodwell *et al*., 1998). Higher CO_2 concentrations may also suppress decomposition of stored C, because C/N ratios in residues may increase and because more C may be allocated below ground (Owensby, 1993; Morgan *et al*., 1994; Van Ginkel *et al*., 1996; Torbert *et al*., 1997). Predicting the long-term influence of elevated CO_2 concentrations on the C stocks of forest ecosystems remains a research challenge (Bolin *et al*., 2000; Prentice *et al*., 2001).

Where plant growth is now limited by nitrogen (N) deficiencies, increased deposition of N associated with intensified production of bio-available N (Schindler and Bayley, 1993; Vitousek *et al*., 1997) may accelerate plant growth. This may, eventually, enhance the carbon stock of the soil (Wedin and Tilman 1996). Nadelhoffer *et al*. (1999) caution, however, that the global impact of N deposition may be comparatively small. Moreover, where the N fertilization effect increases growth, especially in the N-deficient northern forests, it also delays the hardening-off process, resulting in increased winter damage, and thus negating some of the growth enhancement (Makipaa *et al*., 1999).

Increased soil temperatures associated with increased atmospheric CO_2 have long been expected to result in increased soil respiration (Schimel, 1995; Townsend and Rastetter, 1996; Woodwell *et al*., 1998). Data recently reported by Giardina and Ryan (2000), however, suggest that decomposition of organic carbon in mineral soil layers is relatively insensitive to changes in air temperature. Modelling studies by Liski *et al*. (1998) suggest similar results. Nevertheless, IPCC reviews (Bolin *et al*., 2000; Prentice *et al*., 2001) conclude that existing terrestrial C sinks may gradually diminish over time, in part because of increasing losses via respiration.

Over the long term, as climate gradually changes, the time scales for adaptation of ecosystems to climatic conditions will become important. Vegetation types (and other organisms) have adapted to the combination of site conditions, including climate, where they now occur. It cannot be assumed that tree growth will increase with climate change, or that the plant populations will remain optimally adapted to their current sites. Analysis of provenance (seed source) data, in the light of global change, indicates either no net increase in growth rate as a result of warming or small decreases in growth rate. Trees may be under more stress in a changed climate, leaving them more susceptible to insects and diseases.

The various processes of environmental change may occur over different time periods and with varying intensity at different locations. Ecosystems that initially absorb C in response to higher atmospheric CO_2 will become “saturated” or even later release CO_2 if increasing temperatures lead to enhanced decomposition and respiration (Cao and Woodward, 1998; Scholes *et al*., 1999). Fires and other disturbances could increase in frequency and intensity if temperatures increase and precipitation patterns change. The net impact of these, and other global changes, is an area of active research (e.g., Hungate *et al*., 1997; Kauppi *et al*., 1997; Norby and Cotrufo, 1998; Woodwell *et al*., 1998).

The effects of climate change on mitigation activities in the terrestrial biosphere are difficult to anticipate, as they are dependent on the timing and the specific spatial character and distribution of changes. Present climate scenarios are neither spatially nor temporally very precise, and averages over the scale of typical global circulation climate models are inadequate for estimating impacts on very specific, localized mitigation activities. Moreover, the responses of ecosystems are dependent on the ecological mechanisms, the climate change imposed, and the management responses to these factors. For example, planting of species adapted to present conditions may be inappropriate for future conditions and the species might grow more slowly under chronic climate change. Conversely, species planted for an anticipated future climate may not be able to survive current variations.

Climate change can also affect the economic and social dimensions of land use and forestry. Currently, productive lands may become less productive and less attractive for food and fibre production. The current patterns of land use and disturbance could change. Model results reported by Darwin *et al*. (1995, 1996) and others suggest, for example, that conversion from forestland to cropland is a significant adaptive response to climate change in some regions. Protection from fire or insect and/or disease predation, in boreal regions especially, may become increasingly hard to maintain. Reliable estimates of risks to, or enhancements of, mitigation activities carried out today will require increased understanding of the interactions between the important ecological, economic, and social impacts of climate change. As described in this chapter, the carbon stocks in terrestrial ecosystems respond to a combination of ecological, economic, and social drivers. That will not change even if the global environment changes.

4.3 Processes and Practices that Can Contribute to Climate Mitigation

4.3.1 System Constraints and Considerations

In terrestrial ecosystems the carbon cycle exhibits natural cyclic behaviour on a range of time scales. Most ecosystems, for example, have a diurnal and seasonal cycle. Often this means that the ecosystem functions as a source of C in the winter and a sink for C in the summer, and this shows up in fluctuations at the global scale, as shown by the annual oscillations in the global atmospheric CO_2 concentration. Large-scale fluctuations occur at other temporal scales as well, ranging from decades (Braswell *et al*., 1997; Turner *et al*., 1997; Karjalainen *et al*., 1998; Kurz and Apps, 1999; Bhatti *et al*., 2001) to several centuries (Campbell *et al*., 2000) and longer (Harden *et al*., 1992).

The net balance of C flows between the atmosphere and the terrestrial biosphere also undergoes management-induced cycles that occur over long time scales (decades to millennia), and that can cause the transition of terrestrial systems from sink to source and back (Harden *et al*., 1992). Of relevance for C mitigation are the human-induced changes that occur on an annual to centennial time scale. This would include the harvest cycle of managed, production forests.

The intent of any mitigation option is to reduce atmospheric CO_2 relative to that which would occur without implementation of that option. Biological approaches to curb the increase of atmospheric CO_2 can occur by one of three strategies (IPCC, 1996):

- conservation: conserving an existing C pool, thereby preventing emissions to the atmosphere;
- sequestration: increasing the size of existing carbon pools, thereby extracting CO_2 from the atmosphere; and
- substitution: substituting biological products for fossil fuels or energy-intensive products, thereby reducing CO_2 emissions.

The benefits of these strategies show contrasting temporal patterns. Conservation offers immediate benefits via prevented emissions. Sequestration impacts often follow an S-curve: accrual rates are often highest after an initial lag phase and then decline towards zero as C stocks approach a maximum (e.g., *Figure 4.3*). Substitution benefits often occur after an initial period of net emission, but these benefits can continue almost indefinitely into the future (*Figure 4.6*).

This section deals primarily with carbon conservation and sequestration in the terrestrial biosphere, but acknowledges the complementarity and trade-offs among the three strategies. Carbon sequestration in forest products is included here and the substitution benefits of forest products are treated briefly. The role of energy cropping is treated in greater depth in Chapter 3 (Section 3.6.4.3) and in the IPCC Special Report on LULUCF (IPCC, 2000a). Here the discussion is restricted to the secondary use of biomass products for energy (e.g., waste products) and non-commercial uses (e.g., domestic heating, cooking, *etc*.).

The general goal of sequestration activities is to maintain ecosystems in the sink phase. However, if the system is disturbed (a forest burns or is harvested, or land is cultivated), a large fraction of previously accumulated C may be released into the atmosphere through combustion or decomposition (*Figure 4.2*). When the system recovers from the disturbance, it re-enters a phase of active carbon accumulation. Thus, the disturbance history of terrestrial ecosystems involves in large C loss-

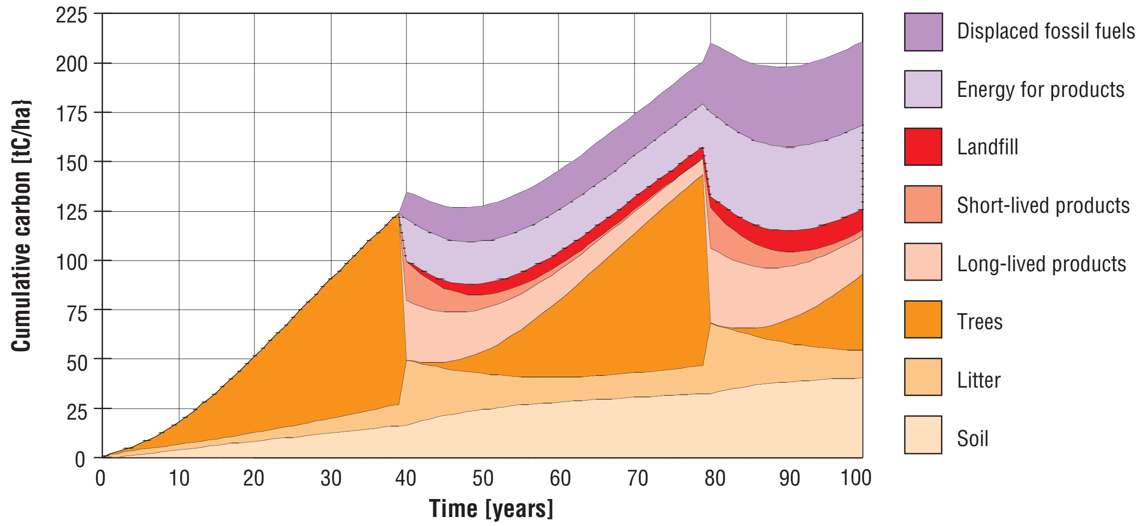

Figure 4.6: *Cumulative carbon changes for a scenario involving afforestation and harvest. These are net changes in that, for example, the diagram shows savings in fossil fuel emissions with respect to an alternative scenario that uses fossil fuels and alternative, more energy-intensive products to provide the same services (adapted from Marland and Schlamadinger, 1999).*

es in the past (Houghton *et al*., 1999; Kurz and Apps, 1999), but opportunities for C sequestration in the present.

A comprehensive systems analysis is useful to fully evaluate mitigation options. Factors to be considered may include: ecosystem C stocks and sinks; sustainability, security, resilience, and robustness of the C stock maintained or created; temporal patterns of C accumulation; other land-use goals and related C flows in the energy and materials sector; and effects on other non-CO_2 GHGs. For example, one option might have both a high maximum C stock and a high or more sustained rate of sequestration, yet be incompatible with other demands placed on the land. A second option may have a high maximum C stock, but reach that level only very slowly. Still another option may offer high short-term sequestration, but reach maximum C stocks very quickly. Yet another option might manage production systems to maximize the flow of harvested carbon into products, thus maximizing the displacement of alternate, energy-intensive products. Thus, while a wide array of practices may be technically possible, options that meet all criteria may be much fewer, and a combination of complementary options may best accomplish C mitigation goals. Although scientists now recognize the value of system-wide analyses (Cohen *et al*., 1996; Alig *et al*., 1997), rarely have mitigation options been subjected to such comprehensive evaluations.

An upper bound for the technical potential for global C mitigation in the terrestrial biosphere, a physical upper limit, can be estimated for conservation, sequestration, and substitution measures. The technical potential for conservation measures would equal the current existing C stock of the world's ecosystems. This assumes that all ecosystems are threatened, but all could be conserved by implementing protection measures. The technical potential for sequestration would roughly equal the carbon stocks lost in deforestation, desertification, and other human-induced changes in land cover and land use over centuries and millennia. The theoretical upper limit would thus correspond to the full recovery of lost biomass in ecosystems, and to a steady state at the natural carrying capacity for biomass on earth. The technical potential for substitution is related to the sustainable production of harvestable biomass and its substitution for fossil fuels and energy-intensive products. Clearly, each of these upper limits violates in practice the ideals of development, equity, and sustainability. And yet, they help to appreciate that there are bounds on the role that managing the biosphere might play in carbon mitigation.

4.3.2 Opportunities in Forests

Many silvicultural and forest management practices have been reported to enhance carbon mitigation (Lunnan *et al*., 1991; Hoen and Solberg, 1994; Karjalainen, 1996; Row, 1996; Binkley *et al*., 1997; Price *et al*. 1998; Birdsey *et al*., 2000; Fearnside, 1999; Anonymous, 1999; Nabuurs *et al*., 2000). Measures suggested for forests include: protecting against fires; protecting from disease, pests, insects, and other herbivores; changing rotations; controlling stand density; enhancing available nutrients; controlling the water table; selecting useful species and genotypes; using biotechnology; reducing regeneration delays; selecting appropriate harvest methods such as reduced-impact logging; managing logging residues; recycling wood products; increasing the efficiency with which forest products are manufactured and used; and establishing, maintaining, and managing reserves.

Sampson *et al*. (2000) provide an overview of the potential impacts of some different management alternatives on carbon mitigation, and examine both additional benefits and some possible unintended, negative effects of these practices. They estimate that 10% of the global forest area could be technically available by the year 2010, and that the global potential of forest management practices could be 0.17GtC/yr. These opportunities rise to 50% of the global forest area and 0.7GtC/yr by the year 2040. Sampson *et al*. (2000) emphasize win-win situations, but also indicate the low level of certainty associated with their estimates and the possibility for certain negative impacts.

