FORESTRY FOR MITIGATING THE GREENHOUSE EFFECT:

An ecological and economic assessment of the potential of land use to mitigate CO_2 emissions in the Highlands of Chiapas, Mexico.

Ben H.J. de Jong

NN08201, 2846

PROPOSITIONS

STELLINGEN

Behorende bij het proefschrift:

Forest management for mitigating the greenhouse effect: An ecological and economic assessment of the potential of land-use to mitigate CO_2 emissions in the Highlands of Chiapas, Mexico, door Ben H.J. de Jong, te verdedigen op 2 oktober 2000.

1. The overwhelming literature on greenhouse gas mitigation measures can not close the gap between recognizing the potential of biotic carbon mitigation and accepting that individual projects in the forestry sector can contribute to this modification, can be monitored and be verified as part of a carbon emissions control system (Trexler, 1993; this dissertation).

2. Land-use systems that provide commercial products combined with ecological services, such as carbon mitigation, can generate capital input for long-term investment in farm-forestry projects. (this dissertation)

3. Farmers in the rural communities of the Highlands of Chiapas prefer individual land-use projects above community-type activities, even if the latter option is more beneficial.

4. The choice of a baseline rate of biomass loss under a "business-as-usual" scenario is a critical issue for estimates of the carbon mitigation potential by forestry (Tipper and De Jong, 1998; this dissertation).

5. Land-use/land-cover change dynamics in the Highlands of Chiapas are particularly complex, with high levels of spatial and temporal variability, so that process-based prediction models have limited reliability (Ochoa-Gaona, 2000; this dissertation).

6. Van der Wal is right when claiming that traditional management practices in communal forests generate complex forest fragments with a high degree of variation in species composition, stand structure, and biomass densities (Van der Wal, 1999).

7. A time-integrated approach, with ton-years as their measuring unit, is the optimal option to calculate CO_2 benefits of forestry projects over time (Various authors, see Chapter 8; this dissertation).

8. A parametric approach for assessing whole-stand biomass dynamics via readily identifiable compartments, growth processes and first-order flow approximations is a flexible and adjustable method to simulate the annual quantities of biomass produced, allocated, or lost in response to natural stand dynamics, silvicultural interventions, and other factors that influence stand development.

9. A self-reporting system with on-site spot checks is the most viable option to monitor carbon mitigation in farm-forestry projects. (this thesis)

10. Global problems can only be solved by global solutions.

11. The certainty of future climate change creates a unique research opportunity, both to anticipate possible outcomes and to develop preventive measures.

12. Farmers are more interested in the frequencies of good and bad crop years than in changes of long-term average crop yields.

Promotor: Dr. Ir. R.A.A. Oldeman

Hoogleraar in de Bosteelt & Bosoecologie

Proefschrift

ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van Wageningen Universiteit,
dr. ir. L. Speelman,
in het openbaar te verdedigen
op maandag 2 oktober 2000
des namiddags te vier uur in de Aula.

Ben H.J. de Jong

Forestry for Mitigating the Greenhouse Effect:
An ecological and economic assessment of the potential of land use to mitigate CO_2 emissions in the Highlands of Chiapas, Mexico.

Keywords: Land use and land cover, carbon mitigation, cost-effective land use systems, forest management, Highlands of Chiapas, Mexico.

ISBN 90-5808-266-0

Inhoud

List of Figures

List of Photos

List of Tables

General acknowledgements

This dissertation is the result of a stimulating collective effort, in which participated various researchers, technicians, and local farmers of various communities. Among others, Richard Tipper and Guillermo Montoya-Gómez helped out in the economic assessments of chapters four and five and John Taylor and Kristen Nelson in the social assessments of chapter four. Richard Tipper and John Taylor designed the Plan Vivo concept of chapter seven. Lorena Soto-Pinto helped significantly in the definitions of the agroforestry systems of chapter four. Susana Ochoa-Gaona assisted in the Land-Use/Land-Cover classification of chapter three. Michael Cairns, Patricia Haggerty and Ignacio March-Mifsut coordinated the GIS work of chapter three, whereas Neptalí Ramírez-Marcial, Mario González-Espinosa and Susana Ochoa-Gaona developed the forest community analysis related to the same chapter. I am very grateful for their contributions.

I am obliged to the former director of "El Colegio de la Frontera Sur (ECOSUR)", Dr. Pablo Farías Campero, who allowed me to spend time to write this book. The idea to bring together the results of the research in a dissertation was mainly a result of his positive persuasion.

I appreciated very much the very important comments on draft versions of the various chapters by Professor Oldeman. His broad and unique vision of forest ecology and highly stimulating discussions were a constant inspiration to continue this arduous work.

Thanks also to Diny de Jong who converted the plain-looking documents into this book.

The financial support of the Stichting Pro Natura (Den Haag, NL) made it possible to include the colored plates, for which I am very grateful.

Susana and David Ochoa-Gaona made me realize that there are other important matters in life than carbon, climate change, or forestry. I am very fortunate to be part of their lives.

1. INTRODUCTION

General

During the last two centuries the concentration of atmospheric CO_2 has risen from 280 to about 345 parts per million (Neftel et al., 1985). On the one hand, this is a major factor of recent climate changes, particularly of the global energy budget. On the other hand, it is the consequence of impaired balances in the organization of life, as decomposition, respiration, and other oxidation processes increases CO_2 production, whereas deforestation and land degradation diminishes the CO_2 absorption capacity through photosynthesis by reducing the leaf surface area. Both processes are also linked to the sun's energy supply (Rossignol et al., 1998).

Identified sources of carbon emissions include burning of fossil fuels, cement production and land-use conversion (Cook et al., 1990). Forests play an important role in the global carbon (C) cycle. High- and mid-latitude forests are currently estimated to be a net C sink of about 0.7 ± 0.2 PgC yr^{-1} (Pg = 10^{15} gram). In contrast, low-latitude forests are estimated to be a net C source of 1.6 ± 0.4 PgC yr^{-1}, caused mostly by clearing and degradation of forests (Dixon et al., 1994).

The third Conference of the Parties to the United Nations Framework Convention on Climate Change (UN-FCCC), held in December 1997 in Kyoto, Japan describe two market-based mechanisms that will allow countries to trade in greenhouse gas emission (GHG) reductions:

1. Between two Annex 1 countries (countries with binding emission limits), known as Joint Implementation (JI), and
2. Between an Annex 1 country and a non-Annex 1 country (countries with no binding emission limits, mainly developing countries), known as Clean Development Mechanism (CDM).

Under a possible future carbon offset trading program, countries would be most likely to pay for greenhouse gas emission reductions in another country where the cost for reducing emissions is lower. With such a program international carbon emission offsets could become a currency for investing in emission reducing activities (Tipper and De Jong, 1998). The value of these offsets have the potential to create a system of incentives to develop C-saving projects worldwide, because it stimulates CO_2 emitters to seek the least expensive emission reduction measures (Swisher and Masters, 1992). Although forestry measures are not yet specifically included within the current articles relating to the CDM, provisions for forestry almost certainly will be included at some stage, given the significance of developing country forests within the global carbon cycle (Tipper and De Jong, 1998).

The potential land area available at the global scale for carbon conservation and sequestration is estimated to be 700 x 10^6 ha. The total potential amount of carbon to be sequestered and conserved through improved land use on this land by 2050 is assessed as some 60 to 87 PgC (Winjum et al., 1992; Dixon et al., 1993, 1996; Brown et al., 1995). The tropics have the potential to conserve and sequester by far the largest quantity (80%), followed by the temperate zone (17%) and the boreal zone (3%). Forestry and land-use mitigation measures can serve other environmental, economic, and social interest simultaneously, and may offer some of the most cost-effective ways to combine climate change mitigation and biosphere restoration. GHG offset projects in the land-use and forestry sector can particularly be attractive if they can be tied to local social, ecological and economic goals (Trexler, 1993).

Swisher (1991) defines carbon sequestration by a relevant unit of measurement for C storage in forestry projects, which is "the increment in CO_2 flux, expressed as Megagram of carbon equivalent (Megagram (Mg) = 10^6 gram = 1 ton of C), out of the atmosphere, compared to existing conditions (in the case of C removal) or to a reference condition (in the case of C emission prevention). The score keeping should explicitly account for the C storage of the land use without the project".

Evidence from a number of specific forestry projects that have been financed on the basis of the expected sequestration potential effect point to relatively low implementation costs of forestry and agroforestry C-mitigation projects (De Jong et al., 1995). The uptake of 1 x 10^9 MgC would require about 400,000 ha to be taken into mitigation management each year for 20 years at a cost of 20 US$ / MgC (Tipper et al., 1998).

Crediting the sequestration effect of forestry activities has hitherto been implemented on a project-by-project basis, estimating the difference in the long-term C-flux to the atmosphere between "baseline" and project scenarios. The lessons learned from these projects will serve as important precursors for future mitigation projects. However, for significant reduction of global carbon emissions, national governments will need to institute measures that both provide local and national benefits, and mitigate excesses of carbon.

The present study is intended to answer some of the important questions that arise from translating projects that have an ecological potential to mitigate carbon excesses, into the actual implementation of these projects in a farmer-dominated landscape. Initial studies by De Jong et al (1995) indicated that in regions such as Chiapas, the most appropriate methods to enhance carbon storage on land managed by small farmers are the introduction of trees within agricultural systems as crop-tree combinations or the restoration of degraded pastureland. Together, such activities are referred to as "farm forestry" (Foley and Barnard, 1984). On communally held areas of natural forest or secondary vegetation the main sequestration strategy is the restoration of degraded forest ecosystems, and the conservation and management of the existing tree stock in initiatives referred to as "community forestry" (Foley and Barnard, 1984). Farm and community forestry projects for GHG mitigation in such environments would be characterized by: - numerous participants, organized in various ways, generally a high variety of small-scaled systems, spread over large areas, with site-specific management, which farmers adapt individually due to personal interest, local conditions, and previous experiences. This is in brief the context of the Scolel Té international pilot project for carbon sequestration by forestry and agroforestry, being developed in Chiapas, southern Mexico (Scolel Té, 1998). The objective of this project is to develop a model for carbon sequestration that will be economically viable and can be technically transplanted in similar regions of Mexico and Latin America.

As part of the scientific backstopping of the Scolel Té project, this study intends to discuss the following key questions: (i) what is the effect of land-use land-cover change on carbon fluxes in highly fragmented landscapes?; (ii) which land-use systems do resource-poor farmers prefer that could contribute to the greenhouse gas problem?; (iii) what is the carbon mitigation potential of farmers' selected forestry and agroforestry systems, and what would be the cost?; (iv) what are the sources and levels of uncertainties in calculating carbon fluxes in forestry systems?; and (v) can a cost-efficient monitoring system be set up for carbon mitigation in a farm forestry project?

Institutional context

The research program was carried out within the context of the research project "Sistemas Silvícolas y Agroforestales" at "El Colegio de la Frontera Sur", unit San Cristóbal de las Casas. In this project, researchers, students and technicians work together with local inhabitants, to develop methodologies, procedures, and data bases, which are required to develop silvicultural options to manage the diversified forests of Southern Mexico and agroforestry techniques to improve current agricultural land-use systems. In Chiapas, the project focuses on the Highlands of Chiapas (Highlands), Northern Mountains of Chiapas, and the Selva Lacandona. The present study was restricted to the Highlands region, which comprises an elevated limestone mass of about 11,000 km^2 (between 1000 and 2,500 m a.s.l.), with steep volcanic rocks, rising up to 2900 m a.s.l. (Figure 1.1).

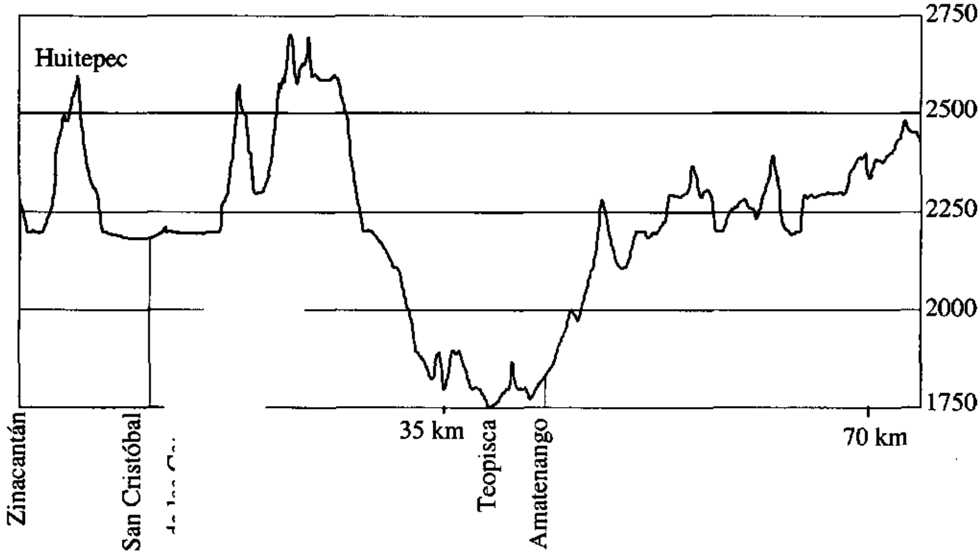


Figure 1.1. *Topographic profile (in m above sea level) of the Highlands of Chiapas. The profile is a little over 70 km long, taken in SSW direction. The profile is based on a digital elevation grid with 60 m resolution.*

In this area, farmers combine shifting cultivation and extensive grazing of cattle and sheep with uncontrolled extraction of timber, fuel, and other products from the forests. The research was carried out in close cooperation with the local farmers´ organization Pajal Ya Cak´tik.

Ecological and social context

The Highlands is one of the most densely populated rural areas in Mexico, habited by Tzotzil, Tojolabal, and Tzeltal Maya indigenous groups. From the 1970s onward, the Highlands were increasingly integrated in the economic activities of Chiapas, due to road construction, introduction of electricity and drinking water, and other government-induced rural development projects. From 1950 to 1990, the population tripled in size, mainly due to a rapid increase in the number of small settlements of between 100 and 2,500 inhabitants. Both this factor, and an individualization of communal lands, are considered as the main driving forces behind the process of fragmentation of the landscape, to the extreme of what can be observed today (Ochoa-Gaona and González-Espinosa, 2000).

The Highlands encompass all or parts of 30 *municipios* (local governmental units). The area contains various forest formations in a complex landscape with high biodiversity resulting from biotope diversification due to interactions among ecological, geological, edaphic, climatological, and anthropogenic factors (Breedlove 1981). The most important ecosystems include pine, pine-oak, oak, and evergreen cloud forests (Breedlove 1981; González-Espinosa et al., 1995b). These forests are broadly representative of the highlands of southern Mexico, Guatemala, Honduras, and northern Nicaragua.

Only a few decades ago large areas of the Highlands were still covered by old-growth forest, whereas the current land cover represents a highly disturbed landscape with plots of 0.5 to 2 ha of cultivated land, tree and shrub fallow, temporary and permanent grazing lands, and highly disturbed forests (Parra-Vázquez et al., 1989; González-Espinosa et al., 1991; De Jong et al., 1999; Ochoa-Gaona and González-Espinosa, 2000). The structure and composition of the remaining forests have been altered due to (i) selective harvesting of pine trees for local timber production, and oak trees for fuelwood and charcoal (De Jong and Montoya-Gómez 1994; González-Espinosa et al., 1995a, b), and (ii) extensive grazing (De Jong and Montoya-Gómez 1994). Currently, only a few small patches of old-growth forest remain.

The land-use changes have numerous implications for the carbon cycle because each land use has a particular carbon density and functions at a particular speed of carbon accumulation, export, and oxidation (Lugo and Brown, 1992). Biomass decreases as a result of land-use change are expected to occur fast due to rapid removal through fire or other mechanisms, while biomass increase occurs more slowly (rate of increase depends primarily upon the type of previous disturbance) from cultivated land and/or grazing land to tree or shrub fallow, secondary forest and old-growth forest, due to relatively slow growth.