Nabuurs *et al*. (2000) also estimate the potential of a broad range of forest-related activities (including protection from natural disturbance, improved silviculture, savannah thickening, restoration of degraded lands, and management of forest products) at 0.6 GtC/yr over six regions in the temperate and boreal zone (Canada, USA, Australia, Iceland, Japan, and EU, *Figure 4.7*). According to their estimates, alternative forest management for C sequestration is technically feasible on 10% (on average) of the forest area in each region examined. *Figure 4.8* shows that the relative importance of the different practices for the various regions depends on the current situation in the respective regions.

The analyses of Sampson *et al*. (2000) and Nabuurs *et al*. (2000) estimate that the hectare-scale effectiveness of these activities ranges from 0.02tC/ha/yr for forest fertilization to 1.2tC/ha/yr for several practices combined in Loblolly pine stands. However, they show that the impact of most practices is in the range of 0.3–0.7tC/ha/yr.

Forest management and protection offer high mitigation potential in some countries. For example, additional pools of 40-160tC/ha and 215tC/ha may be possible in Cameroon and the Philippines, respectively (Sathaye and Ravindranath, 1998). Afforestation or plantation forest options have the potential to increase carbon stocks by 70–100tC/ha in many places, and the potentials for some commercial plantations may be even higher: 165tC/ha for timber estates in Indonesia, 120 tC/ha for timber forestry in India, and 236tC/ha for long rotation forestry in the Philippines (Sathaye and Ravindranath, 1998). The suggested opportunities for mitigation potential in 12 developing countries are summarized in *Table 4.3*.

The study of Sathaye and Ravindranath (1998) suggests that, in 10 tropical and temperate countries in Asia, about 300Mha may be available for mitigation options: 40Mha for conservation, protection, and management; 79Mha of degraded forest land for regeneration; and 181Mha of degraded land for plantation forestry and, hence, for C sequestration (Sathaye and Ravindranath, 1998). A further 172Mha was estimated to be available in these countries for agroforestry. These esti-

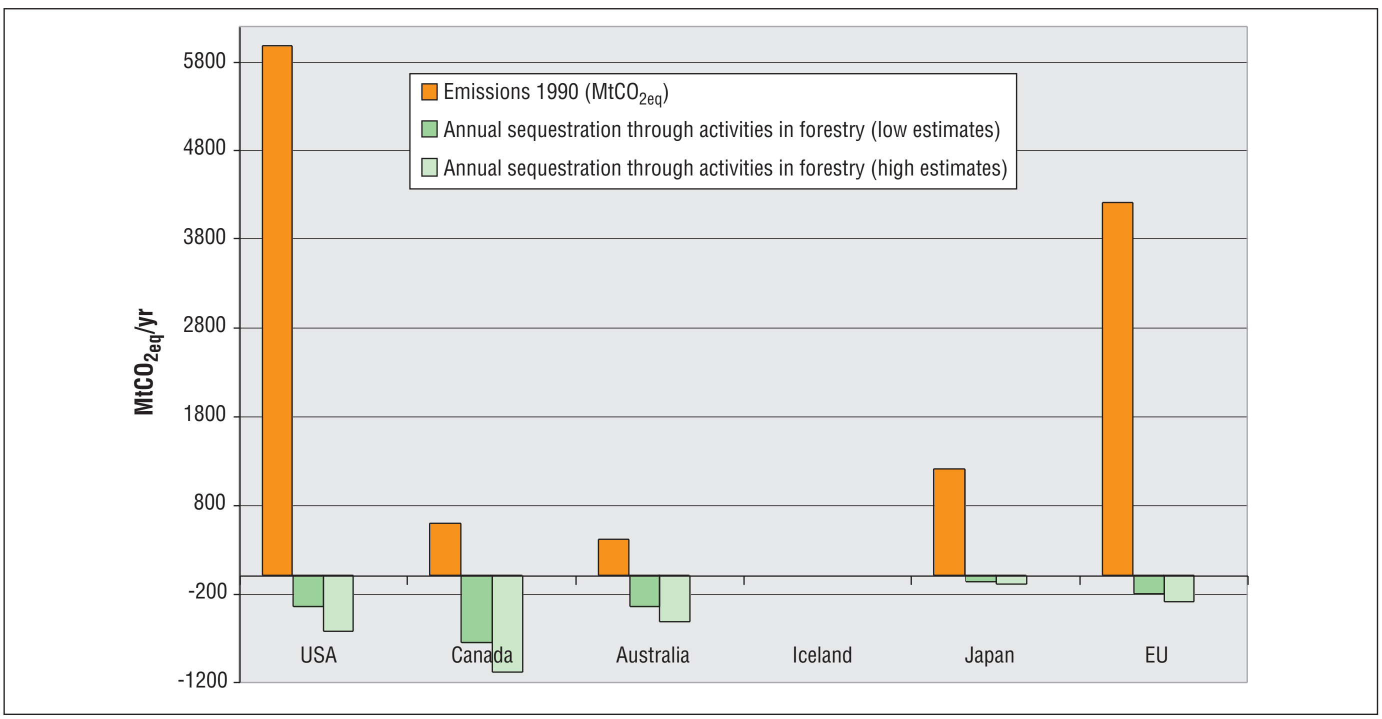

Figure 4.7: *Indications of the magnitude of the carbon sink in case study countries for a set of forest management measures ($MtCO_{2eq}$, adapted after Nabuurs et al. 2000). The values for the three bars for Iceland are 2.6, 2.8, and 2.9, respectively. The figure is based on the forest part of the model "Access to Country Specific Data" (ACSD). It was designed to provide insight into the potential magnitude of carbon sequestration that may be achieved when alternative sets of management measures are adopted. Therefore, the exact numbers provided in this figure result from the assumptions chosen for a certain set of measures. The estimates in this figure are tentative and only illustrative. In these studies all forestry activities under discussion were included, but applied on average on some 10% of mostly the exploitable forest area.*

Table 4.3: *Mitigation options, mitigation potential, and investment cost per tonne of carbon (US$/tC) abated in selected countries* (Sathaye and Ravindranath, 1998)

Mitigation option	Mitigation potential (tC/ha)	Investment cost[1] (US$/tC)	Mitigation option	Mitigation potential (tC/ha)	Investment cost[1] (US$/tC)
ASIA					
China			**India**		
North & North West					
Assisted natural regeneration[2]	13.0	1.3	Natural regeneration[2]	62.0	1.5
Plantation	55.0	1.3	Enhanced natural regeneration[2]	87.5	2.5
Agroforestry	15.0	16.3	Agroforestry	25.4	1.6
South, South West & North East			Community woodlot	75.8	5.6
Assisted natural regeneration	13.9	3.5	Softwood forestry	80.1	7.3
Plantation[3]	71.0	5.0	Timber forestry	120.6	3.3
Agroforestry	6.0	9.8			
Indonesia			**South Korea**		
Timber estate	165.0	1.9	Improved management of natural forest	99.4	6.0
Social forestry	94.0	1.1	Urban forestry	299.0	9.2
Reforestation[4]	214.0	0.9	Enhanced regeneration of		
Private forests	99.0	2.1	*L. leptolepsis*	123.0	13.8
Afforestation	106.0	0.6	*P. koraiensis*	85.0	21.0
Mongolia			**Pakistan**		
Private forests	99.2	0.8	Intensified forest management		
Natural regeneration	67.5	0.6	- *Conifer forest – protection*	41.6	0.1
Agroforestry	9.8	0.8	- *Conifer forest – natural regeneration (enhanced)*	33.8	8.8
Bioenergy	80	-	Reforestation[4]	39.1	19.3
Shelter belt	101.7	0.9	Riverain forest plantation	32.9	40.6
			Commercial forest plantation	54.6	40.6
			Watershed management	26.7	34.8
			Agroforestry	29.7	1.6
			Plantation on agricultural land[2]	7.5	0.7
			Rangeland management.	20.0	17.4
Philippines			**Thailand**		
Forest protection plus sustainable management	215.0	1.3	Short rotation in:		
Forest protection – total log ban	215.0	0.5	- *Managed forests*	185.5	2.5
Long rotation forestry	236.0	2.1	- *Non protected areas*	158.9	2.9
Urban forestry	90.0	5.3	Long rotation in community managed forests	169.0	3.2
			Medium rotation in non protected areas	112.5	4.3
			Forest protection and rotation forestry for conservation in		
			- *Protected area*	38.6	7.5
			- *Community managed forests*	38.1	10.7
Vietnam			**Myanmar**		
Forest protection	106.9	0.1	Natural regeneration	33.0	0.1
Degraded forest protection	64.3	0.2	Reforestation long[4]	155.0	0.8
Natural regeneration (enhanced)	57.1	0.8	Forest protection	47.0	1.6
Scattered trees	64.0	0.9	Reforestation short[4]	55.0	3.8
Reforestation short[4]	43.0	2.2	Bio electricity	78.0	21.4
Reforestation long	68.2	1.7			

(continued)

Tabel 4.3: *continued*

Mitigation option	**Mitigation potential (tC/ha)**	**Investment cost[1] (US$/tC)**	**Mitigation option**	**Mitigation potential (tC/ha)**	**Investment cost[1] (US$/tC)**
		AFRICA			
Ghana			**Cameroon**		
Evergreen forest			Evergreen forest		
- agroforestry	13-88	1-6	*- agroforestry*	16-58	1-5
- slowing deforestation	35-140	1-2	*- slowing deforestation.*	40-160	1-2
Deciduous forest			*- forestation[5]*	73-195	1-19
-slowing deforestation	35-140	1-2	Deciduous forest		
- forestation[5]	31-154	1-27	*- forestation[5]*	27-169	21-19
Savannah			Savannah		
- agroforestry	29-61	4-12	*- forestation[5]*	36-170	1-31

[1] Investment costs (US$/tC):This largely includes forest or plantation establishment costs incurred during the initial 2-3 years; discounted for only the initial 2-3 year period. For forest protection, the costs include expenditure on erecting barriers for protection, training, and other organizational costs incurred during the initial 2-3 year period. Mitigation potential is in pertuity, assuming one full cycle; rotation length for mitigation option subject to harvesting (such as short and long rotation) and for others 40 years.

[2] Natural regeneration of forest is increasing the biomass density to that of closed forests on partially degraded open forest areas; assisted or enhanced natural regeneration would involve planting a (few) trees and/soil and water conservation activity to assist or enhance natural regeneration.

[3] Plantations involve planting of one or more species at high densities.

[4] Reforestation in a short rotation has a 5 to 15 year harvest cycle, reforestation in a long rotation has a 30-100 year harvest cycle.

[5] Forestation includes both afforestation and reforestation.