The indigenous population depends highly on the integrated use of the available natural resources to satisfy their domestic and economic needs. Traditional land-use systems are dynamic, in which land is cultivated for one to three years for corn and bean production, followed by a short fallow period. During the fallow, grazing by animals frequently occurs, whereby the manure is used as soil fertilizer. Subsequent shrub and/or tree fallow restores to some degree the depleted soil fertility through a rapid re-establishment of agricultural soil organic matter content (Van der Wal, 1999). Forest product extraction to satisfy domestic needs is widely practiced and includes collection of fuelwood, food crops, building material, litter for crop fertilization, fodder, medicines, and flowers and leaves for ritual purposes (Soto-Pinto, 1990; De Jong & Ruíz-Díaz, 1997).

Farmer families with extremely small properties practicing semi-subsistence agriculture dominate the rural economy and organization. Each family manages simultaneously various strategies, which are articulated by means of various ecological, technical and social relationships and production systems, such as: (i) the production of corn and beans to sustain the family; (ii) temporal or seasonal off-farm paid labor; and (iii) trading of surplus agricultural or forest products. Any shortage in the outcome of one of these components obliges the family to search for compensation by trying to increase one of the other components. For example, changing coffee plantations into corn-beans fields was one of the farmers´ responses to the low coffee price in the early 1990s, whereas a reversed process could be observed when the prices raised again and more government subsidies became available at the end of the 1990s. However, subsistence corn-and-bean production is always present in the small-farmer enterprise.

The carbon sequestration of a terrestrial system can be expressed as a function of plant productivity, subdivided in leaf material, branches, trunks, and roots, i.e. the processes due to photosynthesis, and decomposition of extracted products, litter, wood, roots and soil organic matter, i.e. the processes due to respiration or other forms of burning. Plant productivity can be further subdivided into turnover rates of leaf material, branches, trunks, and roots, and the increase in each of these pools and the increase in extracted products, necromass, litter, and soil organic matter.

The land uses with the largest potential for biomass accumulation and carbon sequestering are young secondary vegetation and forest plantations with fast growing species (Lugo and Brown, 1992). Forest exploitation could therefore be directed to these successional stages (if necessary using enrichment procedures) and to mixed-species plantation forests. Harvesting of the wood products will maintain the carbon sequestered for at least some 10 to15 years, whereas the growth of remaining secondary forest will rapidly replace the extracted biomass. Such forests therefore are efficient CO_2-pumps, as long as the harvested and processed wood does not burn or rot away quickly.

References

Breedlove, D.E. 1981. Flora of Chiapas, Part 1: Introduction to the Flora of Chiapas. The California Academy of Sciences, San Francisco, CA.

Brown, S. Cannell M.G.R, Kauppi, P., Heuveldop, J., Sathaye J., Singh, N. and Weyers, S., 1995 Establishment and management of forests for mitigation of greenhouse gas emissions. IPCC 1995 Assessment. Cambridge University Press.

Cook, A.G., Janetos, A.C. and Hinds, W.T. 1990. Global effects of tropical Deforestation: Towards an Integrated Perspective. Environmental Conservation 17 (3): 201-212.

De Jong, B.H.J., Montoya-Gómez, G., Nelson, K., Soto-Pinto, L., Taylor J., and Tipper; R. 1995. Community forest management and carbon sequestration: a feasibility study from Chiapas, Mexico. Interciencia 20(6): 409-416.

De Jong, B.H.J. and Ruíz-Díaz, M.de J., 1997. La investigacion forestal y su contribucion al manejo de los bosques en Los Altos de Chiapas. In: Parra-Vasquez, M.R. and Díaz-Hernández, B.M. (Eds.) Los Altos de Chiapas: Agricultura y crisis rural. ECOSUR, San Cristóbal de las Casas, México. 149-165.

De Jong, B.H.J., Cairns, M.A., Haggerty, P.K., Ramírez-Marcial, N., Ochoa-Gaona, S., Mendoza-Vega, J., González-Espinosa, M., and March-Mifsut, I. 1999. Land-use Change and Carbon Flux between the 1970s and 1990s in the Central Highlands of Chiapas, Mexico. Environmental Management 23(3): 373-385.

De Jong, B.H.J., Ochoa Gaona, S., Soto-Pinto, L., Castillo-Santiago, M.A., Montoya-Gómez, G., Tipper, R., and March-Mifsut, I. 1998. Modelling Forestry and Agroforestry Opportunities for Carbon Mitigation at a Landscape Level. In: Nabuurs, G.J., Nuutinen, T.,Bartelink H., and Korhonen, M. (Eds.). Forest Scenario modelling for ecosystem management at landscape level, EFI Proceedings No 19. Pp. 221-237.

Dixon, R., Winjum, J., Andrasko, K., Lee, J. and Schroeder, P., 1994: Integrated systems: assessment of promising agroforest and alternative land-use practices to enhance carbon conservation and sequestration, Climatic Change 30, 1-23.

Dixon, R.K., Andrasko, K.J., Sussman, F.G., Lavinson, M.A., Trexler, M.C., and Vinson, T.S. 1993 Forest Sector Carbon Offset Projects: Near-term Opportunities to Mitigate

Greenhouse Gas Emissions. Water, Air and Soil Pollution 70: 561-577.

Dixon, R.K., Sathaye, J.A., Mayeres, S.P., Masera, O.R., Makarov, A.A., Toure, S., Makundi, W., and Wiel, S., 1996. Greenhouse Gas Mitigation Strategies: Preliminary Results from the U.S. Country Study Program. Ambio 25(1): 26-32.

Foley, G. and Barnard, G. 1984. Farm and Community Forestry. Earthscan, Techn. Rep. 3, IIED, London.

González-Espinosa M., Ramírez-Marcial, N., Quintana-Ascencio, P.F., and Martínez-Icó, M.. 1995b. La utilización de los encinos y la conservación de la biodiversidad en Los Altos de Chiapas. In: Memorias del tercer seminario nacional sobre utilización de encinos, 4-6 de Noviembre de 1992. Reporte Cientifico UANL, Número Especial 15. Linares, Nuevo Leon, Mexico. pp.183-197.

González-Espinosa, M., Ochoa-Gaona, S., Ramírez-Marcial, N., and Quintana-Ascencio, P.F.. 1995a. Current land-use trends and conservation of old-growth forest habitats in the highlands of Chiapas, Mexico. Pages 190-198 in M.H. Wilson and S.A. Sader (eds.) Conservation of Neotropical Migratory Birds in Mexico. Maine Agricultural and Forest Experiment Station, Miscellaneous Publication 727, Orono, ME.

González-Espinosa, M., Quintana-Ascencio, P.F., Ramírez-Marcial, N. y Gaytán-Guzmán, P., 1991. Secondary succession in disturbed Pinus-Quercus forests in the highlands of Chiapas, Mexico. Journal of Vegetation Science 2: 351--360.

Lugo, A.E. and Brown, S., 1992. Tropical forests as sinks of atmospheric carbon. Forest Ecology and Management 54: 239-255.

Montoya, G., De Jong, B., Nelson, K., Soto, L., Farías, P.J., Taylor, J., and Tipper R. 1995. Desarrollo forestal sustentable: captura de carbono en dos zonas Tzeltal y Tojolabal del estado de Chiapas. Cuadernos de Trabajo No.4 Instituto Nacional de Ecología. México, DF pp. 79.

Neftel, A., Moore, E., Oescher, H., and Stauffer, B. 1985. Evidence from Polar Ice Cores for the Increase in Atmosheric CO_2 in the Past Two Centuries. Nature 315: 45-47.

Ochoa-Gaona, S and González-Espinosa, M. 2000. Land-use and deforestation in the Highlands of Chiapas, Mexico. Applied Geography 20: 17-42.

Parra-Vázquez, M.R., Alemán-Santillán, T., Nahed-Toral, J., Mera-Ovando, L.M., López-Mejía, M.L. and López-Meza, A., 1989. El subdesarrollo agrícola en los Altos de Chiapas. Univ. Aut. Chapingo, Dir. de Difusión Cultural, Subdir. de Centros Regionales, México. 405 pp.

Rossignol, M., Rossignol, L., Oldeman, R.A.A., and Benzine-Tizroutine, S. 1998. Struggle of life, or: The natural history of stress and adaptation. Treebook 1, Treemail, Heelsum, The Netherlands.

Scolel Té, 1998. http://www.ed.ac.uk/~ebfr11; http://www.ecosur.mx.

Soto-Pinto, M. L. 1990. Plantas útiles de cuatro comunidades de Chiapas: perspectivas en el uso sostenible de la tierra. Revista Fitotecnia Mexicana 13(2): 149 -168.

Swisher, J. and Masters, G. 1992. A mechanism to reconcile equity and efficiency in global climate protection: international carbon emissions offsets. Ambio 21: 154-159.

Swisher, J.N. 1991. Cost and performance of CO2 storage in forestry projects. Biomass and Biomass and Bioenergy 1(6): 317-328.

Tipper, R. and De Jong, B.H.J. 1998. Quantification and regulation of carbon offsets from forestry: comparison of alternative methodologies, with special reference to Chiapas, Mexico. The Commonwealth Forestry Review 77(3): 219-228.

Tipper, R., De Jong, B.H.J., Ochoa-Gaona, S., Soto-Pinto, M.L., Castillo-Santiago, M.A., Montoya-Gómez, G., and March-Mifsut. I. 1998. Assessment of the Cost of Large Scale Forestry for CO_2 Sequestration: Evidence from Chiapas, Mexico. Report PH12. International Energy Agency Greenhouse Gas R&D Programme. Cheltenham, Glosc. UK. 84 pp.

Trexler, M.C. 1993. Manipulating biotic carbon sources and sinks for climate change mitigation: Can science keep up with practice? Water, Air, and Soil Pollution 70: 579-593.

Van der Wal, H. 1999. Chnantec shifting cultivation: InTERAcTIVE landuse: A case-study in the Chinantla, Mexico, on secondary vegetation, soils and crop performance under indigenous shifting cultivation. Treebook 3, Treemail, Heelsum, The Netherlands.

Winjum, J. K., Dixon, R.K., and Schroeder, P. E. 1992. Estimating the global potential of forest and agroforest management practices to sequester carbon. Water, Air and Soil Pollution 64: 213-222.

2. CONCEPTS

Introduction

Forests are not only impacted upon by climate change, they also influence it. Forests can be serious sources of greenhouse gases due to their destruction or can be important sinks of the same gases by means of their sustainable management. They act as important buffers against the impact of the ongoing climate change. There are three broad categories of forestry interventions that will help to mitigate greenhouse gas emissions: (i) improving existing forest management, (ii) Increasing total tree cover and biomass density, and (iii) using wood as a substitute for energy-intensive products (cement, fossil fuel). Not all interventions are currently considered within the framework of the Kyoto Protocol (UNFCCC, 1997). However, they could make major environmental and socioeconomic contributions to the countries where they are undertaken. Forestry projects that mitigate carbon emissions into the atmosphere have methodological, biophysical, institutional and socio-economic features that need to be defined. The following concepts are adapted from official and unofficial documents of the International Panel on Climate Change, complemented by cited references.

Forest

Forests are usually comprised of many individual stands of trees with differences in architecture and species composition (cf. Oldeman's "silvatic mosaics", 1990). Thus within a forest, there usually is a range of different forested ecosystems composed by different species with different ages and having different carbon stock densities (MgC ha^{-1}). The distinction between the whole forest as a landscape unit and the stand as a unit within a forest is frequently left unstated and the term 'forest' is used for both.

Many definitions of forest are in use throughout the world. Lund (1999, 2000) in a recent review listed over 130 definitions of a forest. Countries have developed very specific definitions suitable for their own environmental and administrative purposes. A comprehensive definition should include the following elements: "An ecosystem covering a minimum area with a minimum width, dominated by trees with a minimum potential height at maturity of 5 m and with a projected crown cover exceeding 10%". Minimum area and width definitions are important for monitoring purposes. If the minimum area is small than the assessment task is greater. If crown cover is averaged over large areas then deforestation may not be detected.

Most monitoring agencies will adopt a minimum area for inclusion that is based on the resolution of their imagery or sampling technique and the resources available. A commonly used attribute in the definition of a forest is the Canopy (or Crown) Projected Cover (CPC), which is the proportion of the area covered by tree crowns. It is estimated by vertically projecting outlines of tree crowns onto the horizontal plane that represents the forest area. Minimum CPC is the most common of the quantitative elements included in a definition of forests. In definitions collected by Lund (1999) the minimum CPC to be included as a forest varied from 10% to 70%. Forest may consist either of ***closed forest*** formations where tree crowns of various heights and undergrowth together cover a high proportion of the ground (often defined as more than 40%); or ***open forest*** formations with a continuous vegetation cover in which tree crown cover exceeds 10% (in our case less than 40%). Young natural stands and all plantations established for forestry purposes which have yet to reach a crown density of 10 percent or tree height of 5 m are included in the definition of forest, as are areas temporarily without trees as a result of human intervention or natural causes, but which are expected to revert to forest soon. In other words, all development phases of a forest area are included.

Land use, land cover, and land-use change

Land-use change in the context of greenhouse gas fluxes implies human activities which:

1) change the way land is used, and/or

2) affect the amount of biomass in existing stocks

There seems to be some confusion in the use of the term "land use and land-use change" as many of the changes in forest stocks are not or barely associated with a change in land use. For example, commercial harvest of industrial roundwood (logs) and fuelwood is a land use but not a land-use change.

The terms land cover and land use are often used indiscriminately. According to the FAO (1997) land cover is "the observed physical and biological cover of the earth's land, as vegetation or man-made features", whereas land use is "the total of arrangements, activities and inputs that people undertake in a certain land cover type". The term land utilization type is also used by FAO and refers to synthetic, simplified and representative land-use types for the purposes of land suitability evaluation, rather than examples of real land uses. Lund (1999) rightly states that from a carbon storage perspective, it would be appropriate to focus on land cover, land-cover change and the land-management activities that affect the type, amount, extent and condition of all woody vegetation. Changes in the amount, type and extent of land cover are directly correlated with changes in biomass, whereas the same cannot be said of changes in land use. When land use changes, the biomass may remain the same. In addition, saying that an area has been deforested or reforested just because the land use has changed can be very misleading.

Deforestation

The majority of the definitions collected by Lund (1999) define deforestation as the long-term or permanent removal of forest cover and a conversion to a non-forested land use or to wasteland. For example, deforestation is the "permanent removal of forest cover and withdrawal of land from forest use, whether deliberately or circumstantially". Similarly the IPCC Guidelines emphasize the conversion of forests: "Conversion of forests is also referred to as "deforestation" and it is frequently accompanied by burning" (IPCC, 1996). "Clear cutting (even with stump removal), if shortly followed by reforestation, is not deforesting" (FAO, 1997, cited in Lund 1999). Many definitions of deforestation include natural events such as landslides, hurricanes, etc. Even if the cause of forest fire can be unambiguously attributed to prescribed burning or escape from deliberate fires, the event may or may not be deforestation. In many instances the forest cover loss is not complete and furthermore is followed by rapid regrowth. In some forest types (e.g. eucalypt and some pine forests), even the most intense fires do not cause significant mortality of mature trees. Significant quantities of carbon may be released but most trunks and large branches survive and reestablish a closed forest within a few years. When the regrowth is complete, the net release of carbon to the atmosphere is zero or near zero. However, the rapid release to the atmosphere is followed by a slow recapture and can thus be considered as a temporal release of CO_2 to the atmosphere (Kurz et al., 1993). In the long-term and over large areas, there is a balance between carbon loss due to fires and carbon sequestered in regrowth providing there is no change in the fire regime. This is the basis of the IPCC (1996) recommendation to treat wildfires as neutral. While this is a pragmatic approach, there is some evidence that fire regimes are changing in many parts of the world (Kurz & Apps 1999). In some cases these changes are due to solar impacts on the climate system, like El Niño with its droughts (Rossignol et al., 1998) or to deliberate human manipulation. In still others it is the indirect consequence of human actions, also including possibly climate change. Furthermore, in many parts of the world, major forest fires are followed by land-use change from forest to agricultural or pastoral uses.

Forest degradation an restoration

The term forest degradation is sometimes used in forestry but its definition is often not well suited in the context of carbon dynamics as it often only refers to changes in the productive capacity of the forest. For example, "Changes within the forest class which negatively affect the stand or site and lower the production capacity" (FAO, 1990). The best-known international scheme and provisional geographic database to classify degradation is the Global Assessment of Soil Degradation (GLASOD, 1998) prepared for UNEP with the participation of many national institutions. Distinguished are: water and wind erosion, and chemical and physical deterioration The World Overview of Soil Conservation Approaches and Technologies (WOCAT, 2000) gives spatial information on human-engendered soil and land improvements for more sustainable agricultural land use, and the technologies applied. Restoration of soils depleted in organic-matter to their original level looks like a realistic option of carbon sequestration (Lal et al., 1998, Paustian et al., 1998). Improvement of soils beyond the original level of soil organic matter, as a form of land and soil aggradation, would appear possible as well (Sombroek et al., 1993).

Land-Use Change and Management

Most of the world's land area is being managed for human habitat and infrastructure, food and wood products, recreation, and ecosystem preservation. Management of croplands, forests and pastures does affect sources and sinks of CO_2, CH_4 and N_2O. Changes in land use and management can cause significant changes in carbon storage and flows. These are, however, not easy to assess. Deforestation and wood harvesting produce carbon dioxide rapidly due to decay of leaf litter, or much more slowly due to decay of wood, wood products and soil organic matter (Sundquist, 1993). After clearance, logging or abandonment of agricultural land or pasture, on the other hand, forest regrowth fixes carbon. A carbon budget is thus required to account for shifting spatial patterns of land use and also the temporal fate of vegetation, soils, debris and wood products. When tropical forests are cleared for conversion to agriculture or pasture, a large portion of the aboveground biomass is generally burnt, thus releasing its carbon rapidly into the atmosphere. Some of the wood may be reserved for other end uses and sometimes thereby preserved for a longer time, though eventually it still decays.. Deforestation also speeds up the decay of other aboveground material such as dead wood and litter, as it does in below ground organic carbon. Local climate determines the rates of decay; in tropical moist regions most of the remaining biomass decomposes in some 10 years (Vooren, 1999). Deforestation for shifting cultivation releases less carbon than permanent deforestation because the fallow crop rotations maintain the total ecosystem carbon at a higher level (Van der Wal, 1999). Carbon fluxes depend on forest type, length of fallow, and frequency of fallowing, which vary across regions.

Logging releases carbon to the atmosphere due to decay of harvest waste and damaged or killed trees. The harvested wood decays at rates dependent on their end use: e.g. fuelwood in 1 year, paper in less than 10 years and construction material in decades. The logged forest will gradually compensate for the decay created during harvest, as it regrows. Forest harvesting also leads to changes in soil carbon dynamics, depending on what happens during and after harvesting. These changes are more difficult to estimate. Soil may loose carbon after wood harvesting, due to soil destructuration caused by tree uprootings (Koop, 1989). Uprooting increases aeration and soil temperatures (Tisdall and Oades, 1982, Elliott, 1986), making soil aggregates more susceptible to breakdown and physically protected organic material more susceptible to decomposition (Elliott, 1986, Beare et al., 1994). Losses due to leaching of soluble organic carbon occur in many soils; although this is seldom a dominant carbon flux in soils, it contributes to the lateral transport of carbon from the terrestrial to marine environment by runoff (Sarmiento and Sundquist, 1992, Meybeck, 1982). In the tropics the recovery to original soil carbon levels after reforestation is quite rapid (Oldeman, 1983).

Tree plantations on recent deforested land may begin with net carbon emissions due to decomposition of the remnant biomass and soil. The plantation begins to fix carbon after some years at rates dependent on site conditions and species grown. The trees and below-ground root biomass, humus and soil accumulate carbon during later stages of the plantation development.

For about two centuries prior to 1950, the high and mid latitudes may have released substantial amounts of carbon because of wood harvesting, deforestation and conversion of forests to agricultural use, but this has since reversed as deforestation has stopped and many forests at present seem to be in a stage of regeneration and regrowth (Kauppi et al., 1992). The low latitudes comprising the tropical belts, on the other hand, have been experiencing high rates of deforestation in the last decades (IPCC, 1996). The considerable variation in vegetation C density of the vegetation in the low latitudes, however, introduces much uncertainty in the estimates of C fluxes as a result of human land-use changes. The estimate of global net tropical emissions of $1.6 * 10^6$ GgC yr^{-1} during the 1980s (Houghton 1994, Dixon et al., 1994) may have been on the high side judging from new data from the Brazilian Amazon and the rest of the world (Schimel, 1995; Phillips et al., 1998).

Land-use change requires human decisions and action, but in some cases the decisions and action may be largely driven by acts of nature. For example, landslips or flood damage may destroy a patch of forest permanently as far as planning terms go. If a decision is made to convert the area to cropping or grazing, it could be argued that humans did not cause this completely and/or directly.

Carbon Pools

For a full accounting of carbon at a site, one needs to examine the aboveground and below-ground carbon pools as parts of a dynamic multi-component system, in which variable bi-directional fluxes between the pools occur (Figures 2.1 and 2.2). The aboveground pools comprise all woody stems, branches and leaves of the living trees, creepers, climbers and epiphytes, the herbaceous undergrowth, as well as the fallen trees and the litter layer, all expressed in oven-dried mass unit per area unit (e.g. Mg ha^{-1}). The belowground biomass comprises the living and dead roots, the soil fauna, the microbial community, and the soil humus in its various forms.

Aboveground plant biomass:

Organisms	Organs
Trees	Leaves
Shrubs	Branches
Herbs	Stems
Lianas	
Epiphytes	

COMPARTMENTS

Soil
- Litter } Organic layer } Humus
- O horizon } Organic layer } Humus
- A_h horizon } Humus
- Mineral soil

in the soil:
Roots
Soil organic carbon
Dissolved organic carbon

Figure 2.1. *Schematic diagram of the vegetative carbon compartments of a terrestrial ecosystem.*

Carbon Flows

From the perspective of carbon sequestration by stands of vegetation alone, we are concerned with the overall Net Ecosystem Productivity (Table 2.2). However, to understand the magnitude of the Net Ecosystem Productivity over a specified period of time in a particular forest development phase or successional sere, and to determine its potential degree of dynamics, the principal processes within the ecosystem need to be understood, as well as those due to impacts from outside the ecosystem.

A mass balance of the aboveground component fluxes enables the estimation of the amount of carbon accumulated aboveground and transported internally to belowground pools. A mass balance of the compartment fluxes belowground enables the estimation of the net changes in the persistent pool of soil carbon and carbon emitted to the atmosphere (Figure 2.2). On the one hand, photosynthesis, which drives the carbon accumulation process, depends on various factors, such as solar radiation and atmospheric CO_2 concentration. On the other hand, all respiratory processes are very sensitive to heat, particularly the fine roots, the mycorrhizal fungi, and the heterotrophs in the soil (Smits 1994, Rayment and Jarvis 1999).

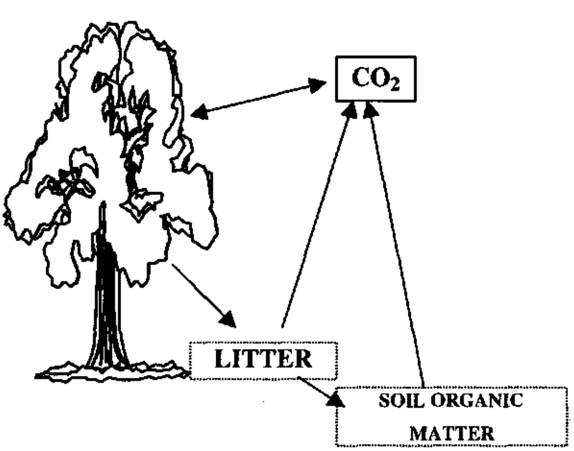


Figure 2.2. *Highly simplified diagram of carbon flows between aboveground and belowground vegetative biomass, and the atmosphere.*

Dynamics of the Carbon Balance over Different Time Scales

In assessing the carbon flow of terrestrial ecosystems now and in the future, it is essential to consider the time scale over which the carbon flux is measured or estimated (Table 2.1):

Table 2.1. *Ecosystem productivity measures and loss processes, quantity of loss, and the time scales of the different measures and processes (after Schulze and Heiman, 1998).*

Productivity	Carbon loss process	Quantity	Time scale
Gross primary productivity		100%	Very short term (days)
	Plant respiration	*50%*	
Net primary productivity		50%	Short term (days to years)
	Microbial respiration, disturbance by directional forcing (CO_2 or temperature rise)	*45%*	
Net ecosystem productivity		5%	Medium term (decades)
	Reduction by strong, rare impacts (fire, harvest)	*4-5%*	
Net biome productivity	(asymptote is zero)	<0.5%	Long-term (centuries)

Daily Fluxes: Carbon fluxes in an ecosystem are closely correlated with solar radiation patterns over the course of a day, with net carbon accumulation in the system during the daytime when solar insulation triggers photosynthesis, and carbon emissions at very low light levels or obscurity, due to plant respiration and biomass decomposition. Depending on the net balance between inward and outward fluxes, for any vegetation there may be a net gain or loss over any one day during a year (Jarvis et al., 1997). In tropical rain forests, the daily drought approximately between 10.00 h and 16.00 h causes the stomata to close and trees to make a "metabolic siesta" (Salager et al., 1990).

Seasonal Fluxes: Over a year, the length of the growing season, or in the case of evergreen vegetation, the photosynthetic season has a major influence on carbon flux. In tropical forests carbon gain is nearly continuous throughout the year, being reduced by occasional periods of stress, e.g. relative cold or persistent cloud cover (Malhi et al., 1999).

Annual Fluxes: The issue of annual dynamics is particularly relevant to forest systems because of the long-term variation in carbon flows. The carbon flux of a forest stand after a natural or human-induced perturbation is dependent on the development stage of the forest at the time of the perturbation and the degree of perturbation. A measurement of carbon flux made for a short period somewhere within the whole succession cycle, may give a misleading picture of both the ongoing fluxes and the long-term carbon accumulation of the system.

Lateral flows of carbon

There are a number of lateral flows of carbon that can occur at the land surface or in the soil. A flow of material that is going away from one place can be deposited elsewhere: (i) wind erosion carries dust elsewhere, sometimes several hundreds of kilometers away; (ii) water erosion in upper catchments deposits material downstream; (iii) continuous percolation and subsequent lateral conduction conveys material due to groundwater flows.

Soils

Fully developed soils are natural bodies with a vertical sequence of horizons: litter, one or more layers of largely decomposed organic materials, a mineral surface horizon, a subsurface mineral horizon, a subsurface mineral horizon with features of accumulation, a mineral horizon consisting of sediment or unconsolidated bedrock, and locally hard bedrock. Some horizons may be lacking, or modified by for example gleying due to ground or stagnant water. Agricultural soils may have some mixed horizons up to about 30 cm, resulting from plowing or other agricultural activities. Some deeper horizons may have been broken up, too, by one-time deep plowing. The thick organic layer in wetlands can be considered as a special horizon, which may have accumulated organic material of some 30 cm up to several meters. They are an important store of organic carbon, which may be released as CO_2 and/or CH_4 if the land is drained and cultivated, or may be subject to wildfires in dry years. Both the topsoil and the subsoil are relevant in the context of carbon sequestration. The topsoil is the layer with accumulation of more labile carbon in some form of soil organic matter. The subsoil is the one with more inert carbon in some form of stable humus. Soil fauna plays an important role in the dynamic interaction between these two layers.

Soil Organic Matter

Soil Organic Matter (SOM) is a generic term for all organic compounds in the soil that are not living roots. SOM has been defined in various ways: on origin, on transformation stage, function, solubility, chemical constituents, elemental carbon to nitrogen ratio (C/N), cation exchange capacity (CEC), or on dynamics and stability. When we look at the stability of the SOM, its dynamics and residence or turnover time, one can distinguish inert, stable and labile pools (Anderson and Ingram, 1993). Labile SOM largely consists of soil microorganisms and their immediate products, while stable SOM are the formations of polymeric substances. None of the current definitions and subdivisions of SOM provide fully quantifiable and universally accepted parameters to define SOM quality and quantity, as needed to measure carbon sequestration in soils. For accounting and modelling purposes, a simplification to inert and labile carbon and possibly carbon to nitrogen ratios today is the only way to estimate changes in time and according to vegetation dynamics and soil clay content (Körschens, 1998).

Measurement Methods

The amount of carbon held in terrestrial ecosystems varies both spatially and temporally as a result of natural variations and processes, and of human-induced activities. Methods exist for measuring the amount of carbon and for measuring changes of this amount. However, the key issue is whether or not the methods of measuring losses and accumulations of carbon are consistent and comparable, as they vary in complexity, precision, accuracy, and cost. Distinct methods are used for individual pools and components of terrestrial carbon and for temporal and spatial scales. The most direct methods are those that measure all carbon or all change in carbon content. Such methods are limited to small areas. The carbon in aboveground plant biomass may be measured directly on plots up to about one hectare. In soils, such measurements are limited to plots as large as about one square meter (Chapter 3). Different methods are used to extrapolate these measurements to larger areas. The use of satellite data to measure changes in the area of forests may be calibrated with ground measurements (cf. Supit 2000). The use of ecosystem models or the use of atmospheric data and models, to infer changes in terrestrial carbon are more indirect and often less robust methods.

Stocks or Fluxes

Flows of carbon on land can be measured by means of assessing temporal differences in stocks of carbon and direct flux measurements. Measurement of stocks at the beginning and the end of a project will estimate the change in stocks that has occurred during this period, without considering intermediate variation. Measuring the flux of carbon into or out the system over the same period will also estimate the net change, but gives also insight in temporal variations (Jarvis et al., 1997). Changes in aboveground biomass may be measured directly with the stock-change method, whereas changes in belowground carbon may be more readily obtained by measuring CO_2 fluxes. If CO_2 fluxes are measured, these should be converted to C value, as one kilogram of CO_2 only contains approximately 0.27 kilograms of C. The changes in terrestrial carbon storage as a result of mayor land-use or land-cover change have been measured directly by assigning stock densities to each unit of land cover and calculating the net change in land cover statistics (Chapter 3; Chapter 5). The difficulty is in attributing changes in land cover to particular causes, as measured changes in land use do not reveal the cause of the change.

Measurement of Areas

Stocks of carbon, changes in stocks, and fluxes of carbon, once they have been determined for small plots, must be apt for extrapolation to regions and countries. Remote sensing is useful for determining areas for land-cover types with different attributes (for example, different stocks of carbon or rates of CO_2 exchange). Satellite data are successful in distinguishing forest cover, deforested areas, and areas of secondary, or regrowing forests. When the satellite images are used on a temporal scale, dynamic patterns of land transformation can be detected (Chapter 8; Ochoa-Gaona and González-Espinosa, 2000).

Accounting Systems

Carbon accounting systems record, summarize, and report the quantity of carbon emitted or sequestered by the application of land-cover change and forestry activities over a specific lapse of time. The accounting system quantitatively demonstrates to what extent a certain activity contributes toward greenhouse gas mitigation. An accounting system should incorporate principles of consistency, transparency, verifiability, and efficiency.

Consistency

Consistency refers to the system's adherence to the concepts of carbon sequestration in space (all relevant sources and sinks should be included) and in time (e.g., a steady state forest should correspond to a zero net change in stock). The objective of the accounting system is to demonstrate compliance by land-use practices with the goals of greenhouse gas mitigation.

Transparency

The logic of the accounting system should be clear and deductive, that is the reported information can be traced back to the underlying data by means of a logical set of procedures. For example, reported carbon fluxes may be estimated by a measurement method that accounts for all relevant pools; in turn, the measurement method is applied to data that represent carbon changes pool by pool (Chapter 3; Chapter 5).

Verifiability
The accounting system should be built on proper data collection, measurement, processing and reporting procedures. Claims of a given quantity of carbon mitigated over a certain period of time should be based on conceptually sound data, models, and methods being both available and feasible.

Efficiency
An efficient accounting system is one that operates at the point where the marginal costs of increased accuracy and precision just equals the marginal benefits of achieving the improvements. The most effective accounting system should be used with the limited resources that are available (Chapter 4).