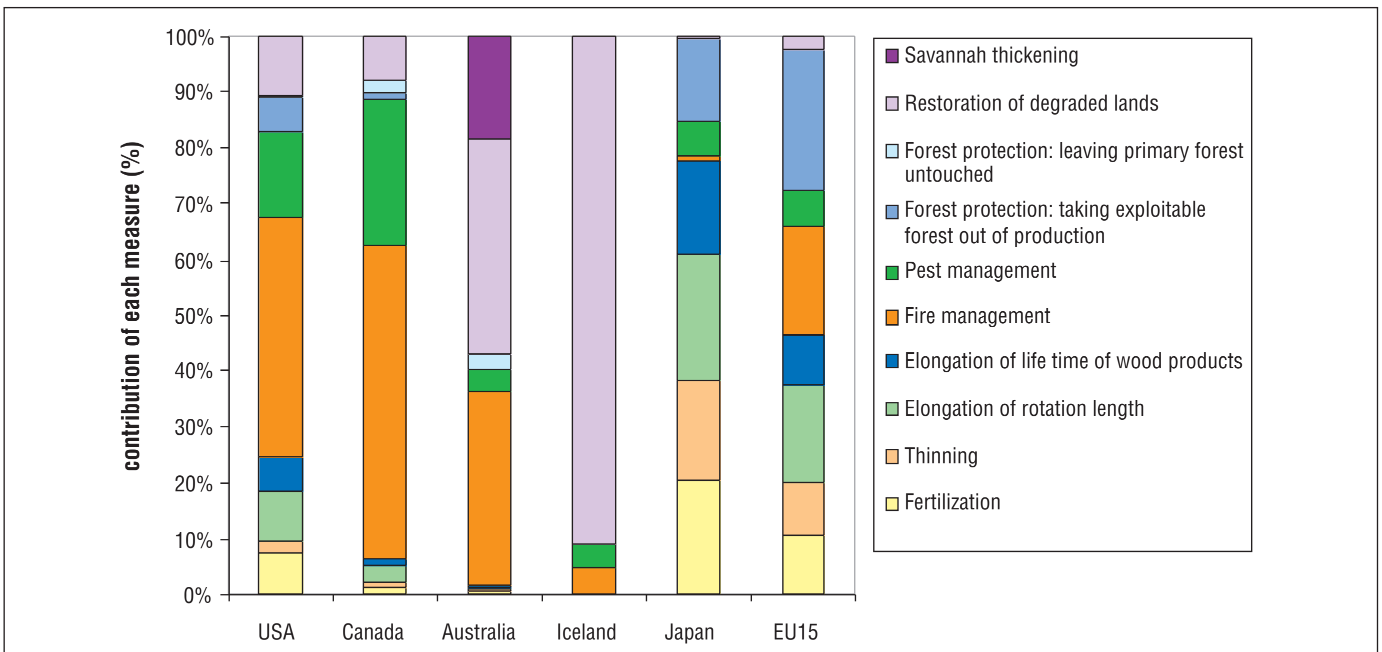

Figure 4.8: *Relative importance of each of the 10 forest management alternatives in the total potential sequestration as given in Figure 4.7. These data give an indication of opportunities and do not necessarily represent national plans. For example, silvicultural practices in Japan generally do not accompany fertilization and the figure for Japan is probably an overestimate. Nevertheless it shows that opportunities vary among countries because of both the national situation, the mix of current forestry practices, and/or the historic management. One common recommendation of which measures would yield the largest carbon sequestration can therefore not be given (adapted from Nabuurs et al., 2000).*

Table 4.4: *Land categories and extent of availability for mitigation in selected developing countries*
(Sathaye and Ravindranath, 1998)

Country	Forest land for conservation, protection, and management (Mha)	Degraded forest land for regener-ation (Mha)	Degraded land for plantation forestry (Mha)	Agroforestry (Mha)	Others (Mha)	Total geographic area (Mha)	Area under forests (Mha)
Asia							
China		19.2	105.2	75.9		932.6	134.0
India		36.9	41.3	96.0		329.0	63.3
Indonesia			30.5			193.0	144.7
Mongolia		2.4	1.6			156.6	17.5
Myanmar	3.3	6.9				65.8	49.3
Pakistan	0.5	0.3	2.6	1. 2		77.1	3.7
Philippines	6.6	2.5			0.60	29.8	6.5
South Korea	0.7	0.3			0.05	9.9	6.5
Thailand	17.8	4.4				51.1	14.0
Vietnam	10.5	6.0			2.50	32.5	19.0
Total	39.4	78.9	181.2	171.9	3.15	1877.4	458.5
Africa							
Cameroon	1.6		7.3	1.6		46.0	36.0
Ghana	0.9		0.3	2.5		23.0	18.0
Total	2.5	0	7.6	4.1	0	69.0	54.0
Total (12 countries)	41.9	78.9	188.8	176	3.15	1946.4	512.5

mates are much larger than those in IPCC (1996) (*Table 4.4*).

Current estimates suggest that the cumulative C mitigation potential of forests in 10 Asian countries is about 26.5GtC, suggesting that the SAR estimates for the tropical region were conservative. China (9.7GtC) and India (8.7GtC) have particularly large mitigation potentials in the forestry sector (Sathaye and Ravindranath, 1998).

Latin America, which accounts for 51% of the global area of tropical forests (FAO, 1997), has an estimated mitigation potential of at least 9.7GtC, an estimate based on analyses of Mexico, Venezuela, and partly Brazil (*Table 4.5*). This total includes native forest management, protected areas, commercial plantations, agroforestry, and restoration plantations. The technical potential C mitigation in forestry is estimated at about 4.8GtC for Mexico, 1.4GtC for Venezuela, and 3.5GtC for Brazil (Da Motta *et al*., 1999, *Table 4.5*). The feasible mitigation potential, which is largely constrained by land tenure policies and socio-economic pressures (land availability), is, however, often much lower than this technical potential. The feasible socio-economic mitigation potential is about 50% less than the technical potential in Mexico and about 44% lower than the technical potential in India (Ravindranath and Somashekar, 1995).

Deforestation in the Brazilian Amazon is a significant source of CO_2 and, with 90% of the originally forested area still uncleared, Brazil remains a large potential source of future emissions. The deforestation rate in Amazonia was estimated to be 1.38 million ha/yr in 1990, corresponding to an emission of 251MtC/yr (Fearnside, 1997). The rate of deforestation has increased in recent years, to 2.91Mha/yr in 1995 and 1.82Mha/yr in 1996 (Fearnside, 1998). Reducing the deforestation rate by 50% would conserve 125MtC/yr. Thus, Brazil alone offers a large potential for mitigation through slowing of deforestation.

What is the permanence of C sequestered by forest management activities? Clearly, tree plantations that are harvested and not re-established do not contribute to long-term carbon sequestration, though they may reduce atmospheric C in the short term. But, if a new forest is maintained so that harvest equals net growth, the forest can both be a source of wood products and still retain the captured C. In other words, the sequestration phase may be finite, lasting only a few decades, but the conservation phase need not be finite. Although there is an exchange of carbon between the atmosphere and the biomass, a considerable pool of carbon can be permanently stored in the steady-state biomass while wood products continue to be produced. This C pool remains withdrawn from the atmosphere as long as the forest exists. The substitution phase, which

Table 4.5: *Biological GHG mitigation potential in Latin America*

Option	Land available in 2030 (Mha)		Unit C seques-tration (tC/ha)	Total C sequestration(MtC)		Unit cost[b] (US$/tC)	Total cost	Reference
	Technical potential	Economic potential		Technical potential	Economic potential			
Native forest management								
Mexico	18.7	13.2	132	2465	1550	0.1-4	4930	Masera, 1995; Masera *et al*., 1997a; Masera and Ordóñez, 1997
Venezuela	9.8		75	735		9	6615	Bonduki and Swisher, 1995
Brazil	60		18	735		1.8	1323	Da Motta *et al*., 1999
Protected areas								
Mexico	6	4.9	89	535	470	1-6	1872.5	Masera, 1995; Masera *et al*., 1997a, Masera *et al*. 1995 and 1997b
Venezuela	4		94	376		4	1504	Bonduki and Swisher, 1995
Brazil	151		18	2718		3	7650	Da Motta *et al*., 1999
Restoration plantations								
Mexico	4.2	2.5	76	320	200	7	2240	Masera, 1995; Masera *et al*., 1997a, Masera and Ordóñez, 1997
Commercial plantations (includes energy plantations)								
Mexico	6.6	2.4	208	1375	1075	5-7	8250	Masera, 1995; Masera *et al*., 1997a, Masera *et al*. 1995 and 1997b
Venezuela	4.9		52-62	295		17	5015	Bonduki and Swisher, 1995
Brazil degraded land								
- Pulp			24			1.4		Da Motta *et al*., 1999
- Charcoal			180			0.7		Da Motta *et al*., 1999
- Timber			43			–9.5		Da Motta *et al*., 1999 '–' means profitable
Agroforestry								
Mexico	1.9	1.5	53	100	80	2-11	650	Masera, 1995; Masera *et al*., 1997a; Masera and Ordóñez, 1997; De Jong *et al*., 1995
Venezuela	1		27	27		20	540	Bonduki and Swisher, 1995
Total								
Mexico	37.4	24.5		4795	3375		17943	
Venezuela	19.7			1433			13674	
Brazil[a]	211			3453			8973	

a Unit carbon sequestration considers the difference between sustainable and unsustainable logging. Unit price is NPV(net of present value of benefits minus present value of costs)

b Unit cost US$/tC is NPV.

begins at the onset of the first harvest, can be sustained. Each timber crop, in a cumulative manner, can substitute for fossil-fuel resources. The forest thus offers a sustainable alternative to the unsustainable use of fossil-fuel resources (Schlamadinger and Marland, 1996).

Land owners are unlikely to manage their forest resources for C sequestration alone. In the absence of financial incentives, any C sequestration will likely be incidental, or have the role of a by-product in the management of forests to produce valued goods and services (ITTA, 1983, 1994). In the tropical biome, the optimal mix of management strategies will likely reflect a balance between various forest management systems and agricultural production. Existing policies for forest and agricultural land management, however, do not yet reflect economic incentives for C management and probably are not optimal (see for example Poore *et al*., 1989).

The effectiveness of various strategies for C sequestration will depend on the initial status of the forest ecosystems. For lands without tree cover, afforestation permits large C gains per hectare (Dyson, 1977; Sedjo and Solomon, 1989). Industrial plantation forests are already being created on a large scale and expansion of this area for C sequestration is possible (Sedjo and Sohngen, 2000). The establishment of forest plantations is generally the most reliable silvicultural method for afforestation, reforestation, and sustainable regeneration (regeneration soon after cutting). Plantation establishment can enhance productivity if desired species are planted on suitable sites. Plantations can reduce the pressures to degrade natural forests (Sedjo and Botkin, 1997). However, following the harvest of a mature or old-growth forest, the land can remain a source of carbon for many decades, even when it is regenerated (Hoen and Solberg, 1994; Cohen *et al*., 1996; Schlamadinger and Marland, 1996; Bhatti *et al*., 2001). Therefore, for primary and mature forests, conserving and protecting the existing C pools is often the only mitigation option that yields near-term benefits.

Because of the diversity in the current global forest status and socio-economic situation, the optimal mix of mitigation strategies will vary with country and region, in both the tropics and the non-tropics. For many countries, slowing or halting deforestation is a major opportunity for mitigation (e.g., Brazil: Fearnside, 1998, and Mexico: Masera, 1995). In countries such as India, where deforestation rates have declined to marginal levels, afforestation and reforestation in the degraded forest and non-forest lands offer large mitigation opportunities (Ravindranath and Hall, 1995). Ravindranath and Hall (1995) have shown the potential of using this degraded land and small biomass gasifiers to sustainably produce electricity from woody biomass and displace 40 million tonnes of C annually. In Africa an important opportunity for mitigation is in conserving wood fuel and charcoal through improved efficiencies of stoves and charcoal kilns (Makundi, 1998). The selection of mitigation strategies or projects in tropical countries, particularly, will be determined by economic development priorities, changing pressures on land use, and resource constraints. In many industrialized countries, adjusting forest management regimes and material flows in the forest products sector (including substitution) appears most promising (Hoen and Solberg, 1994; Binkley *et al*., 1997).

To quantify accurately the effects of changes in forest management on the net transfer of C to the atmosphere, the whole system could be considered (see *Box. 4.1*). Many earlier studies focused on the immediate results of forest management measures, e.g. the higher biomass growth rate following a silvicultural treatment or the protected stock of C if wildfire or logging is prevented. Global assessments based on these studies (e.g., Dixon *et al*., 1994; Brown *et al*., 1996b) have limitations. Estimates, in terms of tC/ha or tC/ha/yr, leave unanswered the critical questions of the timing, security, and sustainability of these effects. Also, recent, more comprehensive studies indicate the importance of complete accounting for all the C flows in and out of the system and the analysis of long-term patterns. For example, Schlamadinger and Marland (1996) showed that the positive effect of short-rotation plantations for fossil fuel substitution is less than implied by the simple substitution of fossil fuels, because of the continued input of fossil fuels needed to operate the system. While the limitations of earlier studies are now evident, data for comprehensive analysis at the global scale are not yet available. This, in part, explains why global-level estimates of the potential for C mitigation in forestry remain unchanged from those in SAR.

4.3.2.1 Wood Products

Wood products are an integral part of the managed forest ecosystem and the forest sector C cycle. They play three roles in the forest sector carbon cycle: (1) a physical pool of carbon, (2) a substitute for more energy-intensive materials and, (3) a raw material to generate energy (Burschel *et al*., 1993; Nabuurs and Sikkema, 1998; Harmon *et al*., 1996; Karjalainen, 1996; Matthews *et al*., 1996; Marland and Schlamadinger, 1997; Apps *et al*., 1999).