Baselines

Baselines are needed to quantify the emissions and their reductions or offsets achieved by a greenhouse gas mitigation project. Baselines are also referred to as 'reference cases' (Tipper and De Jong, 1998). The basic concept is that the quantity of a greenhouse gas mitigation offset is the difference between the greenhouse gas fluxes that would occur in the cases that the project would either be implemented or not. Broadly defined, a project's baseline is the collective set of economic, financial, regulatory and political conditions, within which the project will operate during its lifetime. The calculation of any reference scenario should take into account likely changes in relevant regulations and laws, market developments, and economic and political trends in order to define either a single baseline of medium probability or a range of baseline scenarios.

There are three fundamental approaches to setting baselines. In the first, historical data sets are used to project trends forward into the future based on the best available information regarding government policies and changing economic, social, and physical conditions (Chapter 5; Tipper and De Jong, 1998; Chapter 6). Within this set of data the changes, which the project will bring about, will be defined and the impact on greenhouse gases estimated.

An alternative approach is to develop performance standards or benchmarks for project types, adjusting the standards to local conditions and updating them regularly as methodological improvements are made or additional data become available. The potential benefits of adopting objective international standards are that they would make it easier for local authorities to define project additionality, and thus to estimate emission reductions over the life of the project. Certain activities could be considered standard good management practice, and baselines might be set to reflect the level of carbon sequestration that would occur if these practices were applied.

1984). Estimated deforestation rates in the Chiapas highlands ranged from 3.23% (1974-1984) to 3.58% (1984-1990), for only closed forests, and from 1.58% (1974 -1984) to 2.13% (1984-1990) for open and closed forests combined (Ochoa-Gaona and González-Espinosa 1998). The main causes of forest conversions are agropastoral land creation, tree harvesting, and wildfires (Masera et al., 1992; De Jong and Montoya-Gómez 1994). Driving forces behind the process have been governmental incentives for agricultural development, population growth, shifts from subsistence to commercial production systems, infrastructure development, and insecure land and tree tenure systems (De Jong and Montoya-Gómez 1994).

Lack of accurate data on deforestation and forest degradation, and the associated biomass reduction, hinders the development of C models for region-specific land-use patterns. Research on C emissions from land-use change and forestry in Mexico has thus been identified as a high priority in order to better understand the role of biomass and land-cover types in the overall national C balance (Masera et al., 1992).

In this paper we present results of a case study in an intensively impacted and highly fragmented landscape in which we apply field-measured C-density values to land use/land cover (LU/LC) statistics. Our approach for estimating the flux of C between terrestrial ecosystems and the atmosphere was to assign C densities to LU/LC classes and to thus calculate net C flux due to changes in LU/LC areas derived from land-use and forest inventories conducted by Mexican governmental organizations.

Methods

Study area

The highlands of Chiapas (Figure 3.1), located at 1500-2900 m elevation in southeast Mexico encompass all or parts of 30 *municipios* (local governmental units akin to U.S. counties). The area contains various forest formations in a complex landscape with high biodiversity resulting from interactions among geological, edaphic, climatological, and anthropogenic factors (Breedlove 1981). The most important ecosystems include pine, pine-oak, oak, and evergreen cloud forests (Breedlove 1981; González-Espinosa et al., 1995b). These forests are broadly representative of the highlands of southern Mexico, Guatemala, Honduras, and northern Nicaragua. The regional climate is subtropical to temperate sub-humid, with summer rains and winter droughts. Mean annual rainfall varies between 1100-2000 mm. Life zones in the study area include tropical lower montane and premontane moist and subtropical lower montane and montane wet zones (Holdridge 1967).

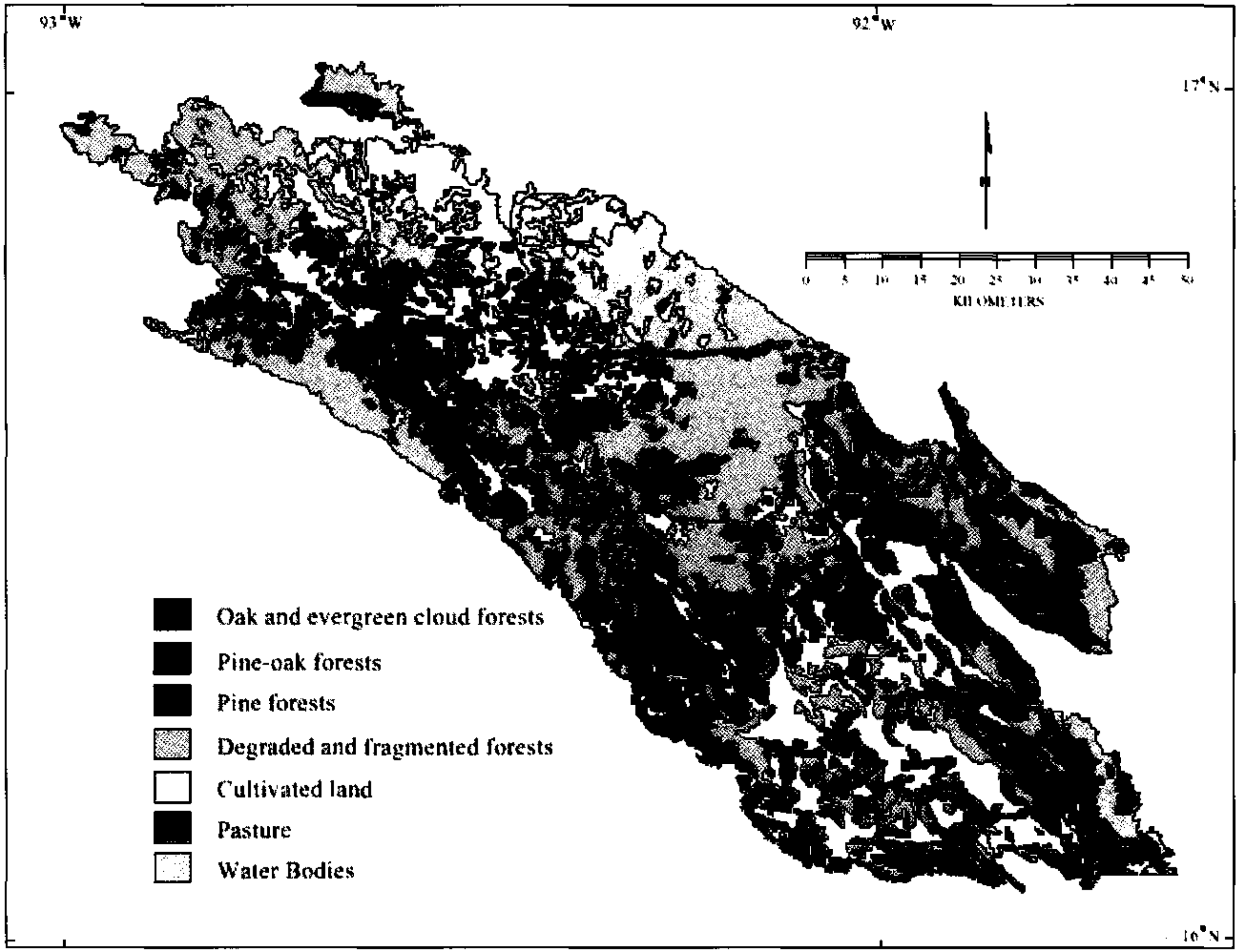


Figure 2. Land use/land cover in the Chiapas highlands, Mex ico, circa 1975.

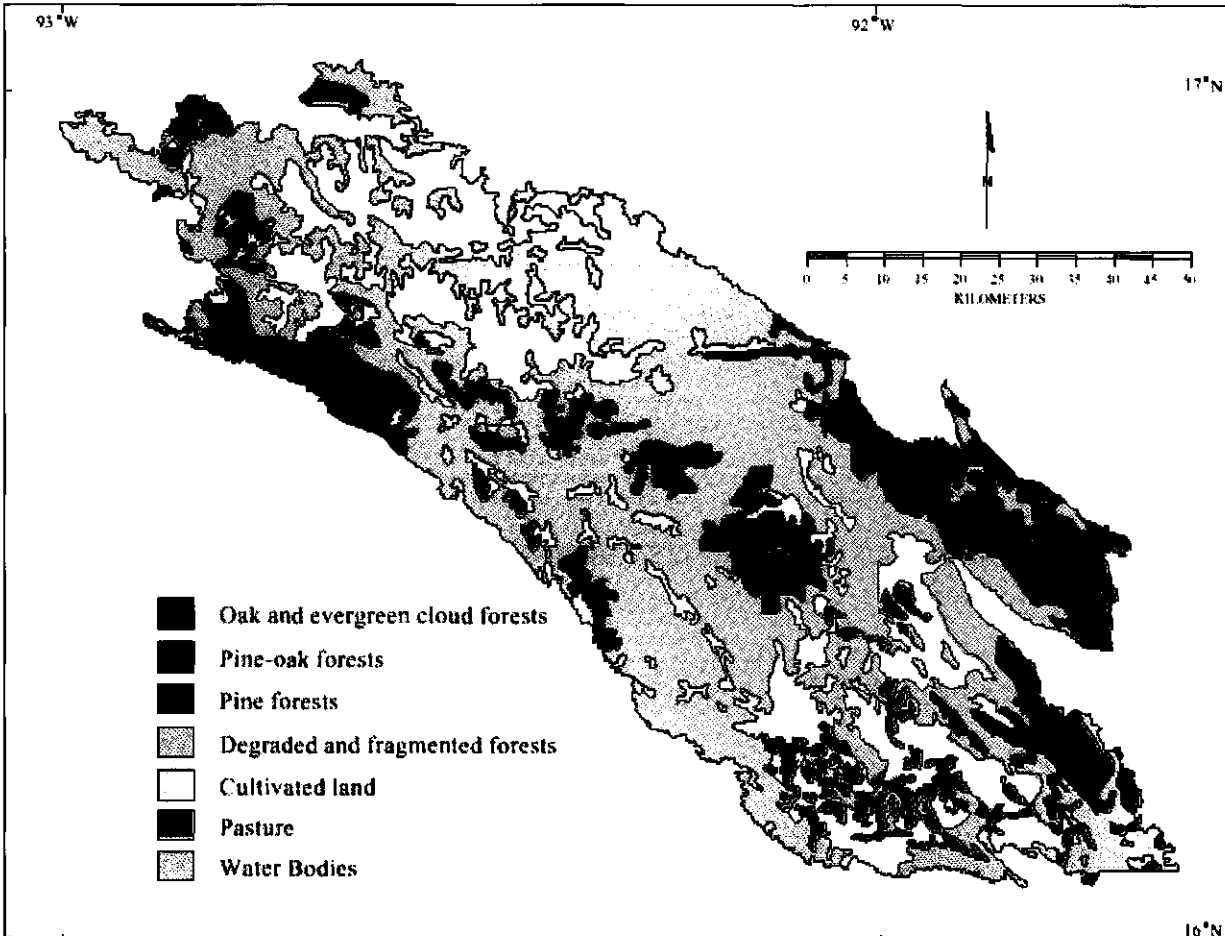


Figure 3. Land use/land cover in the Chiapas highlands, Mex ico, circa 1991.

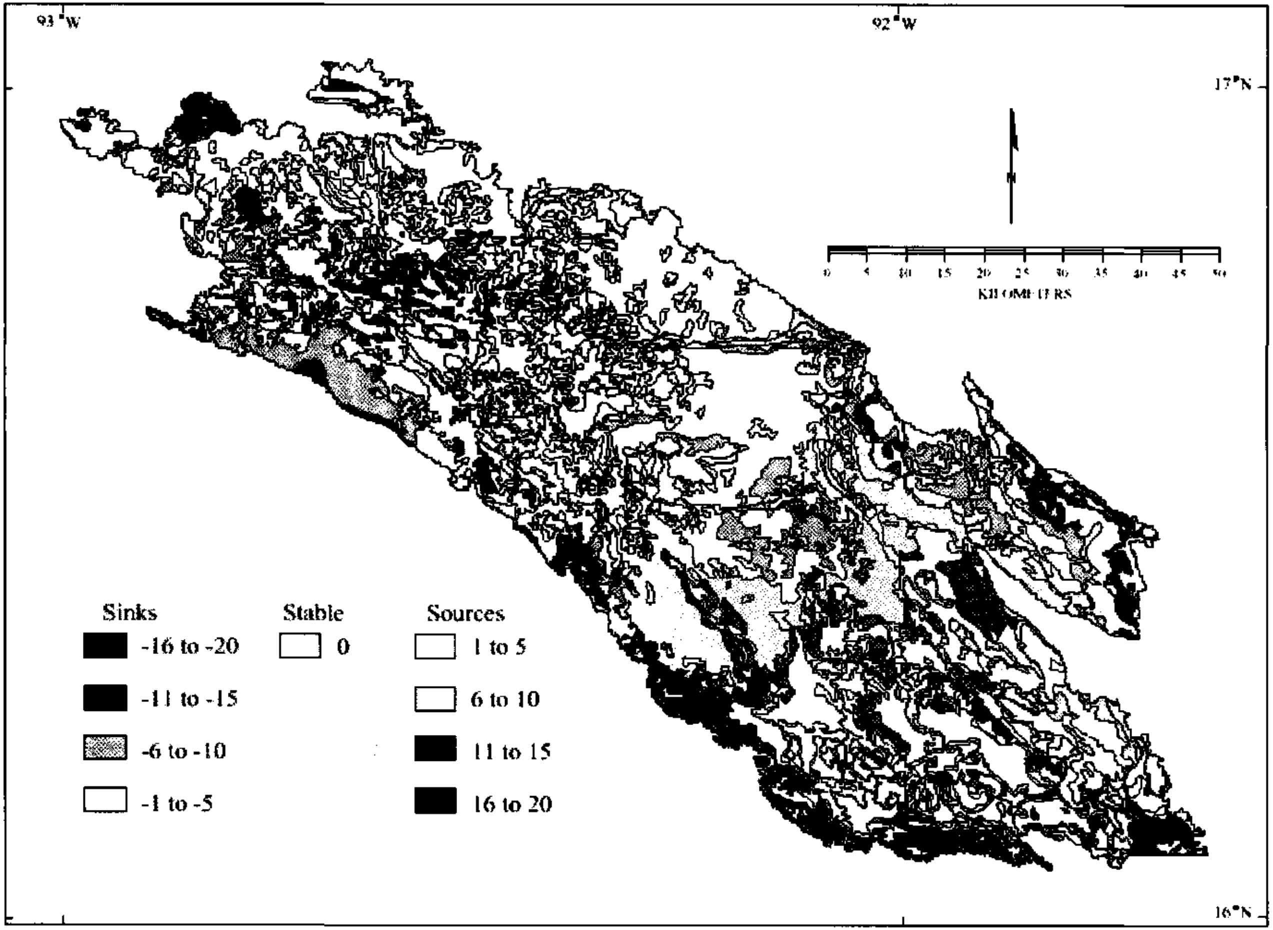


Figure 4. Estimated mean carbon flux (MgC ha^{-1} yr^{-1}) between ca. 1975 and ca. 1991. This map is for demonstration purposes and was not used for carbon pool and flux analyses. Positive values indicate fluxes from the terrestial biosphere; negative values indicate fluxes from the atmosphere.