Wood removed from a forest by harvest, whether by thinning or clear-cut, can be viewed as a replacement for the natural mortality that would otherwise occur eventually (albeit at a faster rate). Harvested wood provides renewable raw material for use as fuel, fibre, and building materials; as well as income and employment for rural populations (Glück and Weiss, 1996). Globally, about 3.4 billion m^3 of wood are harvested per year, excluding wood that is burned on site (FAO, 1997). Harvest rates are expected to increase at 0.5% per year (Solberg *et al*., 1996). Of the total harvest, about 1.8 billion m^3 is for fuelwood, used mainly in the tropics. The total fuelwood consumption in tropical countries increased from 1.3 to 1.7 billion m^3 during the period 1990 to 1995 (FAO, 1997; Nogueira *et al*., 1998).

If the fossil fuel based energy required to produce and transport forest products is less than that needed for alternative products, then CO_2 emissions will be avoided by the use of forest products. Buchanan and Levine (1999) show, for example, that when wood is used for building construction in place of brick, aluminium, steel, and concrete, there can be net savings in CO_2 emissions. For construction of small buildings in New Zealand, the carbon substitution effect was larger than the direct carbon storage in wood building products (Buchanan and Levine, 1999). Forest products can also substitute in the marketplace for alternative materials, such as cement, that involve carbon emissions in their manufacture.

A systems approach has been used recently to recognize interdependencies among products and sectors. For example, Adams (1992) and Alig *et al*. (1997) examined the effects of sequestering C in forests in the USA on the availability of agricultural land, and Sedjo and Sohngen (2000) used a sectoral approach that explicitly recognized interrelations among various wood investment decisions, and between wood investment and C sequestration activities. The systems approach also recognizes the joint product nature of industrial wood and carbon sequestration. In a study in Argentina, for example, Sedjo (1999b) found that timber alone does not generate sufficient returns to justify plantation investment, but the simultaneous sequestration of C can justify investment above some threshold C price. The models do not yet incorporate a potential increase in demand for wood as a fuel to displace fossil fuels.

In the developing world most fuelwood and charcoal use is devoted to satisfying energy needs for cooking (Makundi, 1998). The potential for conservation of fuelwood is significant, both through improved cooking stoves and by substitution with liquefied or gasified biofuels. India, China, and some African countries have large programmes for the distribution of more efficient wood stoves. In India alone 28 million improved stoves have been disseminated (Ravindranath and Hall, 1995). The carbon mitigation costs of improved wood stoves in India range from US\$0.10/tC abated (Luo and Hulscher, 1999) to US\$12/tC abated (Ravindranath and Somashekar, 1995). A review of case studies in Asia showed an average mitigation cost of US\$0.8/tC abated in Thailand to US\$1.7/tC in India, through programmes to encourage use of improved wood stoves (Hulscher *et al*., 1999). The experience with wood stoves shows that – when appropriately designed, implemented, and monitored – efficient stove programmes can provide substantial benefits to local residents. There are no estimates of the global potential for carbon conservation via this option, however, in India alone it is estimated that 20MtC could be saved annually (Ravindranath and Hall, 1995).

There is also a significant potential for saving fuelwood and charcoal in a large number of small industries. Charcoal making, brick making, pottery making, bakeries, etc. use fuelwood as their primary energy source in many areas. Fuelwood and charcoal consumption in tropical countries is projected to increase from 1.34 billion m^3 in 1991 to 1.81 billion m^3 in 2010 (FAO, 1993).

Most of the forest harvest in the boreal and temperate zone is for industrial roundwood (i.e., cut logs). About one-half to two thirds of the roundwood finds its way into final products, and the rest is used for energy or ends up as decomposing residues (e.g., Apps *et al*., 1999). The annual production of roundwood, according to FAO (1997) statistics, corresponds to a harvest flux of about 1.6 billion m^3, resulting in about 0.9 billion m^3 in final products. This represents a C flux of about 0.3GtC/yr into the product pool.

According to the SAR (IPCC, 1996), the current global stock of C in forest products is about 4.2GtC and the net sink is 0.026GtC/yr. Other sources suggest a stock of 10-20GtC (Sampson *et al*., 1993; Brown *et al*., 1996b) and a global sink of 0.139GtC/yr (Winjum *et al*., 1998). There is a large uncertainty in the estimates. Even if the high end of the range is correct, the C sink in wood products appears small compared to the current rate of C sequestration in boreal and temperate forest ecosystems. Whether the physical pool of carbon in wood products in use acts as a sink depends on the relative rates of input and output from the product pool, i.e., the difference between the production of new products and the decay of the C stock in existing products (Apps *et al*., 1999).

Options to increase physical sequestration of carbon in wood products include:

- Increasing consumption and production of wood products;
- Improving the quality of wood products;
- Improving processing efficiency; and
- Enhancing recycling and re-use of wood and wood products.

Several studies have been carried out on the impacts of these measures on the amount of carbon sequestered in wood products. These studies generally conclude that the sink potential is quite small at the national or global level (Karjalainen, 1996; Nabuurs, 1996; Marland and Schlamadinger, 1997).

Use of wood as a fuel reduces CO_2 emissions from fossil fuels (Hall *et al*., 1991; Brown *et al*., 1996a; Nabuurs, 1996; Marland and Schlamadinger, 1997). Where the costs of growing biofuels on agricultural lands are higher than the costs of using fossil fuel, some form of incentive may be required to generate significant shifts to biofuels (Sedjo, 1997). The use of abandoned forest products for energy rather than disposal as waste can provide additional opportunities for displacing use of fossil fuels (Apps *et al,* 1999). Chapters 3 and 6 provide further discussion of the use of bioenergy within the energy sector.

Micales and Skog (1997) estimate that of the total amount of carbon-based products disposed of in the USA in 1993, as either paper or wood products, 28TgC (out of a total domestic harvest of approximately 123TgC/yr) will remain stored in landfills. Heath *et al*. (1996) and Karjalainen *et al*. (1994) emphasize the increasing role of landfills as a store of C.

Production of methane through anaerobic decomposition deserves to be considered when evaluating the mitigation potential.

While C sequestration in wood products can reach saturation, the C benefits of materials substitution can be sustained. Assuming a material substitution effect of 0.28tC/m^3 of final wood product (Burschel *et al*., 1993), and a flux corresponding to a roundwood volume of 0.9 billion m^3 annually, the substitution impact of industrial wood products may be as large as 0.25GtC/yr. Although this estimate is highly uncertain, it is possible that for wood products the substitution impact is larger than the sequestration impact. This substitution is additional to the sinks in wood products mentioned before.

4.3.2.2 Managing Wetlands

Globally, wetlands contain large reserves of organic carbon - about 300 to 600GtC (Gorham, 1991; Eswaren *et al*., 1993; Scharpenseel, 1993; Kauppi *et al*., 1997). A major portion of this carbon is found in peat-forming wetlands (peatlands), often associated with forests, in both northern (302Mha, 397GtC) and tropical (50Mha, 144GtC) biomes (Zoltai and Martikainen, 1996). Over the long term, peatlands gradually accumulate additional carbon, because decomposition is suppressed under flooded conditions (Harden *et al*., 1992; Mitsch and Wu, 1995; Rabenhorst, 1995; Zoltai and Martikainen, 1996; Kasimir-Klemedtsson *et al*., 1997). The beneficial effect of this carbon accumulation, however, is at least partially offset by release of methane, which is also a GHG (Gorham, 1995).

There are few opportunities to augment the accumulation of carbon in wetlands by improved management. Drainage of forested peatlands, largely concentrated in boreal regions, can enhance tree growth significantly, but the net ecosystem carbon changes are less clear – some studies report large net gains while others indicate large net losses of carbon to the atmosphere (see review by Zoltai and Martikainen, 1996). A more important mitigation measure, from the perspective of atmospheric CO_2, is the preservation of the vast carbon reserves already present (van Noordwijk *et al*., 1997) in peatlands. Drainage of wetlands for agricultural or other uses results in rapid depletion of stored C (Kasimir-Klemedtsson *et al*., 1997).

4.3.3 Opportunities in Agricultural Lands

Most ecosystems, under constant conditions, eventually approach a steady-state C stock that is dictated by management, climate, and soil properties. But changes imposed on the ecosystem can alter the balance of C inputs and losses, shifting the ecosystem, eventually, to a new steady state (Paustian *et al*., 1997c). For example, after conversion of forests or grasslands to arable agriculture, losses of C often exceed inputs temporarily, resulting in a net loss of C to the atmosphere until a new, lower equilibrium level is reached (Balesdent *et al*., 1998; Huggins *et al*., 1998; Solomon *et al*., 2000). At least a portion of C lost, however, can often be recovered by adopting management practices that again favour higher C stocks (Cole *et al*., 1997). The accumulation of C in soil can continue until a new steady state is reached, often after several or more decades. Most of the additional C is stored in the soil as organic matter. Apart from agroforests, agricultural lands store very little carbon in plant biomass (*Table 4.1*).

There are two general ways of increasing C stocks in agricultural lands: by changing management within a given land use (e.g., cropland, rice land, grazing land, or agroforests) or by changing from one land use to another (e.g., cropland to grassland or cropland to forest) (Sampson *et al*., 2000). In this section, we review briefly the possible ways of increasing C stocks in agricultural lands, first within a land use and then by a change in land use. We then review recent estimates of the potential for increasing C stocks in agricultural lands globally. A more detailed assessment of management practices and corresponding rates of C accrual is reported in the IPCC Special Report on LULUCF (IPCC, 2000a).

Croplands, as referred to here, are lands devoted, at least periodically, to the production of arable crops (wetland rice, because of its unique features, is discussed separately). Soil C in these lands can often be preserved or enhanced by using farming systems with reduced tillage intensity, thus slowing the rate at which soil organic matter decomposes (Bajracharya *et al*., 1997; Feller and Beare, 1997; Rasmussen and Albrecht, 1997; Dick *et al*., 1998). Another way to promote higher soil C is to increase crop yields. This can be done by applying organic amendments, by effective use of fertilizers, by using improved crop varieties, or by irrigating. These practices help replenish soil organic matter by increasing the amount of crop residues returned to the soil (Raun *et al.,* 1998; Huggins *et al*., 1998; Paustian *et al*., 1997b; Lal *et al.,* 1998; Smith *et al*., 1997; Fernandes *et al*. 1997; Izac 1997). Further, soil C can often be increased by using practices that extend the duration of C fixation by photosynthesis; for example, cover crops, perennial forages in rotation, and avoiding bare fallow tend to increase organic C returns to soil (Lal *et al*., 1997; Singh *et al*., 1997a; Smith *et al*., 1997; Carter *et al*. 1998; Tiessen *et al*., 1998; Tian *et al*., 1999; Paustian *et al*., 1997a, 2000). Farming techniques that reduce erosion (e.g., terracing, windbreaks, and residue management) maintain productivity and also prevent loss of C from agricultural soils. The net effect of soil erosion on atmospheric CO_2 is still uncertain, however, because the C removed may be deposited elsewhere and at least partially stabilized (van Noordwijk *et al*., 1997; Lal *et al*., 1998; Stallard, 1998).

Rice land, as the term is used here, refers to areas that are at least periodically flooded for wetland rice production. Carbon stocks in these systems can be preserved or enhanced by the addition of organic amendments (Singh *et al*., 1997b; Kumar *et al*., 1999) and nutrient management (Yadav *et al*., 1998). Rice lands, however, are an important source of methane and, from the standpoint of overall radiative forcing, management effects on methane emissions may be more important than effects on C storage (Greenland, 1995; Sampson *et al*., 2000). Methane emissions can be suppressed to some extent by soil amendments, altered tillage practices, water management, crop rotation, and cultivar selection (Minami, 1995; Kern *et al*., 1997; Neue, 1997; Yagi *et al*., 1997; Van der Gon, 2000). For more information on CH_4 and N_2O emissions from land use, see Section 3.6.

Grazing lands refer to natural grasslands, intensively managed pastures, savannas, and shrublands used, at least periodically, to graze livestock. One way to increase C stocks in these lands is to introduce new plant species. For example, the introduction of N-fixing legumes increases productivity, thereby favouring C storage (Fisher *et al*., 1997; Conant *et al*., 2001). Large increases in soil C have been also reported from the introduction of deep-rooted grasses in South American savannas (e.g., Fisher *et al*., 1994), though the area over which these findings apply is still uncertain (Davidson *et al*., 1995). Other management practices that can affect C storage include: changing grazing intensity and frequency (Manley *et al*., 1995; Ash *et al*., 1996; Burke *et al*., 1997, 1998); adding nutrients, especially phosphorus (Barrett and Gifford, 1999); controlling fire (Burke *et al*., 1997; Kauffman *et al*., 1998); and irrigation (Conant *et al*., 2001).