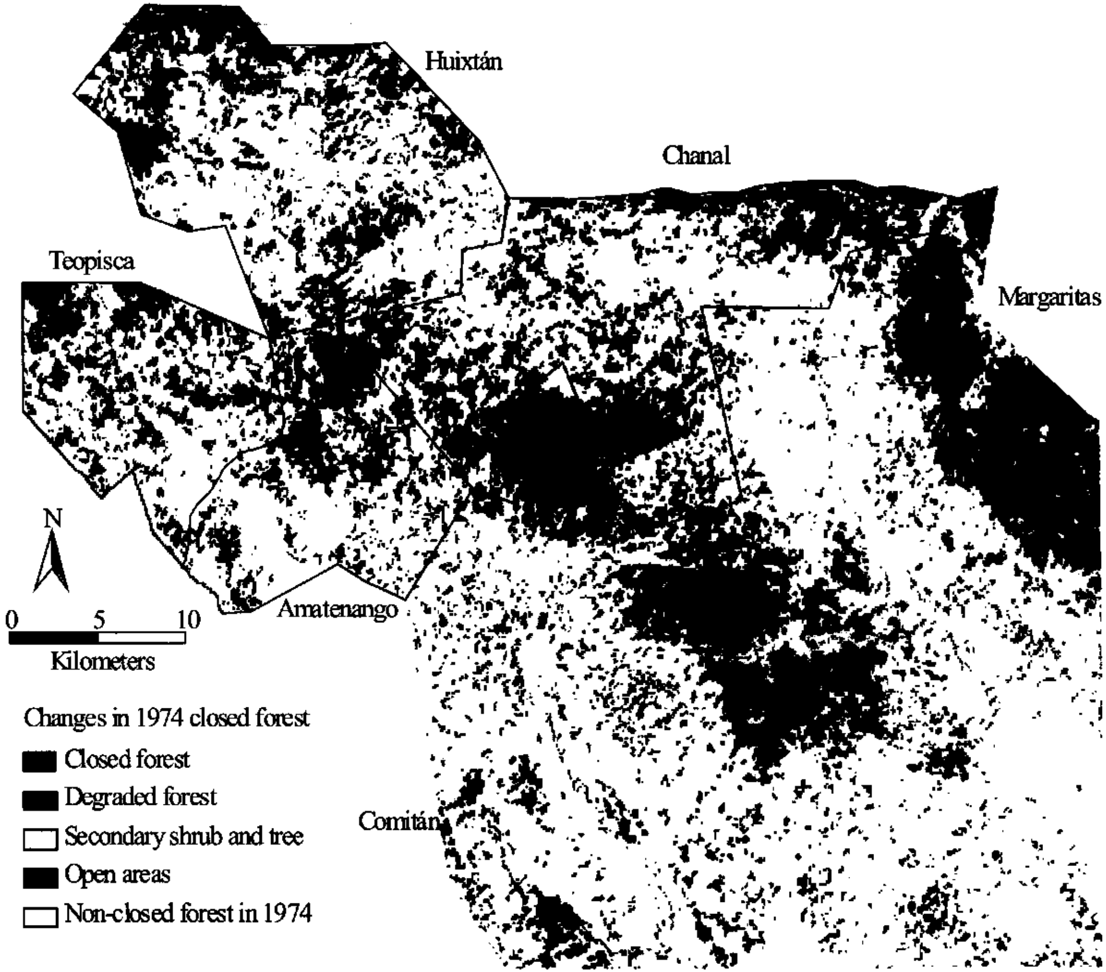


Figure 8.3. Spatial variation of the change in closed forest from 1974 to 1996. Closed forest include oak, pine-oak, and pine forest; degraded forest include open pine and disturbed pine-oak forest; open areas include agricultural land, pasture and settlements; non-closed forest in 1974 include shrub and tree fallow, open pine forest, and open areas.

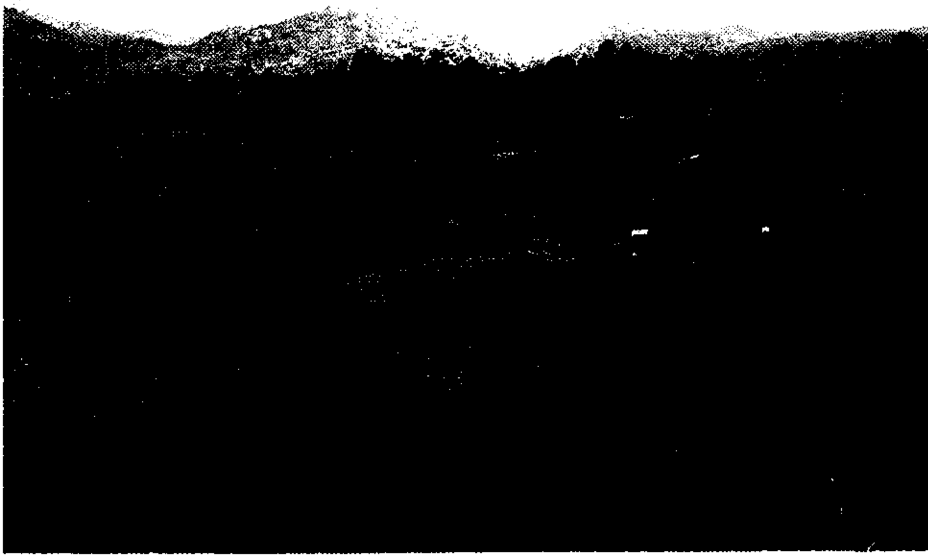


Photo 1. A general overview of a highly fragmented landscape with sharply defined borders between patches of agricultural land, grassland, and shrub and tree fallow.

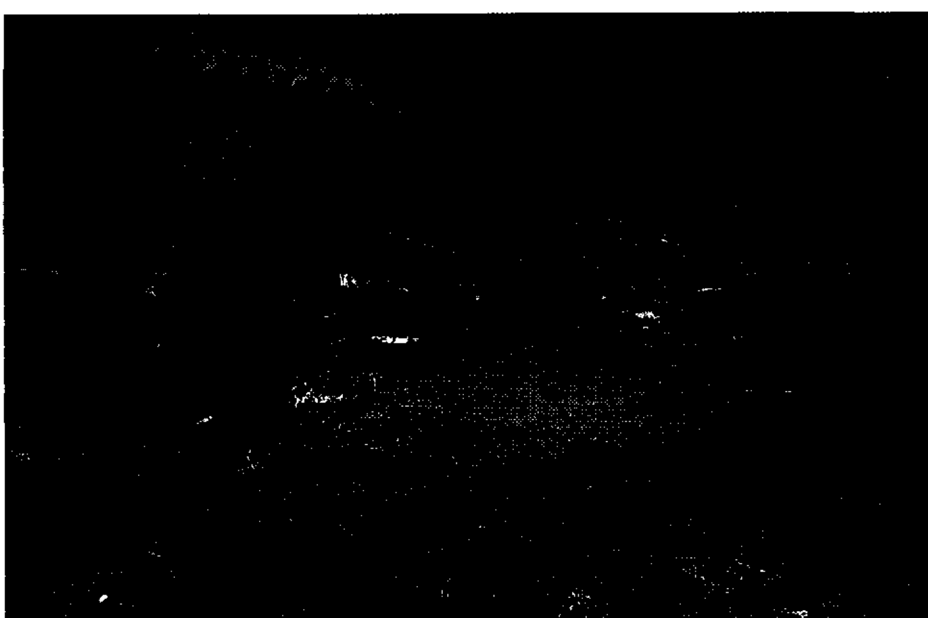


Photo 2. An overview of fragments with fuzzy borders between closed forest at the left, open forest on the right, and open grassland in front.

Photo 3. An overview of an area under traditional slash-and-burn management, with plots of fallow in various stages of succession, mixed with cropland.

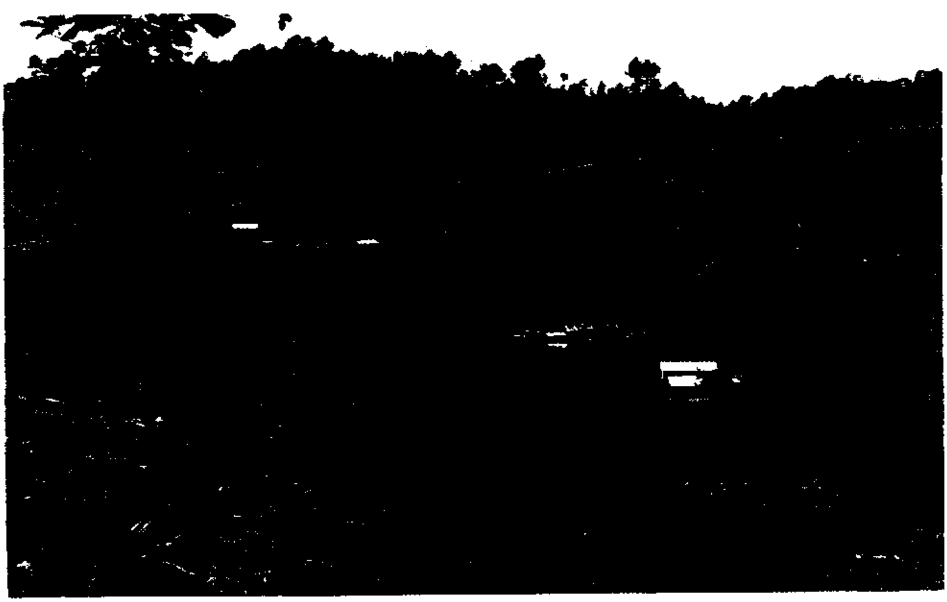


Photo 4. An overview of a highly fragmented landscape with the tree element almost removed completely. Note the various degrees of erosion at the upper part of the hillside, where reforestation is currently attempted.

Photo 5. Cattle grazing in closed pine-oak forest. Note the scarce understory vegetation.

Photo 6. Fuelwood production resulting from the conversion of a tree fallow to cropland.

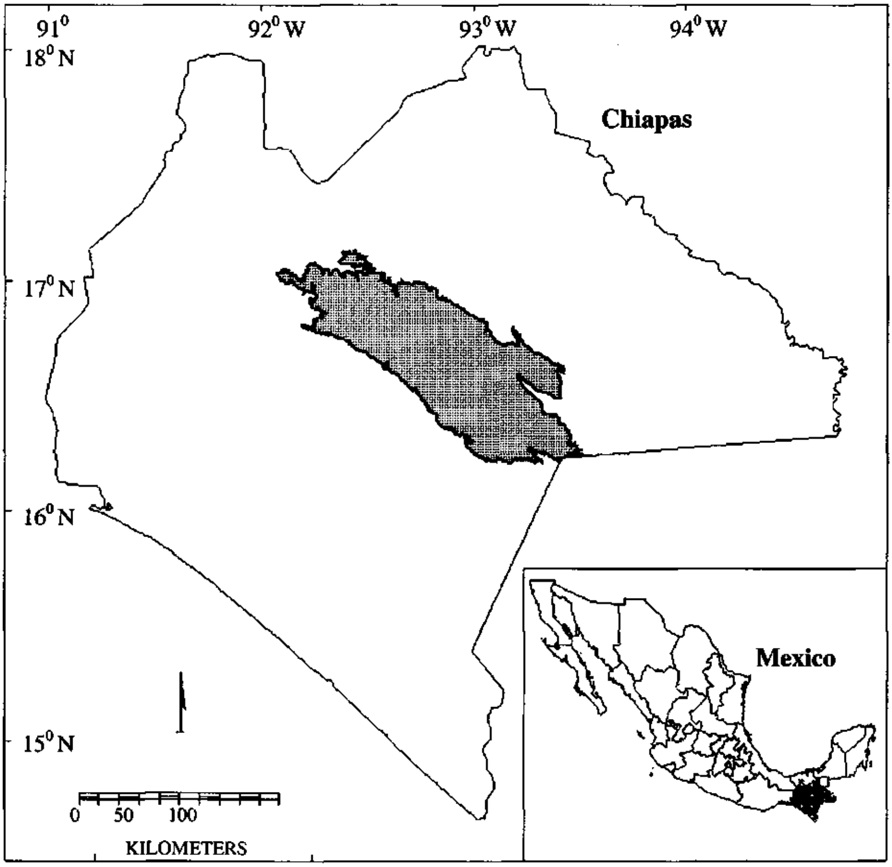


Figure 3.1. *Chiapas highlands study area.*

The soils are predominantly dark brown, clayey loam, derived from calcareous rocks (Parra-Vázquez et al., 1989). Soil pH varies between 4.6 and 7.3.

Recent changes in land-use patterns have been both extensive and intensive. Only a few decades ago large areas were still covered by old-growth forest, while the current land cover represents a highly disturbed landscape which includes plots of 0.5-2 ha of cultivated land, secondary forests (or tree and shrub fallow), temporary and permanent grazing lands, and disturbed forests (Parra-Vázquez et al., 1989; González-Espinosa et al., 1991). The structure and composition of the remaining forests have been altered due to (i) selective harvest of pine trees for local timber production, and of oak trees for fuelwood and charcoal (De Jong and Montoya-Gómez 1994; González-Espinosa et al., 1995a, b), and (ii) extensive sheep grazing

(De Jong and Montoya-Gómez 1994). Currently, only a few small patches of old-growth forest remain.

Map data

The data used in this study were derived from two 1:250 000 scale map series bounding the period between the mid- 1970s and the early 1990s, a time of dynamic land use change. The first series is designated *Carta de Uso del Suelo y Vegetación* (Land Use and Vegetation Maps). For the Chiapas highlands coverage we used three individual LU/LC maps generated from aerial photography covering the years 1972-1982 at scales ranging from 1:50,000 to 1:80,000. Classifications were verified in the field between 1980 and 1985. Individual maps were published by the Mexican federal government agency *Instituto Nacional de Estadística, Geografía e Informática* (INEGI 1984, 1987, 1988). The maps were digitized into Arc/Info format from paper sources using standard digitizing techniques, maintaining an RMS error of less than 0.004 in the map registration. The individual maps were joined into a single layer in a Transverse Mercator projection following edge-matching procedures. We assigned a nominal date of 1975, corresponding to the mean of the air photo dates, to represent the former LU/LC in the Chiapas highlands.

The second map series was published by the *Secretaria de Agricultura y Recursos Hidraulicos* (SARH 1994) as the *Inventario Nacional Forestal Periódico* (Periodic National Forest Inventory, Sorani et al., 1993), and is described as a "hybrid map" for forest inventory assessment (Sorani and Alvarez 1996). It was derived by updating existing INEGI land use classifications with 30-m resolution Landsat TM imagery and improved geographic control. Forested areas were visually classified from 1:250,000 mosaic color composites georeferenced with a 3 arc-second DEM (digital elevation model) by a team of photo interpreters experienced with former forest inventories (Sorani and Alvarez 1996). Classifications in non-forested areas were of lesser importance in this inventory. Because this map was based on cartographic units of the previous map (*Carta de Uso de Suelo y Vegetación*), it had the same accuracy and precision for cartographic purposes and varied only in its areal coverage of the various LU/LC classes (Sorani and Alvarez 1996). A digital copy of this map series was obtained from the *Instituto of Geografía, Universidad Nacional Autónoma de México.* The individual maps were placed into a common projection (Lambert), edge-matched, and joined into a single layer. Based on an average of the dates of the TM imagery employed, we assigned a nominal date of 1991 to represent the current LU/LC in the Chiapas highlands. Our temporal scale was therefore the 16-year period between 1975 and 1991.

In both map layers, discontinuity of polygons was evident at map edges. This was observed on paper *Uso del Suelo y Vegetación* maps and was also present in the digital *Inventario Nacional Forestal Periódico* maps. These were examined in the individual map tiles and were not altered in the final joined map layers. The map layers were minimally adjusted after reprojection to a common projection (UTM zone 15) and prior to clipping to the study area of the highlands region.

The perimeter of the Chiapas highlands region was demarcated by a 30-m DEM file (EPA 1993), generated by digitizing 1:250,000 scale INEGI topographic maps (UTM zone 15 projection). The 1500-m elevation level was defined as the lower limit of the study area. Overlays of the two digital maps clipped to the highlands region were made and areal summations of the LU/LC classes represented on the two map series were calculated.

In order to quantify LU/LC changes between 1975 and 1991, we devised a composite classification system for purposes of comparison. The INEGI maps used more than 60 classes, some of which were equivalent to one of the 39 UNAM classes (Sorani et al., 1993). However some differences had to be reconciled. For example, the first map reported secondary forests and shrub lands within each forest type, four tropical medium stature forest classes, 14 scrub vegetation classes, seven pasture classes, and six agriculture classes, but did not differentiate between open and closed forests nor report fragmented forests. By contrast, the second map reported fewer forest, scrub, and agricultural types, grouped all high and medium-stature tropical forests, and did not show secondary forest or scrub, but did separate open and closed forest and reported fragmented forests and perturbed areas (*áreas perturbadas*, defined as forested in the INEGI maps, but no longer so in 1991; SARH 1994). Rates of LU/LC change were calculated by comparing the areal coverage of the various equivalent classes between the two maps.