Agroforests include trees on farms as part of the agricultural landscape (Sampson *et al*., 2000). Unlike most other agricultural systems, agroforests store C in the above and below ground vegetation as well as in soil organic matter (Fernandes *et al*., 1997; Woomer *et al*., 1997). Examples of practices that can enhance C stocks include: integrated pest management, optimum tree densities, superior tree or crop cultivars, and better nutrient management (Sampson *et al*., 2000).

Land-use conversion involves transferring a given land area from one use to another. Where the shift is to a land use with higher potential C storage, the conversion can result in increased C stocks. For example, conversion of cropland to grassland often increases soil C (e.g., Paustian *et al*., 1997b; Reeder *et al*., 1998; Potter *et al*., 1999; Post and Kwon, 2000). Carbon stocks may also be enhanced by conversion of cropland to forests (reforestation, afforestation) or to agroforests (e.g., Fernandes *et al*., 1997; Woomer *et al*., 1997; Falloon *et al*., 1998; Post and Kwon, 2000). In some cases, cultivated lands can be restored as wetlands (Paustian *et al*., 1998; Lal *et al.,* 1999), resulting in carbon gains, though this practice may also result in higher net CH_4 emissions (Willison *et al*., 1998; Batjes, 1999; Sampson *et al*., 2000).

Another form of land-use conversion is the rehabilitation of severely degraded lands. Severely degraded lands are those where previous management has caused a drastic decline or disruption of productivity. Large areas of degraded lands occur on lands previously used for agriculture; lands abandoned after excessive erosion, over-grazing, desertification, or salinization (Oldeman, 1994; Lal and Bruce, 1999). Often the degradation was caused by social and economic pressures, and land rehabilitation may depend on the amelioration of the underlying causes of degradation. Specific rehabilitation practices include: introduction of new species (e.g., reforestation), addition of nutrients, and organic amendments (e.g., Lal and Bruce, 1999; Lal *et al*., 1998; Izaurralde *et al*., 1997).

Various attempts have been made to estimate potential C storage by improved management of agricultural lands. In the IPCC Second Assessment Report, Cole *et al*. (1996) estimated the potential for C storage in agricultural soils from improved management of existing croplands, restoration of degraded lands, and conversion to grass or forestlands. By assuming that one-half to two-thirds of the estimated historic C loss from cultivated soils could be recovered in 50 years, they proposed potential soil C increases of about 0.4 to 0.6GtC/yr from better management of existing agricultural soils. According to their estimates, additional C could be stored by set-aside of surplus upland soils (0.015 to 0.03GtC/yr), restoration of wetlands (0.006 to 0.012GtC/yr), and restoration of degraded lands (0.024 to 0.24GtC/yr), yielding a combined potential of about 0.44 to 0.88GtC/yr over a 50-year period. Later studies have provided similar estimates. Lal and Bruce (1999), using rates of soil C gain from the literature, estimated global C storage potentials of 0.43 to 0.57GtC/yr in the next 20-50 years, from erosion control, soil restoration, conservation tillage and residue management, and improved cropping practices. Batjes (1999), based partly on C gains estimated by Bruce *et al*. (1999), proposed that an additional 14GtC (±7) could be stored in agricultural soils over the next 25 years by improved management of "degraded" and "stable" agricultural lands. Including "extensive grasslands" and "regrowth forests" increased the estimate to 20GtC (±10), corresponding to an average rate of 0.58 to 0.80GtC/yr.

Sampson *et al*. (2000) recently completed a comprehensive assessment of potential net C storage from land management as part of the IPCC Special Report on LULUCF (IPCC, 2000a). According to their estimate, improved management within a land use could result in global rates of C gain, in 2010, of 0.125GtC/yr for cropland, <0.008GtC/yr for rice paddies, 0.026GtC/yr for agroforestry, and 0.237GtC/yr for grazing land. Potential rates of C gain in 2010 for land use conversion were 0.391 GtC/yr for conversion of unproductive cropland and grasslands to agroforests, <0.004GtC/yr for restoring severely degraded land, 0.038GtC/yr for conversion of cropland to grassland, and 0.004GtC/yr for conversion of drained land back to wetland. Corresponding rates of potential C gains for 2040 were consistently higher than those for 2010, often by a factor of about 2, though confidence in these values was lower. Sampson *et al*. (2000) cautioned that their estimates "are approximations, based on interpretation of available data" and that, "for some estimates of potential carbon storage, the uncertainty may be as high as ±50%".

Most of these estimates assume widespread, concerted adoption of C-conserving practices, and all have high uncertainty, stemming in part from the difficulty of predicting adoption of

C-conserving practices. The various estimates, furthermore, cannot always be compared directly because of differences in practices, scope, time-frame, and underlying assumptions. Most of the more recent estimates, however, are within the same order of magnitude as those presented in the SAR (Cole *et al*., 1996).

Increases in soil carbon content in response to improved practices cannot continue indefinitely. Eventually, soil C storage will approach a new equilibrium where C gains equal C losses (Paustian *et al*., 2000). This new equilibrium will depend on the management practices adopted, as well as on soil type and climatic conditions. Consequently, rates of C gain will diminish with time, and estimates for a given year cannot be extrapolated far into the future.

Once soils reach a new equilibrium, there is little further accumulation of C. And if the C-conserving practice is discontinued (e.g., reversion from no-tillage to intensive tillage), much of the previously gained C may be lost back to the atmosphere as CO_2 (Dick *et al*., 1998; Stockfisch *et al*., 1999). Consequently, the C stocks stored in soils are not necessarily permanent and irreversible.

4.4 Environmental Costs and Ancillary Benefits

4.4.1 Environmental Costs and Ancillary Benefits in Forests

Forests serve many environmental functions aside from carbon mitigation. Natural forests with various stages of stand development, including old-growth forests with snags and fallen logs, provide diverse habitats necessary for biodiversity (Harris, 1984; Franklin and Spies, 1991). Stopping or slowing deforestation and forest degradation, therefore, not only maintains carbon stocks but also preserves biodiversity, as shown by studies in Belize (EPA/USIJI, 1998) and Paraguay (Dixon *et al*., 1993).

Although plantations usually have lower biodiversity than natural forests (Yoshida, 1983: Kurz *et al*., 1997; Frumhoff and Losos, 1998), they can reduce pressure on natural forests, leaving greater areas to provide for biodiversity and other environmental services (Sedjo and Botkin, 1997). Plantations can negatively affect biodiversity if they replace biologically rich native grassland or wetland habitat, but non-permanent plantations of exotic or native species can be designed to enhance biodiversity by stimulating restoration of natural forests (Keenan *et al*., 1997; Lugo, 1997; Parrotta *et al*., 1997a, 1997b). Measures to promote biodiversity of intensively managed plantations include the adoption of longer rotation times, reduced or eliminated clearing of understory vegetation, use of native tree species, and reduced chemical inputs (Allen *et al*., 1995; Da Silva Jr *et al*., 1995; Fujimori, 1997).

Preserving forests conserves water resources and prevents flooding. For example, the flood damage in Central America following hurricane Mitch was apparently enhanced by loss of forest cover. By reducing runoff, forests control erosion and salinity. Consequently, maintaining forest cover can reduce siltation of rivers, protecting fisheries and investment in hydroelectric power facilities (Chomitz and Kumari, 1996).

Afforestation and reforestation, like forest protection, may also have beneficial hydrological effects. After afforestation in wet areas, the amount of direct runoff initially decreases rapidly, then gradually becomes constant, and baseflow increases slowly as stand age increases towards a mature stage (Kobayashi, 1987; Fukushima, 1987), suggesting that reforestation and afforestation help reduce flooding and enhance water conservation. In water-limited areas, afforestation, especially plantations of species with high water demand, can cause significant reduction of streamflow, affecting inhabitants in the basin (Le Maitre and Versfeld, 1997). The hydrological benefits of afforestation may need to be evaluated site by site.

Forest protection may, however, have negative social effects, such as displacement of local populations, reduced income, and reduced flow of subsistence products from forests. Conflicts between protection of natural ecosystems and their other functions, such as production of food, fuelwood, and roundwood, can be minimized by appropriate land use on the landscape (Boyce, 1995; Forman, 1995) and appropriate stand management.

In arid and semi-arid regions, where deforestation is advancing (Kharin, 1996) and leading to carbon loss (Duan *et al*., 1995), restoring forests by afforestation and proper management of existing secondary forests can help combat desertification (Cony, 1995; Kuliev, 1996). Afforestation of desertified lands may be limited, however, by costs and insufficient knowledge of ecology, genetics, and physiology (Cony, 1995). In relatively arid regions, fuelwood plantations may reduce pressure on natural woodlands, thereby retarding deforestation (Kanowski *et al*., 1992).

Agroforestry can both sequester carbon and produce a range of economic, environmental, and socioeconomic benefits. For example, trees in agroforestry farms improve soil fertility through control of erosion, maintenance of soil organic matter and physical properties, increased N, extraction of nutrients from deep soil horizons, and promotion of more closed nutrient cycling (Young, 1997). Thus, agroforestry systems improve and conserve soil properties (Nair, 1989; MacDicken and Vergara, 1990; Wang and Feng, 1995). Examples of mitigation projects that promote soil conservation through agroforestry include the AES Thames Guatemala project, and the Profator project in Ecuador (Dixon *et al*., 1993; FACE Foundation, 1997).

We note that decisions to protect or enlarge forest cover on a large scale could also have secondary climate consequences through their feedbacks on the earth's albedo, the hydrological cycle, cloud cover, and the effect of surface roughness on air movements (see, for example, Pielke and Avissar, 1990; Nobre *et al*., 1991; Garratt, 1993). Analyses by Bonan and Shugart (1992) suggest that large-scale changes in vegetative cover in the boreal zone may be especially important, with potentially global-scale impacts. In the boreal zone the albedo contrast between forested and unforested land during the winter is particularly large (differences as large as 40%). Indications are that the nature, magnitude, and even direction of climate changes driven by changes in surface vegetative cover will depend on the nature, location, hydrological setting, etc. of the vegetative change.

4.4.2 Environmental Costs and Ancillary Benefits in Agricultural Lands

Management strategies that conserve C in agricultural soils may have ancillary benefits quite apart from atmospheric CO_2 removal. Foremost among these is a favourable effect on soil productivity. Numerous studies have shown a strong link between the organic C content of a soil and its quality for crop production (e.g., Carter *et al*., 1997; Christensen and Johnston, 1997; Herrick and Wander, 1997). Consequently, a gain in soil C may promote crop yields, and preserve or enhance future soil productivity (Cole *et al*., 1997; Rosenzweig and Hillel, 2000). For example, application of fertilizers to agro-pastoral systems in parts of South America may not only induce soil C accumulation, but also enhance agricultural productivity (Fisher *et al*., 1997). Many of the practices advocated for soil C conservation – reduced tillage, more vegetative cover, greater use of perennial crops – also prevent erosion, yielding possible benefits for improved water and air quality (Cole *et al*., 1993). As a result of these benefits, adoption of practices that promote C conservation in agricultural lands is often justified even without the additional benefits arising from CO_2 mitigation.

Soil carbon sequestration, however, may sometimes have some potential adverse effects on the emission of other GHGs, notably nitrous oxide (N_2O). Where the C accumulation requires addition of higher amounts of N as fertilizer or manure, it carries the risk of increased N_2O emissions (Cole *et al*., 1993; Batjes, 1998). Furthermore, some C-conserving practices like reduced tillage may increase N_2O emissions by favouring higher soil moisture content (Cole *et al*., 1993; MacKenzie *et al*., 1997; Ball *et al*., 1999), though this effect is not always observed (e.g., Jacinthe and Dick, 1997; Lemke *et al*., 1999). Because the radiative forcing of N_2O is about 310 times that of CO_2 (kg per kg), when calculated over a 100-year time frame (IPCC, 1996), even a small increase in N_2O emissions, if confirmed, can significantly offset gains from C sequestration.

Carbon sequestration strategies may also have an effect on energy use and, hence, CO_2 emission from fossil fuel use. Changes in fertilizer use, pesticides, and agricultural machinery may enhance or offset any gains in soil C because of CO_2 released from fossil fuel. For example, roughly 1 kgC (or more) is released into the atmosphere as CO_2 per kgN used (Flach *et al*., 1997; Janzen *el al*., 1998; Schlesinger, 1999). In tropical areas where shifting cultivation is now practiced, intensification of crop production may maintain higher C stocks, by leaving more land under natural forest, but additional fossil fuel may have to be used to compensate for the fuelwood previously collected from the fallow period (van Noordwijk *et al*., 1997). In some cases, the adoption of C-conserving practices may reduce energy use. For example, using less intensive tillage may not only favour soil C gains, but also permits savings in CO_2 emission from fossil fuel combustion (Kern and Johnson, 1993). An evaluation of the net benefit of a C-sequestering practice, therefore, must consider energy use in addition to changes in C stocks. Whereas the duration of soil C gain in response to improved management may be finite, savings in CO_2 emissions from energy use continue indefinitely (Cole *et al*., 1997).

Aside from their secondary effects on GHG emissions, practices that sequester soil C may also have other potential adverse effects, at least in some regions or conditions. Possible effects include enhanced contamination of groundwater with nutrients or pesticides via leaching under reduced tillage (Cole *et al*., 1993; Isensee and Sadeghi, 1996), and possible environmental effects from widespread application of manures or sludges (Batjes, 1998). These possible negative effects, however, have not been widely confirmed nor quantified, and the extent to which they may offset the environmental benefits of C sequestration is uncertain.

4.5 Social and Economic Considerations

4.5.1 Economics

The method of calculating costs for forestry and agricultural projects differs. Forestry almost always looks at private market costs. However, many, if not most, forestry projects have positive externalities (or ancillary benefits) in the form of erosion control, water protection, flora and fauna habitat, non-timber forest products, water protection, and so forth (Makundi, 1997; Frumhoff *et al*., 1998; Trexler and Associates, 1998). For agricultural projects the approach is typically tied to the idea that the carbon-sequestering projects are essentially productivity enhancing and therefore can be viewed as “no regrets” activities; these are actions that have benefits in themselves aside from climate mitigation, which make the project socially desirable even without its carbon benefits. Such “no regrets” activities generally take the form of soil management activities, which both generate increased sequestered carbon and improve agricultural productivity.

There are basically three different ways of estimating the costs of sequestration of forestry projects – point estimates, i.e., cost for a particular level of output; partial equilibrium estimates, e.g., a cost function construction with the prices of inputs being held constant; and more general equilibrium types of approaches, e.g., a market equilibrium model in which some other prices, such as the prices of land inputs and the relative price of all other goods, are allowed to change owing to market forces. Additionally, economic models can incorporate changing climate conditions to estimate changes in economic variables as the climate and ecosystem change. Early studies tended to look at individual projects, relating the private costs of establishing a project to the cumulative carbon sequestered over the life of the project (see Sedjo *et al*., 1995). Many of the point estimate type studies provide undiscounted private market cost point estimates of the carbon sequestration in afforestation projects. However, this approach usually reveals little about how costs might change if the project were expanded to involve truly large land areas, as they do not recognize rising costs required to increasingly bid land away from alternative uses. These types of estimates tended to be biased downwards, partly because the opportunity costs of the land (land rents) were often ignored.

Point Estimates: The cost estimates of actually sequestering carbon obtained in point estimate type of studies tend to be quite low; in the SAR (IPCC, 1996) a range was given of US$3-US$7 per tonne of carbon. Additionally, a large number of more recent point estimate country studies reported most unit abatement costs in this low range, or lower. The earlier IPCC estimates for SAR were that of an investment of US$ 168-220 billion required to mitigate 45-72GtC in the tropical regions. More recent work provides estimates that the cumulative investment required for mitigating 26.53GtC to be US$63.6 billion at an overall cost of US$2.4/tC (Sathaye and Ravindranath, 1998). The unit cost given in *Table 4.3* shows that the investment cost of mitigation is generally quite low for carbon conservation options in selected developing countries and South Korea (e.g., US$0.10/tC in Vietnam, and US$1- 2/tC in Cameroon and Ghana). The mitigation cost is lower than US$2/tC for the majority of the options in Indonesia, the Philippines, Vietnam, and Mongolia.

Partial Equilibrium: Partial equilibrium involves a more complete estimation of a static cost function that estimates rising costs (e.g., as a result of land price increases as one moves to lands with higher opportunity costs) associated with increased sequestration activities. These studies generate marginal cost functions that tend to suggest most costs are higher than those of the simple point estimates. This is because, for example, they include in the cost estimates the opportunity costs of the land, and they recognize rising costs associated with additional planning activity and, for some, because they apply a discount rate to future physical carbon sequestered. The costs for modest amounts of carbon sequestered in specific areas are generally in the US$20-US$100/tC range (Moulton and Richards, 1990; Adams *et al*., 1995; Parks and Hardie, 1997;Stavins, 1999; Plantinga *et al.,* 1999). Costs tend to depend on the forest growth rates anticipated and the opportunity costs of the land. Where projects are small, land prices would be expected to be stable. However, in regions where projects are large, land prices, and hence sequestration costs, will tend to rise.

Market Equilibrium Models: This approach incorporates sectoral and general equilibrium interrelationships. It recognizes that expanding the forest for carbon sequestration purposes has implications for current and future industrial forest production and prices, and for agricultural production and prices. These price and production changes then generate feedbacks through the market to the forest and agricultural sector behaviour. Alig *et al*. (1997), for example, examine the effects on welfare costs of meeting alternative carbon sequestration targets by land re-allocations between agriculture and forestry in the USA. This model explicitly treats agriculture and wood production as interrelated. Allocating more land to trees to capture carbon has implications on the price and quantity of agricultural products, as well as on timber. Thus, the costs of carbon plantations are found both in the price of establishing the plantations and in the higher agricultural prices, and thus involve welfare shifts across sectors. A different approach, also recognizing sectoral interrelationships, is that of Sedjo and Sohngen (2000). This approach expands on earlier global timber supply models by explicitly incorporating the interrelations between the industrial wood sector and carbon plantations by recognizing the joint product nature of industrial wood and carbon. This approach finds that tree planting carbon sequestration activities tend to have a somewhat more modest effect than anticipated, since the tree planting for carbon purposes leads to an expected increase in future timber supplies and a corresponding decrease in expected future prices. Through the effects of price expectations on the timber market, carbon activities may discourage industrial timber investments and thereby lose some of the carbon gains made from the initial project. This is a form of leakage not often recognized.

Climate Feedback Models: These market equilibrium models incorporate the impact of the climate-driven changing ecology into their assessment of the potential and costs. Perez-Garcia *et al*. (1997) examine the effects of climate change, using a global trade model (CGTM). This approach imposes a global circulation model (GCM) and a terrestrial ecosystem model on the world's industrial wood economy, and estimates the welfare effects on forest owners and forest consumers of such changes. Sohngen and Mendelsohn (1998) use a timber model of the USA to estimate the changes in the forest market sector that would be expected to occur with a climate warming using GCM and terrestrial ecosystem models. However, neither study considers the impacts of increased fuelwood demand to replace fossil fuels.

In summary, most studies, of all methodologies, suggest that there are many opportunities for relatively low-cost carbon sequestration through forestry. Estimates of the private costs of sequestration range from about US$0.10-US$100/tC, which are modest compared with many of the energy alternatives (see *Table 3.9* and *Figure 4.9*). Additionally, it should be noted that most forest projects have positive non-market benefits, thus increasing their social worth. However, as the studies have become more sophisticated, incorporating both the full private

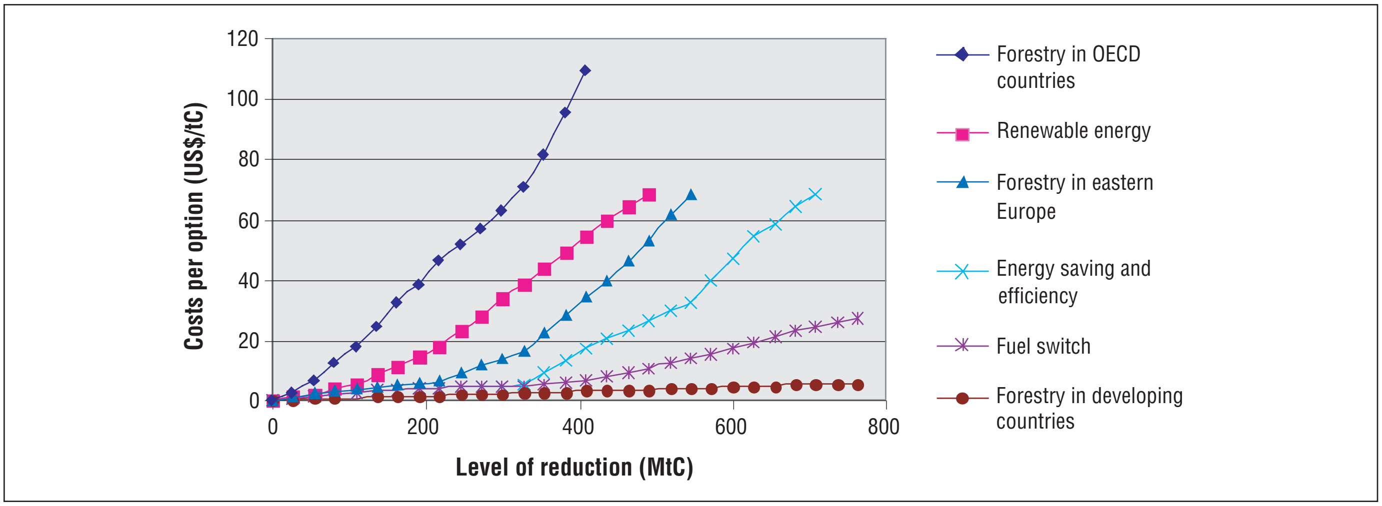

Figure 4.9: *Indicative curves of costs (US\$/tC, cost of US\$28/tC is equivalent to US\$100 per tonne of CO_2) of emission reduction or carbon sequestration by level of total reduction. The curves display how comparable options vary in costs between world regions. However, costs per option are also reported to vary widely at comparable total levels of reduction. This is mainly because cost studies have not been carried out in the same way. In some options net monetary profit may occur as well (i.e., costs may be negative as well) (Brown et al., 1996a, Hol et al., 1999; Jepma et al., 1997; Sedjo et al., 1995).*

opportunity costs of the land and market effects on land and resource prices, estimates of carbon sequestration costs have tended to rise. The cost estimates tend to vary for regions, with high costs generally associated with high opportunity costs for land. In the many regions that have low opportunity costs for land, including many subtropical regions, the costs tend to remain low.

4.5.2 Institutional Structures and Equity Issues

In order to realize the mitigation potential in part or in full, it would be helpful to have a set of institutions to translate the policies and measures into avoided emissions or carbon sequestration. In the United Nations Conference on Environment and Development (UNCED) held in 1992, the importance of sustainable forest management was emphasized under the "Forest Principles." Subsequently, the formulation of criteria and indicators was worked out under the Helsinki and Montreal Processes, in which the maintenance and enhancement of forest resources to contribute to the global carbon cycle is described. The same is a criterion under the United Nations (UN-ECE/FAO, 2000). The three main types of necessary institutions are global and/or regional, national and local, and/or community based (IPCC, 2000b). At the global level, there exist government-based multilateral institutions such as the United Nations Food and Agriculture Organization (FAO), United Nations Environment Programme (UNEP), the World Bank, and the International Tropical Timber Organization (ITTO). All of these institutions are involved in natural resource management, and can play a significant role in integrating mitigation objectives in tropical forest management. Also, a wide array of non-governmental organizations (NGOs) active in resource conservation and sustainable utilization, as well as bilateral aid organizations, could play a more important role in incorporating mitigation in their policy objectives. For example, investment promotion agencies could be created to assist in the co-ordination of investment into carbon projects (e.g., see Moura-Costa *et al*., 1999). Additionally, global private enterprises could be encouraged to include climate mitigation measures in their plans. Financial incentives may be required to achieve broad participation.

In tropical countries, forestry is dominated by government-based institutions, mostly the departments of forestry and agriculture and/or those involved in environmental management (WRI, 1987). These departments may need support and new insight in order to effectively incorporate mitigation policies and measures in their resource management activities. At a national level, there also exist some institutions involving NGOs that focus on conservation and forest expansion, as well as those dedicated to encouraging sustainable agriculture. Such institutions may also include umbrella organizations involved in developmental activities such as gender, poverty alleviation, etc. A few institutions, including non-governmental and especially those involved in nature conservation and environmental services, e.g., game reserves, tourism companies, and large-scale agricultural production, could also incorporate mitigation considerations in their efforts.