Based on a 16.3:83.7 ratio derived from the relative distribution of pasture and cultivated land in the INEGI (1984, 1987, 1988) maps of the Chiapas highlands, land use in the *áreas perturbadas* class of the second map was assigned to either pasture or cultivated land based on adjacent polygons until 16.3% of the area was assigned as pasture. In cases where there were no adjacent pasture or cultivated land polygons, perturbed sites in lands that had previously been forested were assigned to pasture. The remaining area of perturbed polygons (83.7%) was then assigned to the cultivated land class.

After carefully reviewing the coding systems and definitions of terms in documentation for both maps, the following composite classes were adopted as a common basis for comparison:

Oak and evergreen cloud forests. Forests in which oak trees constitute >80% of the basal area, and remaining small patches of mixed evergreen cloud forests.

Pine-oak forests. Forests in which oak and pine trees together constitute >80% of the basal area.

Pine forests. Forests in which pine trees constitute >80% of the basal area.

Total canopy cover of the first three classes is >40%.

Degraded and fragmented forests. Includes open forests (10 - 40% canopy cover), shrubland, and fragmented forest from SARH (1994), and secondary forest, secondary shrubland, tree plantations, and nomadic agriculture from INEGI (1984, 1987, 1988).

Cultivated land. Includes seasonal and permanent agricultural systems.

Pasture. Includes induced and introduced pasture and grasslands.

Water bodies. Includes lakes and streams. Assumed to contain minimal biomass and excluded from C flux analyses.

Carbon densities

Carbon density data for the major LU/LC classes and the main pools in each class were collected in 39 field plots of 60 * 90 m (0.54 ha) each. The number of plots of each class and their locations were selected to be generally representative of a combination of (1) the distribution of LU/LC classes in the study area and (2) the variability in C densities (based on González-Espinosa et al., 1995a), ranging from three plots for the low C-density land uses (pasture and cultivated land) to 10-11 for the pine-oak and degraded and fragmented forest classes (the most common land-use classes and those expected to exhibit high variability in C density). Each plot was divided into six 30 * 30 m subplots to account for intrasite variability.

The average C storage was calculated per unit area (MgC ha^{-1}) for the various pools of each major LU/LC class, with a transformation of data to account for slope . Diameter at breast height (dbh, 1.3 m) and total height were measured and species determined for all trees >5 cm dbh. Locally developed allometric equations of biomass in relation to dbh and height were used to calculate C densities of the tree component in each plot. All saplings, herbaceous plants, and litter were sampled in four 1-m^2 plots, dried and weighed in the laboratory, and the C content was determined on a subsample for each plot. Soil samples were collected in four 1-m deep pits and separated into 1-10, 11-20, 21-30, 31-50, and 51-100 cm deep strata. Soil organic matter (SOM) C content was determined for each sample in the laboratory on three replicates (Walkley and Black 1934). The volume of fallen dead twigs, branches, and trees was measured using a line intersect method with corrections for vertical angles (Brown and

Roussopoulos 1974). Samples of branches and twigs were taken to the laboratory to determine wood density and C content. Small roots (<0.5 cm) were collected in four 0.25 m^2, 1 m deep pits. The samples were oven dried and weighed (Saldarriaga et al., 1988). Carbon content was determined on sub-samples of the roots from each pit. Carbon densities of large roots (>0.5 cm) were estimated from published allometric relationships between root and aboveground biomass (Pinard and Putz 1997; Cairns et al., 1997).

A one-way analysis of variance (ANOVA) was performed to test for differences among the LU/LC classes for overall, aboveground, root, and SOM C densities (SAS Institute Inc. 1988). Where significant differences were detected by the ANOVA, the Scheffe multiple comparison procedure was used to identify specific pairwise differences among the various classes. The level of significance was $\alpha = 0.05$.

The C density of the oak and evergreen cloud forest class was calculated as an area-weighted average of the C densities of the sampled oak (n=5) and evergreen cloud forest (n=2) plots. The C density of the degraded and fragmented forest class was calculated as an area-weighted average of the C densities of the sampled tree (n=6) and shrub (n=4) fallows. In both cases, weighting was done according to the relative 1975 areas of these land covers.

To calculate the C pools and flux resulting from land-use changes between the 1970s and 1990s, we separated vegetation pools, including live biomass, necromass, and roots, from the SOM pool. For each LU/LC class we then multiplied the total area present in both 1975 and 1991 by the average C densities of the vegetation and SOM pools. For demonstration purposes, but not for analytical use, we presented C flux in the study area for the period of time represented by our study.

To examine possible errors in estimates associated with field-derived C densities, we calculated 95% confidence intervals (CI) for vegetation and SOM C pools (SAS Institute Inc. 1988) and reported the resulting ranges in C flux.

Results

Changes in land use and land cover

During the 16-year period between the two inventories, the area of oak and evergreen cloud forests decreased by 50%, while both pine-oak and pine forests decreased by 51% (Table 3.1).

Table 3.1. *Chiapas highlands land use/land cover (LU/LC) change from the 1970s and 1990s.*

LU/LC Class	**Area (ha) 1975[a]**	**Area (ha) 1991[b]**	**Change**
Oak and evergreen cloud forests	65,312	32,699	-50%
Pine-oak forests	119,139	58,249	-51%
Pine forests	72,109	35,001	-51%
Total of closed forests:	256,560	125,949	-51%
Degraded and fragmented forests[c]	203,236	316,414	56%
Cultivated land	136,182	146,563[d]	8%
Pasture	23,603	30,685[d]	30%
Water bodies	581	551	-5%
Totals	620,162	620,162	

a Nominal year of INEGI (1984, 1987, 1988) land use and vegetation inventory.

b Nominal year of SARH (1994) national forest inventory.

c Includes open forests and fragmented forests from SARH (1994) and secondary forest, secondary shrub, tree plantations, and nomadic agriculture from INEGI (1984, 1987, 1988).

d Includes áreas perturbadas of SARH (1994) (61,507 ha) parceled to cultivated lands (83.7%) and pasture (16.3%).

Thus the total area of closed forests was reduced by about half, from 256,560 to 125,949 ha. Meanwhile, the degraded and fragmented forests increased by 113,178 ha, cultivated lands grew by 10,381 ha, and pastures increased by 7,082 ha. The net LU/LC change during this time interval was thus a trend away from closed forest ecosystems toward open, fragmented, and secondary forests and, to a lesser extent, toward various agricultural uses of the land (Figures 3.2 and 3.3).

While the net trend was from closed forests toward disturbed forests and agricultural land uses, there is also evidence of abandonment and forest regeneration for some of the land in agropastoral uses between 1975 and 1991. The 1991 LU/LC areas include 61,507 ha classified as *áreas perturbadas* in the SARH (1994) forest inventory parceled 83.7% to cultivated land (51,481 ha) and 16.3% to pasture (10,026 ha). When these areas are subtracted, the result is that the agropastoral land use decreased between 1975 and 1991. Thus abandonment of 41,100 ha (30%) of the cultivated land and 2,944 ha (12%) of the pasture, and regrowth as forest, may be deduced. The distribution of forest classes to which these lands regenerated is unknown.

Carbon densities and flux

Mean total C densities in the LU/LC classes varied more than three-fold, from 147 to 504 Mg ha^{-1} in pastures and oak and evergreen cloud forest, respectively (Table 3.2). Only cultivated land and pasture had statistically lower total mean C densities than did the forest classes. These modified LU/LC types had lower C densities due largely to smaller aboveground and root biomass pools. The belowground C densities showed a decrease from approximately 279 Mg ha^{-1} in oak and evergreen cloud forests to 129 and 154 Mg ha^{-1} in pasture and cultivated land, respectively. The average CV for total C density was 29%, with higher values associated with pasture aboveground (94%) and root biomass (114%), pine forest aboveground biomass (58%), and pine-oak forest SOM (53%) (Table 3.2). Lower coefficients of variation were observed in oak and evergreen cloud forests (14%) and cultivated land (14%).

A close examination (Table 3.2) reveals that the lower belowground C densities are a result of lower root biomass C densities, which range from <1 to 36 Mg ha^{-1} , while SOM C densities were not significantly different, ranging from 125 to 243 Mg ha^{-1}. The proportion of roots in the total belowground C pools decreased from a range of 13-15% in the intact forest types to a range of 0.4-5% in the modified LU/LC types (Table 3.2). When only the forest LU/LC classes were compared (agropastoral land uses excluded), only the degraded and fragmented forests had significantly lower aboveground and root C densities.

Table 3.2. *Estimated carbon (C) densities (MgC ha-1) for Chiapas highlands land use/land cover (LU/LC) classes. Aboveground includes tree and herbaceous phytomass, coarse woody debris, and litter. Belowground includes roots and soil organic carbon. Means with the same superscript are not statistically different (α = 0.05) from other means in the same column.*

LU/LC Class	n	Aboveground C Density	Belowground C Density		Total C Density
			Roots	Soil	
Oak and evergreen cloud forests	*7*				
Mean		189.0^{A}	36.0^{A}	242.8^{A}	503.7^{A}
Range		108.4-302.0	20.7-55.1	101.5-460.8	232.3-818.0
CV (%)		14	16	16	14
Pine-oak forests	*11*				
Mean		135.4^{A}	30.9^{A}	174.4^{A}	340.7^{A}
Range		77.6-168.6	19.8-40.5	85.4-412.0	229.2-595.6
CV (%)		21	24	53	32
Pine forests	*5*				
Mean		120.0^{AB}	25.7^{AB}	172.6^{A}	318.3^{A}
Range		79.7-244.0	16.7-46.1	129.7-275.9	243.1-566.0
CV (%)		58	46	34	44
Degraded and fragmented forests	*10*				
Mean		29.1^{B}	8.9^{BC}	184.2^{A}	222.2^{A}
Range		9.6-106.2	1.3-26.8	144.5-315.5	168.2-401.7
CV (%)		32	36	46	44
Cultivated land	*3*				
Mean		6.0^{B}	0.6^{C}	153.3^{A}	159.9^{B}
Range		4.0-8.1	0.2-0.8	135.2-177.8	143.6-184.4
CV (%)		34	53	14	14
Pasture	*3*				
Mean		18.1^{B}	3.8^{BC}	124.8^{A}	146.7^{B}
Range		4.1-37.1	0.8-8.7	88.2-183.5	117.7-188.4
CV (%)		94	114	41	25

Table 3.3. *Carbon (C) pool (95% CI[a]) and flux (range[b]) estimates resulting from land use change between the 1970s and 1990s in the Chiapas highlands. Positive values indicate flux from the terrestrial biosphere; negative values indicate flux from the atmosphere.*

Land Use/Land Cover Class	C Pools 1975 (Tg)		C Pools 1991 (Tg)		C Flux 1975-1991 (Tg)		
	Vegetation	Soils	Vegetation	Soils	Vegetation	Soils	Total
Oak and evergreen cloud forests	14.7 (1.6)	15.9 (1.9)	7.4 (0.8)	7.9 (0.9)	7.3 (5,9.7)	7.9 (5.1,10.7)	15.3 (10.1, 20.4)
Pine-oak forests	19.8 (2.5)	20.8 (6.5)	9.7 (1.2)	10.2 (3.2)	10.1 (6.4,13.9)	10.6 (1,20.3)	20.8 (7.4, 34.2)
Pine forests	10.5 (5.1)	12.5 (3.7)	5.1 (2.5)	6.0 (1.8)	5.4 (-2.2,13.0)	6.4 (0.9,11.9)	11.8 (-1.3, 24.9)
Degraded and fragmented forests	7.7 (1.6)	37.4 (10.8)	12.0 (2.5)	58.3 (16.8)	-4.3 (-8.4,-0.2)	-20.9 (-48.4,6.7)	-25.1 (-56.7, 6.4)
Cultivated land	0.9 (0.3)	20.9 (3.4)	1.0 (0.3)	22.5 (3.7)	-0.1 (-0.6,0.5)	-1.59 (-8.6,5.4)	-1.66 (-9.3, 5.9)
Pasture	0.5 (0.6)	3.0 (1.4)	0.7 (0.7)	3.8 (1.8)	-0.2 (-1.5,1.2)	-0.9 (-4.0,2.3)	-1.0 (-5.5, 3.4)
Total C Pools and Flux	54.2 (11.6)	110.4 (27.6)	35.8 (8.0)	108.7 (28.1)	18.4 (-1.3,38)	1.6 (-54.1,57.3)	20 (-55.4, 95.3)
As % of Total 1975 C Pools					34%	1%	12%

a CI = confidence interval.

b Range is based on error associated with estimating C flux from differencing 1975 and 1991 pools only. Upper range = (maximum pool for 1975 - minimum pool for 1991), and lower range = (minimum pool for 1975 - maximum pool for 1991). Negative pools were assumed equal to zero for flux calculations.

From differences in C storage between 1975 and 1991, we estimate that a net total of 19.99 TgC were released to the atmosphere during the study (Table 3.3). This is equal to approximately 12% of the total 1975 biospheric C pool. Approximately 34% of the 1975 vegetation C pool was lost, while the SOM C pool remained nearly stable. Total C flux in the study area is shown for demonstration purposes only (Figure 3.4). The primary flux of terrestrial C to the atmosphere may be attributed to the loss of above and belowground biomass from the three closed forest ecosystems in the study area.

A large degree of uncertainty exists in our flux estimates, even regarding the sign, i.e., whether the region is a net source or sink (Table 3.3). If extreme estimates were correct, e.g., maximum 1975 pools compared to minimum 1991 pools, or vice versa, then based only on field-derived C-densities estimates, the highlands study area could have been either a highly aggrading or degrading landscape.

Discussion

Although the study area has been intensively managed by indigenous Mayan populations for centuries with slash-and-burn (*milpa*) management systems, these systems have substantially changed in recent decades to more permanent agriculture and extensive cattle and sheep ranching. Lands under *milpa* have been intensified from long fallow and short crop rotations to systems with short fallow and long crop rotations, more resembling permanent agriculture. Furthermore, the fallow land is often intensively grazed by sheep or cattle, thus limiting regeneration of shrub or tree biomass (Parra-Vázquez et al., 1989).

Changes in forest cover can be largely attributed to opening of closed forests. Pine trees in both pine and pine-oak forests are cut for timber production, while oak trees are used for fuelwood and making charcoal. Logged pine forests are later used for extensive grazing of cattle and sheep, while pine-oak and oak forests are continuously logged for timber, fuelwood and charcoal production, or are cultivated or converted to pasture. Resulting landscapes include small patches of lightly perturbed old-growth forests surrounded by large areas of young secondary shrub and woody vegetation, heavily degraded forests, induced pastures, and cultivated lands (González-Espinosa et al., 1991; De Jong and Montoya-Gómez, 1994). Soil C densities measured in oak and evergreen cloud forests, pine-oak forests, and pine forests were somewhat higher than the range of values reported from similar forest types (Table 3.4).

Table 3.4. *Reported above- and belowground biomass and soil organic carbon (C) densities to 100 cm depth (Mg ha^{-1}) compared to the three closed forest land use/land cover classes, oak and evergreen cloud, pine-oak, and pine, represented in the Chiapas highlands.*

Forest type	**Biomass C**	**Soil C**	**Source**	**Comments**
Tropical montane	145	130	Adams, 1997	Pre-anthropogenic
Temperate coniferous	56	109	Masera et el., 1992	Nomenclature used in Mexico
Temperate broadleaf	39	30	Masera et al., 1992	
Temperate coniferous	50	102	Cairns et al., 1995	
Temperate broadleaf	60	102	Cairns et al., 1995	
Montane	60		Olson et al., 1983	Low values
Tropical premontane wet	142		Golley et al., 1975	Cited in Brown and Lugo, 1982
Tropical montane wet	214		Brun, 1976	
Tropical premontane moist	90		Bandhu et al., 1973	
Tropical premontane moist	181	44	Greenland and Kowal, 1960	
Tropical premontane moist	228		Huttel, 1975	
Subtropical lower montane wet	172		Tanner, 1980	
Tropical lower montane wet		193	Holdridge et al., 1971	
Tropical lower montane moist		170	Holdridge et al., 1971	
Subtropical/tropical wet/rain		175	Detwiler, 1986	Adapted from Brown and Lugo, 1982
Subtropical/tropical moist		98	Detwiler, 1986	
Tropical montane moist		120	Detwiler, 1986	from Post et al., 1982
Tropical montane wet		160	Detwiler, 1986	
Tropical closed		123	Detwiler, 1986	all tropical life zones
Reported averages	**120**	**120**		
Reported ranges	**39-228**	**30-193**		
This study averages	**179**	**197**		
This study range of averages	**146-225**	**174-243**		

Oak and evergreen cloud forests had a particularly high soil C density, 243 Mg ha^{-1}. Soil C densities were generally high across all forest classes because the soils are calcareous and the active calcium carbonate and, to a lesser degree, the exchangeable calcium in saturated environments are effective stabilizers, protecting organic matter against microbial biodegradation (Duchaufour 1982). Other factors affecting soil C storage include rainfall and soil fertility (Lugo and Brown 1993). Detwiler (1986) notes no reduction in soil C density due to harvesting. This observation is in agreement with our results. Thus the soil C density of the degraded and fragmented forests (184 Mg ha^{-1}) was not lower than the three closed forest classes (Table 3.2). In contrast, Houghton et al. (1983) report a 35% decrease in soil C following selective logging in tropical forests.In this study soil C were reduced 22% in cultivated land and 36% in pasture, relative to the average values for the intact forests. These results may be compared with those reported by Detwiler (1986) that the cultivation of tropical soils reduces their C content by 40% five years after clearing and that use of these lands for pasture reduces their soil C content by about 20% within one year. Similarly, Houghton et al. (1983) report that a conversion of tropical forests to permanent agriculture decreases soil C by 50% and conversion to pasture decreases soil C by 25%. A possible explanation for the difference is that the cultivated plots in our study may have been created from forests more recently and the pastures were more intensively impacted and/or were older than those observed by Detwiler (1986) and Houghton et al. (1983).

Because these land use changes not only cause changes in land cover, but may also result in changes in C density within each LU/LC, we suspect that the average C density for each land cover class may have been higher in 1975 than in 1991, the latter being close to when C densities were measured in our study. For example, activities such as grazing, or pole and firewood harvest, can reduce forest biomass without affecting crown cover when observed by remote sensing. Thus the C flux that occurred between 1975 and 1991 might be higher than we have estimated. This could not be verified because of the lack of previous studies of soil and biomass C in Chiapas highlands forests. However, such biomass degradation has been reported for other tropical forests. (Flint and Richards 1991; Brown and Lugo 1992).

Potential sources of error in our analysis associated with cartographic data included the wide range of dates encompassed in the *Uso del Suelo y Vegetación* maps, the multiple organizations participating in the generation of both map layers, and the different classification systems employed by the original map creators. Because the aerial photography covered a decade, it is impossible to define precisely the time frame over which the modeled LU/LC occurred, thus we used a nominal date. While the SARH (1994) map is estimated to have an accuracy rate of >85% (Sorani et al., 1993), exact accuracy of the INEGI (1984, 1987, 1988) maps is unknown although the aerial photography was field-verified at a minimum of 50 points per map, indicating a relatively high degree of accuracy at the time of compilation of the maps. A complete independent assessment of the accuracy of the registration of the two map layers was not possible due to a shortage of relocatable points within the layers; however, best efforts were made to find and align identifiable locations.

In a project in Mexico's Yucatán Peninsula employing a comparison of the same cartographic data sources we used in our present work, Carranza-Sánchez et al. (1996) concluded that because of the differences between the two sources in terms of their use of aerial photography vs satellite imagery, relative degree of ground truthing, and the different terminology used to label LU/LC classes, that these maps are unsuitable for estimating land use change and deforestation. While we agree that the ideal situation for comparing past and future LU/LC inventories would use similar data sources, others have recently reported land use change analysis based on comparisons of different map layers and overlay techniques (Schreier et al., 1994; Kress et al., 1996; Thomlinson et al., 1996). Because the 1990s map (SARH 1994) uses as its basis the same cartographic data as the 1970s map (INEGI 1984, 1987, 1988), the primary factor making a comparison difficult is the difference in labeling LU/LC classes. We feel that for C-flux purposes, unlike for strictly land use change or deforestation purposes, comparisons based on our assumptions regarding equivalent LU/LC classes is clearly warranted. Our analysis used two digital maps of the Chiapas highlands produced by Mexican governmental agencies for these time periods that are the best available sources.

Forest biomass C densities are commonly estimated in one of two ways. One method uses inventory data, either volume estimates or stand tables, to calculate total aboveground biomass (Brown 1997). Advantages of this approach are that inventory data are usually more abundant and gathered from large sampling areas using a planned sampling method designed to represent the population of interest. Common problems with the inventory approach are that such inventories are usually focused on commercial forests and species, only large trees are measured, many of the inventories are very old, critical data and regions are omitted, only the aboveground tree biomass is estimated, and the maximum diameter class is open-ended. The second approach is based on ecological studies that have sampled the various biomass pools in plots of various sizes and using different methods (Brown and Lugo, 1982; Olson et al., 1983). The main advantages are that local conditions are accounted for and all the ecosystem components can be measured. Disadvantages are that study sites are frequently not representative of the population of interest, results cannot be extrapolated to larger areas, and sites are often selected in forests with many large diameter trees thus tending to overestimate biomass density (Brown et al., 1989; Brown and Lugo, 1992).

Our approach incorporates some strengths of both approaches. All LU/LC class biomass and soil C pools were accounted for, locally developed allometric equations were used to calculate tree biomass C densities, and all species and sizes of plants were measured. Even though our plots were not strictly randomly selected and the total area sampled was <0.01% of the study area, the locations and numbers of plots were selected to be representative of both the distribution and variability of the LU/LC types present in the Chiapas highlands.

A measure of statistical confidence in our mean C-density estimates provides a unique opportunity to demonstrate the uncertainty of estimating either regional C flux, as in this study, or national greenhouse gas inventories, as required by the UN Framework Convention on Climate Change (United Nations 1992). Because cartographic errors were not quantified, the errors indicated for C pool and flux estimates are only those associated with experimental plot sampling (Table 3.3). The resulting degree of uncertainty, as expressed as ranges, in our flux estimates is large. While some of the confidence intervals are larger than the mean C pool values, we may rightfully assume these values are unrealistic, because this would infer negative pools (i.e., pasture vegetation). We therefore assumed them to be equal to zero when calculating C flux. Because zero pools of known LU/LC ecosystems are unrealistic, ranges in our reported C flux values may be considered to reflect worst-case errors. For instance, because we know that the area of pine forests decreased in our study area between the 1970s and 1990s, it is reasonable to assume that the vegetation and total net fluxes could not have been from the atmosphere (= negative values in Table 3.3). Similarly the areas of both cultivated land and pasture increased, thus net fluxes could not have been from the biosphere (= positive values in Table 3.3). While the large range in SOM flux estimates indicates a high degree of heterogeneity in soil C concentrations, the net result is assumed to be approximately no change in the SOM pool. These constraints on the error associated with our C-flux estimates due to field sampling make them more realistic and strengthen our confidence in the values.

Means of reducing errors in C-flux estimates would include using larger sample sizes and conducting sequential LU/LC inventories in an identical manner so that spatial matching of cartographic areas and C densities would be possible. These means would also increase the costs of producing greenhouse gas emissions estimates. Standard guidelines for reporting greenhouse gas emissions encourage the quantification of uncertainty (UNEP/OECD/IEA/-IPCC 1995) but the default values that are commonly employed make such quantification difficult.

The latest Mexican greenhouse gas inventory update reports that land use change and forestry result in a national annual net emission of 37.1 TgC as CO_2, composed of a source of 59.4 TgC, partially offset by a sink in managed forests and abandoned lands of 22.3 TgC (Masera et al., 1995). Therefore, based on our findings, the Chiapas highlands, representing only 0.3% of Mexico's territory, contributed 3% of the nation's annual LU/LC C emissions [(17.5 Tg/16 yr)/37.1 Tg, Table 3.3)]. This indicates that the Chiapas highlands have experienced a rapid process of deforestation and land use change during these 16 years compared to the rest of Mexico.

Acknowledgments

The authors thank Frank Golley, Warren Cohen , T. Hernandez, Sandra Brown, Omar Masera, Robert Rice, Arturo Sánchez, Ernesto Alvarado, and Roman Alvarez for thoughtful reviews of this manuscript. Thanks also to Ronald Shea, Valentino Sorani, and Greg Baumgardner for critical assistance and discussions. The information in this document has been funded by the U.S. Environmental Protection Agency under Cooperative Agreement CR822200 with El Colegio de la Frontera Sur, Interagency Agreement RW72936017 with the U.S. Agency for International Development/Mexico, and Contract 68-W5-0065 with OAO Corporation. It has been subjected to the Agency's peer and administrative review and approved for publication as an EPA document. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

References

Adams, J. 1997. Estimates of pre-anthropogenic carbon storage in global ecosystem types. In An inventory of data, for reconstructing 'natural steady state' carbon storage in terrestrial ecosystems. http://www.esd.ornl.gov/ern/qen/carbon1.htm. Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN.

Bandhu, D., Gist, G.A., Ogawa, H., Strand, M.A., Bullock, J., Malaisse, F., and Rust, B. 1973. Seasonal model of the tropical forests. In L. Kern (ed.) Modeling forest ecosystems. EDFB-IBP 737. Oak Ridge National Laboratory, Oak Ridge, TN. p. 285-295.

Breedlove, D.E. 1981. Flora of Chiapas, Part 1: Introduction to the Flora of Chiapas. The California Academy of Sciences, San Francisco, CA.

Bremer, J., Babb, T., Dickinson, J., Gore, P., Hyman, E., and Andre, M. 1984. Fragile lands: a theme paper on problems, issues, and approaches for development of humid tropical lowlands and steep slopes in the Latin American Region. Development Alternatives, Washington D.C.

Brown, J.K., and Roussopoulos, P.J. 1974. Eliminating biases in the planar intersect method for estimating volumes of small fuels. Forest Science 20: 350-356.

Brown, S. 1997. Estimating biomass and biomass change of tropical forests. A primer. FAO Forestry Paper 134. Food and Agriculture Organization of the United Nations, Rome. 55 pp.

Brown, S., and Lugo, A.E. 1982. The storage and production of organic matter in tropical forests and their role in the global carbon cycle. Biotropica 14: 161-187.

Brown, S., and Lugo, A.E. 1992. Aboveground biomass estimates for tropical moist forests of the Brazilian Amazon. Interciencia 17: 8-18.

Brown, S., Gillespie, A.J.R., and Lugo, A.E. 1989. Biomass estimation methods for tropical forests with applications to forest inventory data. Forest Science 35: 881-902.

Brun, R. 1976. Methodik und ergebnisse zur biomassenbestimmung eines nebelwald-okosystems in den Venozolanischen Anden. In Proceedings of Division I, 16th IUFRO World Congress. Oslo, Norway. 490-499.

Cairns, M.A., Brown, S., Helmer, E. H., and Baumgardner, G. A. 1997. Root biomass allocation in the world's upland forests. Oecologia 111: 1-11.

Cairns, M.A., Kolchuniga, T.P., Turner, D.P., and Winjum, J.K. 1995. The contribution of forest land use to total national carbon flux: case studies in the former Soviet Union, United States, Mexico and Brazil. EPA/600/R-95/044. U.S. Environmental Protection Agency, Corvallis, OR. 212 pp.

Carranza-Sánchez, J., Barreto-Salinas, C., Molina-Islas, C., Bezaury-Creel, J.E., and Loreto-Viruel, A.A. 1996. Análisis cartográfico del cambio de uso de suelo en la Peninsula de Yucatán, México. Amigos de Sian Ka'an A.C., Cancún. Quintana Roo, México. 83 pp.

Cook, A.G., Janetos, A.C., and Hinds, W.T. 1990. Global effects of tropical deforestation: towards an integrated perspective. Environmental Conservation 17: 201-212.

De Jong, B.H.J., and Montoya-Gómez, G.. 1994. Sustainable management of forest resources: a proposal for the highlands of Chiapas, Mexico. Pages 7-17 in Proceedings of the 1994 symposium on systems analysis in forest resources: management systems for a global economy with global resource concerns, Sept 6-9, 1994. Pacific Grove, CA.

Detwiler, R.P. 1986. Land use change and the global carbon cycle: the role of tropical soils. Biogeochemistry 2: 67-93.

Detwiler, R.P., and Hall, C.A.S. 1988. Tropical forests and the global carbon cycle. Science 239: 42-47.

Duchaufour, P. 1982. Pedology. George Allen & Unwin, London. 448 pp.

EPA (United States Environmental Protection Agency). 1993. North American Landscape Characterization (NALC). Research Plan. EPA/600/R-93/135. U. S. Environmental Protection Agency, Washington, DC. 419 pp.

FAO (Food and Agriculture Organization of the United Nations). 1993. Forest resources assessment 1990 tropical countries. FAO Forestry Paper 112. FAO, Rome. 59 pp.

Flint, E.P. and Richards, J.F. 1991. Historical analysis of changes in land use and carbon stocks of vegetation in South and Southeast Asia. Canadian Journal of Forest Research 21: 91-110.

Golley, F.B., McGinnis, J.T., Clements, R.G., Child, G.I., and Duever, M.J. 1975. Mineral cycling in a tropical moist forest ecosystem. University of Georgia Press, Athens, GA. 248 pp.

González-Espinosa, M., Quintana-Ascencio, P.F., Ramírez-Marcial, N., and Gaytán-Guzmán, P.. 1991. Secondary succession in disturbed Pinus-Quercus forests in the highlands of Chiapas, Mexico. Journal of Vegetation Science 2: 351-360.

González-Espinosa, M., Ochoa-Gaona, S., Ramírez-Marcial, N., and Quintana-Ascencio, P.F. 1995a. Current land-use trends and conservation of old-growth forest habitats in the highlands of Chiapas, Mexico. Pages 190-198 in M.H. Wilson and S.A. Sader (eds.) Conservation of Neotropical Migratory Birds in Mexico. Maine Agricultural and Forest Experiment Station, Miscellaneous Publication 727, Orono, ME.

González-Espinosa M., Ramírez-Marcial, N., Quintana-Ascencio, P.F., and Martínez-Icó, M. 1995b. La utilización de los encinos y la conservación de la biodiversidad en Los Altos de Chiapas. In: Memorias del tercer seminario nacional sobre utilización de encinos, 4-6 de Noviembre de 1992. Reporte Cientifico UANL, Número Especial 15. Linares, Nuevo Leon, Mexico. 183-197.

Greenland, D.J., and Kowal, J.M.L. 1960. Nutrient content of the moist tropical forest of Ghana. Plant Soil 12: 154-173.

Hall, C.A.S., and Uhlig, J. 1991. Refining estimates of carbon released from tropical land-use change. Canadian Journal of Forest Research 21: 118-131.

Holdridge, L.R. 1967. Life zone ecology. Tropical Science Center, San José, Costa Rica. 206 pp.

Holdridge, L.R., Grenke, W.C., Hatheway, W.H., Liang, T., and Tosi, J.A. 1971. Forest environments in tropical life zones. Pergamon Press, New York. 747 pp.

Houghton, R.A., Hobbie, J.E., Melillo, J.M., Moore, B., Peterson, B.J., Shaver, G.R., and Woodwell, G.M. 1983. Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: a net release of CO_2 to the atmosphere. Ecological Monographs 53: 235-262.

Houghton, R.A., Skole, D.L., and Lefkowitz, D.S. 1991. Changes in the landscape of Latin America between 1850 and 1985: II. Net release of CO_2 to the atmosphere. Forest Ecology and Management 38: 173-199.

Huttel, C.H. 1975. Root distribution and biomass in three Ivory Coast forest plots. Pages 123-130 in F.B. Golley and E. Medina (eds.) Tropical ecological systems. Ecological Studies 11. Springer-Verlag, New York.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica). 1984. Carta Uso del Suelo y Vegetación 1:250 000. Tuxtla Gutierrez E15-11. INEGI, Mexico, D.F.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica). 1987. Carta Uso del Suelo y Vegetación 1:250 000. Villahermosa E15-8. INEGI, Mexico, D.F.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica). 1988. Carta Uso del Suelo y Vegetación 1:250 000. Las Margaritas E15-12. INEGI, Mexico, D.F.