At the local level, effective institutions include community leaderships, religious institutions, schools, traditional organizations, and indeed the family. These institutions are essential with regard to natural resource management and agricultural practices, as well as for introducing mitigation-type activities that do not contravene their basic needs to use their land and natural resources for sustenance.

Public, NGO, and private institutions, at each spatial level where they exist, can focus on including GHG mitigation as one of their considerations, while they oversee the use of forest and land resources to meet the developmental aspirations of those in tropical countries and elsewhere. For example, a recent study on sustainable livelihoods and carbon management discussed arrangements to facilitate the involvement of small-scale farmers and rural communities in carbon trading (Bass *et al*., 2000). An optimal mix of conservation, sequestration, and substitution will be incidental or arise from the optimal management of forest resources for producing desired goods and services as shown under various tropical forest management stipulations (ITTA, 1983, 1994). In the tropical biome, the most likely use of the optimal mix of management strategies will be based on optimal management of forestry and agricultural resources in each country. For example, balance between forest conservation, afforestation, reforestation, and multiple land use of the forest and agricultural areas will predetermine the extent of utilization of the land-use sectors for mitigation activities. However, the existing policies in managing forest and agricultural resources have been criticized as non-optimal (see, for example, Poore *et al*., 1989). Optimal levels of substitution will be determined by the energy and industrial policies of these countries, rather than by carbon sequestration criteria.

The so-called "no regrets options" can be identified and pursued (see Chapters 7 and 8 for a discussion of no regrets options). Analysis has suggested that adequately designed and implemented GHG mitigation options in forestry and agriculture could help advance the countries' own development priorities, at the same time providing significant carbon sequestration (see Sheinbaum and Masera (2000) for analysis at the country level).

4.6 Market and Non-market Options to Enhance, Maintain, and Manage Carbon Pools

4.6.1. Introduction, Taxes, and Quotas

There are a host of market and non-market options to manage carbon pools in the terrestrial biosphere. Some of the most relevant questions related to carbon sequestration deal with the types of instruments, policies, and mechanisms that could play a role in promoting increased sequestration and how the various arrangements would actually affect outcomes. Market mechanisms could be important in promoting or discouraging carbon sequestration. Potential mechanisms might include taxes or subsidies for activities that affect carbon directly or that affect activities with large carbon implications. The UK, for example, has proposed a "Climate Change Levy" to be adopted by the UK's 2001 Budget. The Kyoto Protocol introduces flexible mechanisms allowing joint implementation, emissions trading, and the clean development mechanism. When dealing with terrestrial systems any policies that influence land use can affect carbon sequestration. Finally, there is the question of how the various instruments and policies are likely to influence leakage of carbon flows outside the targeted system.

Agricultural subsidies are common in many, if not most, countries. Agricultural subsidies and absent forestry subsidy policies can often be viewed as discouraging forest production and thus, inadvertently, discouraging some possibilities for carbon sequestration. Similarly, tax policies can promote or discourage certain types of land use. In some countries, however, subsidies do promote afforestation and reforestation. The movement of land from agriculture to forests generally leads to gains in the forest sector and losses in the agricultural sector. The cost of any additional carbon storage can involve a change in welfare across two sectors. Lower taxes for agricultural lands and subsidies for forest clearing may be part of the package of instruments to promote development.

To reach objectives for carbon sequestration, market mechanisms are important, but an appropriate institutional setting is also useful. In some tropical countries the profitability of maintaining forests could be improved in order to prevent conversion to alternative uses of the land. Success could entail revising policies that directly or indirectly subsidize cattle ranching (as has been historically the case in Latin America) or agriculture. Success in C sequestration could also entail technical and financial training and capacity building at the local level. It should be recognized, however, that in many tropical countries, particularly within Asia and Africa, forests are harvested and used according to the subsistence needs of local communities. In these cases, some have argued that approaches based on market mechanisms will not be effective. Also, non-timber forest products are an important component of the total demand for forest products and could be considered.

It is clear that some measures aimed at sequestering C in the biosphere have relatively low cost compared to other approaches for mitigating the atmospheric increase of CO_2 (Section 4.5). However, to date only a small number of projects involving a small and varied group of stakeholders has been initiated in the terrestrial biosphere. These projects (forest expansion, forest management, soil carbon management, community forestry, and agroforestry) have covered, worldwide, an area of 3.6 to 6.4 million hectares in 1999 (for an overview see Brown *et al*., 2000). Incentives that would create projects aimed at carbon sequestration in the biosphere on a large scale are not yet in place.

An important change in motivating carbon sequestration has been the creation of the Kyoto Protocol, in December 1997. Although few countries have yet to ratify the Kyoto Protocol, it introduces ceilings and/or quotas for CO_2 emissions for Annex B countries. In addition, the Kyoto Protocol explicitly recognizes afforestation, reforestation, and deforestation (ARD) as having carbon implications, and it provides credits (and debits) for these activities in meeting carbon-emissions targets. This arrangement has contributed to pressures to find ways to give sequestered carbon value in the market place. A detailed explanation of how the Kyoto Protocol might influence management of C stocks is given in the IPCC Special Report on LULUCF (IPCC, 2000a).

Through setting emissions targets and introducing taxes on CO_2 emission in some countries, carbon gains monetary value and could become a new product for the forestry sector. From existing emissions taxes this value is estimated at US\$200/tCO_2 in Norway and US\$100/tCO_2 in the Netherlands (Solberg, 1997; Nabuurs, 1998). In the case of the Netherlands, this carbon value is equivalent to US\$17.5/m^3 of roundwood, more than the stumpage value of wood as a raw material. However, in the first trades of certified carbon credits, Moura-Costa and Stuart (1998) found that prices ranged between US\$5–US\$10/tC. More generally, Moura-Costa and Stuart (1998) found that the average price for carbon credits for carbon sequestered in developing countries ranged from US\$0.19 to US\$12/tC, and that these differences are very much linked to uncertainty about long term policy.

The Dutch Government is considering the introduction of CO_2 certificates as part of a test of CO_2 emissions trading. In this system, each economic sector and each firm could achieve its targets partly through certificates. Funds generated from these certificates would be used to establish forests.

4.6.2 Carbon Offsets, Tradable Permits, and Leakage

Markets created for carbon credits from management of the biosphere, of course, will be heavily influenced by the many other commodities produced by the biosphere (food, wood, etc.). Food security may, for example, be a reason for a government to continue its policy of agricultural subsidies in the absence of forestry (carbon) subsidies. On the other hand, some studies (e.g., Callaway and McCarl, 1996) have shown that when diverting agricultural subsidies to carbon payments, the net impact on the national budget could be zero. In tropical countries, the institutions and subsidies for forest clearing may remain as part of the package to promote economic development. Only if the monetary value of carbon stocks and sinks is recognized and paid for will markets be efficient in encouraging C sequestration. Some developing countries see markets for C offsets as providing resources to facilitate capital inflows to finance conservation and other activities.

An emerging instrument that is likely to have a large effect on carbon sequestration is the tradable emissions permit. Tradable permits to deal with environmental pollutants have precedents in other areas. In the USA, for example, there is an active market for sulphur emissions permits (Burtraw, 2000). Firms with excess emissions permits can trade these to firms in need of additional permits. Thus, incremental emissions are no longer free, but incur additional costs to the firm. Firms that have excess permits can either sell those permits or forego the opportunity of receiving a payment – an opportunity cost. Such an approach allows the market to reallocate emission permits, and thus emissions, to the users that receive the highest return from the permits, thereby distributing carbon emissions permits to the most efficient users. This approach is beginning to be contemplated in addressing the problem of increasing atmospheric carbon and is endorsed in the Kyoto Protocol.

Currently, there are a series of brokers prepared to trade carbon credits in the USA and Europe, e.g., Natsource and Canto Fitzgerald (Stuart and Moura-Costa, 1998), and the Sydney Futures Exchange in Australia is planning to begin trading in the latter part of 2000[1]). In addition to tradable carbon emission permits, the door is open for consideration of an analogous instrument, tradable "carbon offsets". Activities, such as planting and protecting forests, could provide carbon sequestration services that could be sold or traded.

To date there is only limited experience with certified carbon offset instruments. In the USA, the electrical power industry, through the Edison Electric Institute (EEI - an association of private electrical power companies), has formed the Utility Carbon Management Tree Program whereby the various member companies invest money into a project fund to develop or purchase carbon offset credits (Sedjo, 1999a). Another market approach has been created, the Certified Tradable Offsets, issued by the Costa Rican government, and the first carbon-backed securities worldwide (Stuart and Moura-Costa, 1998). These offsets are like JI or CDM as defined in the Kyoto Protocol, but would be tradable.

A potentially serious problem with carbon offsets is that there may be carbon leakage. Leakage refers to the situation in which a carbon sequestration activity (e.g., tree planting) on one piece of land inadvertently, directly or indirectly, triggers an activity which, in whole or part, counteracts the carbon effects of the initial activity. It can be shown that most of these types of problems arise from differential treatment of carbon in different regions and circumstances, and the problem is not unique to carbon sequestration activities but pervades carbon mitigation activities in the energy sector as well.

In land use, leakage can occur from either protection or planting activities. Suppose, for example, that a forest or wetland that was to be cleared is instead protected. Protection of one such forest or wetland may simply deflect the pressure to another piece of land that is not protected and will be cleared instead. Leakage can occur across both spatial and temporal boundaries. Additionally, a forest protected in one year is subject to the possibility of clearing in subsequent years.

A similar situation may also exist with activities such as tree planting. Trees provide at least two services: producing industrial wood and sequestering carbon. Trees planted for carbon sequestration, because they may eventually be used for wood,

[1] See *International Herald Tribune*, 31/08/1999.

can affect expectations about future industrial wood prices, thereby influencing the planting decisions of forest products companies. If carbon credits are provided to carbon forests but not to industrial forests, and if some carbon forests are anticipated to enter future timber markets, then forest industrial firms may reduce investments in new forests. Such a reduction would partly offset carbon sequestered in the newly planted carbon forest, thereby reducing the net total carbon that would have accumulated by both industrial and carbon forests (Sedjo and Sohngen, 2000). This leakage effect would not occur if both industrial and carbon forests could expect to receive payment for both their carbon and their wood.

Leakage from industrial forests, resulting from forests established for carbon purposes, has been estimated by Sohngen and Sedjo (1999) to be about 40%, globally, assuming that all carbon forests are made available to the timber market. This compares with estimated leakages in the energy sector of about 5%–20%. No estimates of leakage generated from protection activities are available, but it is suggested that it may vary by country and site, unlike planted forests that are linked through the global timber market.

The leakage problem may be addressed reasonably well within nations by caps imposed on total emissions, but leakage of emissions across national boundaries may still occur in the absence of global coverage.

Conceptually, a permanent net carbon offset should be equivalent to a tradable emissions permit. If a new activity permanently reduces net atmospheric carbon by one tonne, the climatic implications are the same as if the tonne of carbon was never released. Thus, a carbon-offset credit would be equivalent to a tradable emission credit. However, since carbon offset can quickly be liquidated, offset credits have greater liability problems. One approach might be treated on an annual (or decadal) basis as the rental of (perhaps temporary) carbon sequestering services. Although different from carbon emissions permits, they nevertheless would expand the number of "credits" available, and thus have a mitigating effect on the market price of the credits. A discussion of some of the options is presented in IPCC (2000a).

4.6.3 Risks, Rights, and Practical Economics

Protecting forestlands, grasslands, and other natural ecosystems is often proposed as the best way to maintain large carbon reservoirs at lowest cost. The cost of such an approach, however, may in fact be significant, although low in comparison with many of the options in the energy sector and attempts at forest protection have failed in many parts of the world. The incentives to convert often far outweigh the incentives to protect. This problem is often exacerbated by the absence of well-defined, enforceable property rights, either private or public, and the absence of other necessary institutions. In an open access situation the incentives are to "use it or lose it", since there are no certain claims on the future use of the resource. Because there is no long-term claim on the resource in the future, the result is that resources may be used wastefully in excess of their economic optimum. Thus, deforestation and land clearing are a form of the open access problem (Hardin, 1968).

The costs of carbon management may not be distributed in the same way as the benefits. Carbon management options in developing countries may have low market costs but high local social costs in land commitments, and the benefits that arise may not be shared with local peoples. Analysis of a forest protection project in Madagascar suggests that there are financial benefits for local inhabitants and social benefits for the global community, but short-term debits at the national level (Kremen *et al*., 2000). Formal adoption of markets for forest carbon could increase incentives for forest protection, especially if mechanisms assure that local peoples share in the benefits. Similarly, costs and benefits may be realized at different times; future benefits are often weighted against current costs. How communities value present and future costs depends on wealth, culture, and economic and environmental priorities.

International consensus on carbon management begins to have important implications for national sovereignty and personal property rights, an issue brought to prominence by recent turmoil regarding international trade agreements (see Chapter 6 for a detailed discussion on policies, measures and instruments).

4.7 Biological Uptake in Oceans and Freshwater Reservoirs, and Geo-engineering

The net primary production of marine ecosystems is roughly the same as for terrestrial ecosystems (50GtC/yr for marine ecosystems and 60GtC/yr for terrestrial ecosystems), and there are opportunities to increase the net carbon flow into the marine biosphere. There are fundamental differences between the two systems, however, as the marine biosphere does not include large stores of carbon in the living and dead biomass. There are some 3 GtC in marine biota versus nearly 2500GtC in terrestrial vegetation and soils (*Table 4.1*). The key to increasing the carbon stocks in ocean ecosystems is thus to move carbon through the small reservoir of the marine biota to the larger reservoirs of dissolved inorganic carbon (the "biological pump") in ways that will isolate the carbon and prevent its prompt return to the atmosphere. The biological pump serves to move carbon from the atmosphere to the deep oceans, as organisms take up CO_2 by photosynthesis in the surface ocean, and release the carbon when the organic material sinks and is oxidized at depth.

Several researchers have suggested that ocean productivity in major geographical regions is limited by the availability of primary or micronutrients, and that productivity could be increased substantially by artificially providing the limiting nutrients. This might involve providing nitrogen or phosphorus in large quantities, but the quantities to be supplied would be much smaller if growth were limited by a micronutrient. In particular, there is evidence that in large areas of the Southern Ocean productivity is limited by availability of the micronutrient iron. Martin (1990, 1991) suggested that the ocean could be stimulated to take up additional CO_2 from the atmosphere by providing additional iron, and that 300,000 tonnes of iron could result in the removal of 0.8GtC from the atmosphere. Other analyses have suggested that the effect may be more limited. Peng and Broecker (1991) examined the dynamic aspects of this proposal and concluded that, even if the iron hypothesis was completely correct, the dynamic issues of mixing the excess carbon into the deep ocean would limit the magnitude of the impact on the atmosphere. Joos *et al*. (1991) reported on a similar model experiment and found the ocean dynamics to be less important, the time path of anthropogenic CO_2 emissions to be very important, and the maximum potential effect of iron fertilization to be somewhat greater than reported by Peng and Broecker (1991).

Some of the concepts of iron fertilization have now been tested with 2 small-scale experiments in the equatorial Pacific Ocean. In experiment IronEX 1 (November, 1993) 480 kg of iron were added over 24 hours to a 64 km^2 area of the equatorial Pacific. In IronEX 2 (May/June, 1995) a similar 450 kg of iron (as acidic iron sulphate) were added over a 72 km^2 area, but the addition occurred in 3 doses over a period of one week.

The IronEx 1 experiment showed unequivocally that there was a biological response to the addition of iron. However, although plant biomass doubled and phytoplankton production increased fourfold, the decrease in CO_2 fugacity (in effect the partial pressure of CO_2 decreased by 10 micro atm) was only about a tenth of that expected (Martin *et al*., 1994; Watson *et al*., 1994; Wells, 1994). In the IronEX 2 experiment the abundance and growth rate of phytoplankton increased dramatically (by greater than 20 and twice, respectively), nitrate decreased by half, and CO_2 concentrations were significantly reduced (the fugacity of CO_2 was down 90μatm on day 9). Within a week of the last fertilization, however, the phytoplankton bloom had waned, the iron concentration had decreased below ambient, and there was no sign that the iron was retained and recycled in the surface waters (Monastersky, 1995; Coale *et al*., 1996; Cooper *et al*., 1996; Frost, 1996).

These two experiments have demonstrated that week-long, sustained additions of iron to nutrient-rich, but iron-poor, regions of the ocean can produce massive phytoplankton blooms and large drawdowns of CO_2 and nutrients. While the results of these two experiments cannot be uncritically extrapolated, they suggest a very important role for iron in the cycling of carbon (Cooper *et al*., 1996). The consequences of larger, longer-term introductions of iron remain uncertain. Concerns that have been expressed relate to the differential impact on different algal species, the impact on concentrations of dimethyl sulphide in surface waters, and the potential for creating anoxic regions at depth (Coale *et al*., 1996; Frost, 1996; Turner *et al*., 1996). There is much to be learned of the ecological consequences of large-scale fertilization of the ocean.

Jones and Young (1998) suggest that the addition of reactive nitrogen in appropriate areas, perhaps in conjunction with trace nutrients, would increase production of phytoplankton and could both increase CO_2 uptake and provide a sustainable fishery with greater yield than at present.

Chemical buffering of the oceans to decreases in pH associated with uptake of CO_2 leads to an increase in dissolved inorganic carbon that does not rely on alteration of the biological pump. Buffering of the oceans is enhanced by dissolution of alkaline minerals. Dissolution of alkaline materials in ocean sediments with rising pH occurs in nature, but does so on a time-scale of thousands of years or more (Archer *et al*., 1997). Intentional dissolution of mined minerals has been considered, but the quantity (in moles) of dissolved minerals would be comparable to the quantity of additional carbon taken up by the oceans (Kheshgi, 1995).

Stallard (1998) has shown that human modifications of the earth's surface may be leading to increased carbon stocks in lakes, water reservoirs, paddy fields, and flood plains as deposited sediments. Burial of 0.6 to 1.5GtC/yr may be possible theoretically. Although Stallard (1998) does not suggest intentional manipulation for the purpose of increasing carbon stocks, it is clear that human activities are likely leading to carbon sequestration in these environments already, that there are opportunities to manage carbon via these processes, and that the rate of carbon sequestration could be either increased or decreased as a consequence of human decisions on how to manage the hydrological cycle and sedimentation processes.

The term "geo-engineering" has been used to characterize large-scale, deliberate manipulations of earth environments (NAS, 1992; Marland, 1996; Flannery *et al*., 1997). Keith (2001) emphasizes that it is the deliberateness that distinguishes geo-engineering from other large-scale, human impacts on the global environment; impacts such as those that result from large-scale agriculture, global forestry activities, or fossil fuel combustion. Management of the biosphere, as discussed in this chapter, has sometimes been included under the heading of geo-engineering (e.g., NAS, 1992) although the original usage of the term geo-engineering was in reference to a proposal to collect CO_2 at power plants and inject it into deep ocean waters (Marchetti, 1976). The concept of geo-engineering also includes the possibility of engineering the earth's climate system by large-scale manipulation of the global energy balance. It has been estimated, for example, that the mean effect on the earth surface energy balance from a doubling of CO_2 could be offset by an increase of 1.5% to 2% in the earth's albedo, i.e. by reflecting additional incoming solar radiation back into space. Because these later concepts offer a potential approach for mitigating changes in the global climate, and because they are treated nowhere else in this volume, these additional geo-engineering concepts are introduced briefly here.

Summaries by Early (1989), NAS (1992), and Flannery *et al*. (1997) consider a variety of ways by which the albedo of the earth might be increased to try to compensate for an increase in the concentration of infrared absorbing gases in the atmosphere (see also Dickinson, 1996). The possibilities include atmospheric aerosols, reflective balloons, and space mirrors. Most recently, work by Teller *et al*. (1997) has re-examined the possibility of optical scattering, either in space or in the stratosphere, to alter the earth's albedo and thus to modulate climate. The latter work captures the essence of the concept and is summarized briefly here to provide an example of what is envisioned. In agreement with the 1992 NAS study, Teller *et al*. (1997) found that ~10^7 t of dielectric aerosols of ~100 nm diameter would be sufficient to increase the albedo of the earth by ~1%. They showed that the required mass of a system based on alumina particles would be similar to that of a system based on sulphuric acid aerosol, but the alumina particles offer different environmental impact. In addition, Teller *et al*. (1997) demonstrate that use of metallic or optically resonant scatterers can, in principle, greatly reduce the required total mass of scattering particles required. Two configurations of metal scatterers that were analyzed in detail are mesh microstructures and micro-balloons. Conductive metal mesh is the most mass-efficient configuration. The thickness of the mesh wires is determined by the skin-depth of optical radiation in the metal, about 20 nm, and the spacing of wires is determined by the wavelength of scattered light, about 300nm. In principle, only ~10^5t of such mesh structures are required to achieve the benchmark 1% increase in albedo. The proposed metal balloons have diameters of ~4 mm and a skin thickness of ~20nm. They are hydrogen filled and are designed to float at altitudes of ~25km. The total mass of the balloon system would be ~10^6t. Because of the much longer stratospheric residence time of the balloon system, the required mass flux (e.g., tonnes replaced per year) to sustain the two systems would be comparable. Finally, Teller *et al*. (1997) show that either system, if fabricated in aluminium, can be designed to have long stratospheric lifetimes yet oxidize rapidly in the troposphere, ensuring that few particles are deposited on the surface.

One of the perennial concerns about possibilities for modifying the earth's radiation balance has been that even if these methods could compensate for increased GHGs in the global and annual mean, they might have very different spatial and temporal effects and impact the regional and seasonal climates in a very different way than GHGs. Recent analyses using the CCM3 climate model (Govindasamy and Caldeira, 2000) suggest, however, that a 1.7% decrease in solar luminosity would closely counterbalance a doubling of CO_2 at the regional and seasonal scale (in addition to that at the global and annual scale) despite differences in radiative forcing patterns.

It is unclear whether the cost of these novel scattering systems would be less than that of the older proposals, as is claimed by Teller *et al*. (1997), because although the system mass would be less, the scatterers may be much more costly to fabricate. However, it is unlikely that cost would play an important role in the decision to deploy such a system. Even if we accept the higher cost estimates of the NAS (1992) study, the cost may be very small compared to the cost of other mitigation options (Schelling, 1996). It is likely that issues of risk, politics (Bodansky, 1996), and environmental ethics (Jamieson, 1996) will prove to be the decisive factors in real choices about implementation. The importance of the novel scattering systems is not in minimizing cost, but in their potential to minimize risk. Two of the key problems with earlier proposals were the potential impact on atmospheric chemistry, and the change in the ratio of direct to diffuse solar radiation, and the associated whitening of the visual appearance of the sky. The proposals of Teller *el al*. (1997) suggest that the location, scattering properties, and chemical reactivity of the scatterers could, in principle, be tuned to minimize both of these impacts. Nonetheless, most papers on geo-engineering contain expressions of concern about unexpected environmental impacts, our lack of complete understanding of the systems involved, and concerns with the legal and ethical implications (NAS, 1992; Flannery *et al*., 1997; Keith, 2000). Unlike other strategies, geo-engineering addresses the symptoms rather than the causes of climate change.

4.8 Future Research Needs

This chapter suggests a host of future research needs. A combination of statistical, ecological, and socio-economic research would be helpful to better understand the situation of the land, the forces of land-use change and the dynamic of forest carbon pools in relation to human activities and natural disturbance. More precise information is needed about degradation or improvement of secondary and natural forests throughout the world, but particularly in developing countries.

Some specific examples are:

- assessment of land available for mitigation options based on socio-economic pressures and land tenure policies. Furthermore, it would be beneficial if the impact of market price of carbon mitigated on land available for mitigation opportunities in different countries was understood;
- implications of financial incentives and mechanisms on LULUCF sector mitigation potential in different countries;
- comparative advantage (mitigation cost, ancillary benefits, etc.) of LULUCF sector mitigation options over energy sector opportunities;
- development and assessment of different approaches to developing baselines for LULUCF activities and comparison with other sectors; and
- socio-economic and environmental costs and benefits of implementing LULUCF sector mitigation options in developing countries, including issues such as property rights and land tenure.

Finally, an important consideration is the problem of leakages. Research would help to determine the conditions under which leakage is likely to be a serious problem and when it may be less so. Estimates of the degree of leakage under varying circumstances could be made so that appropriate adjustments in carbon credits can be made.

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