Kress, M.R., Graves, M.R., and Bourne, S.G.. 1996. Loss of bottomland hardwood forests and forested wetlands in the Cache River Basin, Arkansas. Wetlands 16: 258-263.

Lambin, E.F. 1994. Modelling deforestation processes: a review. Tropical Ecosystem Environment Observations by Satellites, TREES Series B, Research report No. 1. Office of the European Commission, Directorate-General XIII. Luxemburg. 113 pp.

Lugo, A.E., and Brown, S. 1993. Management of tropical soils as sinks or sources of atmospheric carbon. Plant and Soil 149: 27-41.

Masera, O., Ordóñez, M. de Jesús, and Dirzo, R. 1992. Carbon emissions from deforestation in Mexico: Current situation and long–term scenarios. Pages 1–49 in W. Makundi and J. Sathaye (eds.) Carbon emission and sequestration in forests: Case studies from seven developing countries, Summary. Report No. LBL–32665. Lawrence Berkeley Laboratory, University of California, Berkeley, CA.

Masera, O., Hernández, T., Ordoñez, A., and Guzmán, A. 1995. Chapter 5. Land use change and forestry. In: C. Gay, L.G. Ruíz-Suarez, M. Imaz, and J. Martínez (eds.) Preliminary national inventory of greenhouse gas: Mexico. September, 1995. Secretaría de Medio Ambiente, Recursos Naturales y Pesca. México City. Updated March, 1997 by Table: Summary Report for National Greenhouse Gas Inventories, J. Martinez. p. 56-100.

Ochoa-Gaona, S. and González-Espinosa, M.. 1999. Land-use patterns and deforestation in the Highlands of Chiapas, Mexico. Applied Geography 20: 17-42.

Olson, J.S., Watts, J.A., and Allison, L.J. 1983. Carbon in live vegetation of major world ecosystems. Report ORNL-5862, ESD Publication No. 1997. Oak Ridge National Laboratory, Oak Ridge, TN. 152 pp.

Parra-Vázquez, M.R., Alemán-Santillán, T., Nahed-Toral, J., Mera-Ovando, L.M., López-Mejía, M., and López-Meza, A. 1989. El subdesarrollo agrícola en los altos de Chiapas. Universidad Autónoma de Chapingo, Chapingo, México. 405 pp.

Pinard, M.A., and Putz, F.E.. 1997. Monitoring carbon sequestration benefits associated with a reduced-impact logging project in Malaysia. Mitigation and Adaptation Strategies for Global Change 2: 203-215.

Post, W.M., Emanuel, W.R., Zinke, P.J., and Stangenberg, A.G. 1982. Soil carbon pools and world life zones. Nature 298: 156-159.

Saldarriaga, J.G., West, D.C., Tharp, M.L., and Uhl, C. 1988. Long-term chronosequence of forest succession in the upper Rio Negro of Colombia and Venezuela. Journal of Ecology 76: 938-958.

SARH (Secretaria de Agricultura y Recursos Hidraulicos). 1994. Inventario nacional forestal periódico. Secretaría de Agricultura y Recursos Hidraulicos, México, D.F. 81 pp.

SAS Institute Inc. 1988. SAS/STAT User's Guide, Release 6.03 Edition. SAS Institute, Cary, NC. 1028 pp.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. Global Change Biology 1: 77-91.

Siegenthaler, U., and Sarmiento, J.L. 1993. Atmospheric carbon dioxide and the ocean. Nature 365: 119-125.

Schreier, S., Brown, S., Schmidt, M, Shah, P., Shrestha, B., Nakarmi, G., Subba, K., and Wymann, S. 1994. Environmental auditing – gaining forests but losing ground: a GIS evaluation in a Himalayan watershed. Environmental Management 18: 139-150.

Sorani, V., and Alvarez, R. 1996. Hybrid maps: updating Mexico's forest cartography using Landsat TM imagery and land use information. Geocarto International 11: 17-23.

Sorani, V, Alvarez, R., Baca, J.C., and Varela, S. 1993. The forest inventory of Mexico, Phase 1: Classification with TM imagery. In: Proceedings of the 25th international symposium, remote sensing and global environmental change, Volume II, 4-8 April 1993, Graz, Austria. Environmental Research Institute of Michigan, Ann Arbor, MI. 423-434.

Tanner, E.V. 1980. Studies on the biomass and productivity in a series of montane rain forests in Jamaica. Journal of Ecology 68: 573-588.

Thomlinson, J.R., Serrano, M.I., Lopez, T.M., Aide, T.M., and Zimmerman, J.K. 1996. Land-use dynamics in a post-agricultural Puerto Rican landscape (1936-1988). Biotropica 28: 525-536.

United Nations. 1992. Framework Convention on Climate Change. Geneva. Switzerland.

UNEP/OECD/IEA/IPCC (United Nations Environmental Programme, The Organisation for Economic Co-operation and Development, The International Energy Agency and the Intergovernmental Panel on Climate Change). 1995. IPCC guidelines for national greenhouse gas inventories. Volume I, II, III. IPCC, Geneva, Switzerland.

Walkley, A., and Black, A. 1934. An examination of the Degjareff method for determining soil organic matter and a proposed modification of the chromic acid titration method. Soil Science 37: 29-38.

4. FORESTRY AND AGROFORESTRY LAND-USE SYSTEMS FOR CARBON MITIGATION: AN ANALYSIS IN CHIAPAS, MEXICO[2]

Introduction

The potential role of forests in carbon sequestration has been evaluated by a number of authors (Marland, 1988; Sedjo and Solomon, 199; Andrasko et al., 1991; Houghton et al., 1991). They suggest that forest conservation, establishment and management, as well as agroforestry, could contribute to global carbon mitigation. Forestry and agroforestry compensate for greenhouse-gas emissions in two ways: (i) creating new sinks for carbon dioxide (CO_2) by increasing the mass of woody material within growing trees and in harvested timber converted to durable products; and (ii) protecting natural forests and soils as carbon stores.

Strategies for carbon sequestration or conservation strategies in the forestry and the agroforestry sector include: the establishment of permanent agroforestry plots to substitute slash-and-burn agriculture; and the conservation of standing old-growth forests as carbon sinks. In addition, carbon sequestration can be enhanced through increased harvesting efficiency in forests and utilizing a higher percentage of total biomass; improving forest productivity on existing forest lands through management and genetic manipulation, establishing plantations on surplus cropland and pastures; restoring degraded forest ecosystems through natural regeneration and enrichment planting; establishing plantations and agroforestry projects with fast-growing and high-biomass species on short rotations for biomass and timber; and increasing soil carbon by leaving dead wood, litter, and slash from harvests.

Estimates of the global biological and economic potential of forest-management practices for controlling atmospheric CO_2 concentrations are highly speculative. Dixon et al. (1993) consider that economically viable forestry and agroforestry management practices could roughly conserve and sequester 1 PgC year^{-1} globally (1 PgC = 10^{15} gC = 1GtC). They estimate the marginal cost of implementing these options at $10 for each MgC. Trexler (1991) identifies a range of forestry policy options for US forests, which could make emission savings of 0.75 to 1.15 PgC yr^{-1} at a marginal cost of $US30 to 50 per MgC. A number of private projects,

[2] *Published as: De Jong, B., Soto, L., Montoya, G., Nelson, K., Taylor, J., and Tipper, R. 1997. Forestry and agroforestry alternatives for carbon mitigation: an analysis from Chiapas, Mexico. In: Adger W., Pettenella D., Whitby M.C., (eds.). Climate Change Mitigation and European Land Use Policies. CAB International, Wallingford, UK. p. 263-284.*

sponsored by power generators have claimed sequestration costs of less than $US5 per MgC (FACE, 1994), with some less than $US1 per MgC (Swisher, 1991).

These carbon-storage technologies can potentially be achieved at relatively low costs in most tropical countries, because the two principal inputs, land and labor, can, in some localities, be relatively cheap. With a greater understanding of the local social context of an (agro)forestry project, it appears possible to develop projects of CO_2 mitigation and simultaneously help finance promising sustainable development programs (Swisher and Masters, 1992). Accordingly, Gay et al. (1995) state that efforts should only be made to implement options that are inexpensive or economically profitable; consistent with social and cultural factors; provide tangible benefits at the local level; and provide environmental benefits in addition to greenhouse-gas mitigation. Typically, these requirements need identification of options that can be implemented on different scales, for various orientations of output (commercial, subsistence), and with a range of beneficiaries (rural communities, private sector, governments).

Mexico ranks among the top ten countries in carbon releases from forest clearing during recent decades (Detwiler and Hall, 1988). Carbon-sequestration scenarios developed by Masera and colleagues (1994) suggest that Mexico can reduce significantly the country's net carbon emissions to the atmosphere, through forest management of native temperate and tropical forests and reforestation of degraded areas. Masera et al. (1995) outline seven forest-mitigation options for Mexico: four options related to increasing forested areas (restoration, pulpwood and energy plantations and agroforestry systems) and three related to the conservation of existing forests and resources (conservation of natural protected areas, management of native forests and dissemination of improved wood-burning cooking stoves). Apart from the fact that planting trees to sequester carbon is a prominent strategy for mitigating greenhouse-gas emissions (Adams et al., 1993, De Jong et al., 1995), it can also create a low-cost source of biomass and an alternative source of fuel and wood-based material for construction and other uses. The previous mentioned study results suggest that a modest carbon-sequestration program can achieve reasonable goals, without major negative impacts on agriculture and commodity production.

Based on the 1992 land reform, Mexico has sought to implement a progressive forest policy, particularly in tropical areas, which contain the most important forest extensions of the country. This policy has been underlined by a number of activities: the development of a national Tropical Forestry Action Plan; the establishment of the headquarters of the Forest Stewardship Council in Oaxaca, southern Mexico; the elimination of subsidies and credit for extensive cattle ranching in the tropical states; and government support to various community-controlled forest-management projects (examples are Plan Pilot Forestal Quintana Roo, Plan Piloto Forestal Marques de Comillas and Selva Lacandona).

The 1992 change in the Mexican Land Tenure Law (Article 27) gives legal title to the rural communities for the land they manage as an *ejido* or community. *Ejido* is a term used for a productive grouping of people with land, given in common ownership after the 1917 revolution. The 1992 changes in the Land Tenure Law allow rural farmers legal status to establish join ventures with investors, so that capital can be invested in alternative land-use systems. To ensure mutually profitable and just business partnerships, farmers´organizations will need to play an important role in the negotiations between the farmers they represent and interested investors.

Mexico has also played an important role in the development of joint initiatives for greenhouse-gas emission reductions, particularly in conjunction with the US Initiative on Joint Implementation (USIJI). Joint implementation (JI) refers to the idea that greenhouse-gas emission reductions or offsets in one country may counterbalance emissions generated in other countries, in order to achieve reductions agreed to under the United Nations Framework Convention on Climate Change. Legal provisions to allow co-funding of emission-mitigation projects in Mexico are in place. However, discussions relating to the distribution of the "carbon credits" from JI projects are still ongoing.

The Commission for Environmental Cooperation, formed by the environmental side-agreement to the North American Free Trade Agreement (NAFTA), is supporting greenhouse-gas mitigation projects in Mexico. Current free-trade negotiations between Mexico and the European Union could possibly enhance the Mexican JI potential. Certain countries, such as Germany, Austria and the Netherlands, which are having difficulties approaching their emission targets, appear to be potential partners for a JI program with Mexico.

The Mexican government is very keen to identify potential projects that are acceptable for rural communities and ecologically and economically viable, in order to incorporate them into a JI program. To this end, a case study has been carried out to identify farmer-preferred and ecologically viable forestry and agroforestry systems and their *ex ante* carbon-sequestration potential and to assess the economic potential of carbon offsets of such systems.

Using participatory methods, current agroforestry systems have been analyzed and potential system improvements and new alternative systems proposed, discussed and evaluated, based on social, economic, technical and carbon-sequestration criteria. Selected systems were used for carbon-offset calculations. In this chapter, total accumulated carbon for each system and region is presented. The costs of carbon sequestration for each system are estimated, based on the discounted direct costs of improving the current systems or establishing the new systems and the discounted opportunity costs during the first rotation for those systems where land use is diverted from agriculture or animal husbandry to forestry. Costs are presented on a per-hectare basis. In addition, we discuss the importance of community organization for an agro-forestry project.

We point out the advantages and disadvantages of available carbon-sequestration models tested in this study for the calculation of the potential carbon mitigation of the selected systems. Combining systems that provide commercial products and ecological services, such as carbon mitigation, could generate the necessary capital for the long-term investment in farm-forestry projects, which has always formed a constraint on such activities.

Description of the study areas

The state of Chiapas is located in southern Mexico. Tropical rain forests, pine-oak forests, and montane rain forests are among the important vegetation types. The majority of the rural communities consist of subsistence or semisubsistence farmers, who rely heavily on forest resources for fuel wood, construction materials, and food supplements.

The study was carried out in two ecological zones, one inhabited by Maya-Tojolabal (Tojolabal region) and the other by Maya-Tzeltal (Tzeltal region) indigenous groups. These particular zones were selected by the farmers´ organization Unión de Crédito Pajal Ya Kac´tic (Pajal), based on the biological potential of the regions for agroforestry and the interest demonstrated by their members in these zones. Five communities from each region were invited to participate in the feasibility study. All the participating communities retain legal land-tenancy titles as *ejidos* (for further details, see De Jong et al., 1995; Montoya et al., 1995).

The climate of the Tojolabal region is subtropical subhumid (García, 1973), with a mean annual temperature of 18°C, and 1,030 mm of annual rainfall. The communities lie at an average altitude of 1600 m above sea-level. Within these communities are well-preserved extensions of pine-oak forest. The five Tojolabal communities - Jusnajab, Yaluma, Lomantan, Bajucu and Palma Real - are situated in the municipalities of Comitán and Las Margaritas.

Each community is an *ejido*. The total area of the five communities is 9,281 ha, with 5,336 habitants, of which 439 are members of Pajal.

The climate of the Tzeltal region is tropical subhumid (García, 1973) with a mean annual temperature of 24 °C, and 1,800 mm of annual rainfall. The communities lie at an average altitude of 800 m above sea-level. There remains in the communities only fragments of primary evergreen rain forest and large extensions of shaded coffee. The five Tzeltal communities - Chapullil, Segundo Cololteel, Alan Cantajal, Muquenal, and Jol-Cacualha - are situated in the municipality of Chilón. The total area of the five communities is 2,387 ha, with 907 habitants, of which 170 are members of Pajal. Participants have shown great interest in improving their land management with agroforestry systems.

Methods

The suitability of agroforestry for carbon sequestration requires insights into the institutions involved at the implementation scale. The study therefore uses participatory methods to identify constraints to sustainability in the case-study areas. To begin the project, a multi-disciplinary team of scientists and farmers was established. The Pajal members of each community appointed two delegates to represent them during the study. These two representatives gathered information and designed the community agroforestry/forestry options together with the members in the communities, while receiving technical assistance from the researchers of El Colegio de la Frontera Sur (ECOSUR) and Pajal.

The design and evaluation of the options were carried out during two workshops, in which all of the representatives and scientific advisers participated. During the first workshop, the agroforest-management alternatives were discussed. This was also the first time that the concept of carbon sequestration was presented as a potential component of a forestry project. During the sessions, ECOSUR scientists and Pajal technicians explained the technical and social implications of agroforestry alternatives, while the community representatives supplied detailed descriptions of the land-use systems in their communities, the common tree species of the regions and how they manage these species. They explained their major land-use constraints and which of the farm-forestry and agroforestry alternatives they considered attractive.

Between the first and second workshop, the Pajal representatives collected data on: (i) community members' interest in a farm-forestry and agroforestry project, (ii) how much land they would have available for each activity, and (iii) the system(s) they would prefer, including species selection and tree distribution within the system. Data were also collected on the current social organizations, the land and tree tenure, benefit distributions, current community norms for forest use and possible target groups.

During the second workshop the final community proposals were developed and evaluated, incorporating the results of the first workshop, the visits and data collected by the representatives.

The following variables were included in the system analysis: (i) area available for farm forestry; (ii) potential species or species combinations; (iii) current land-use system in which they would plant the trees; (iv) planting design; and (v) how to obtain the planting material.

The simulation model CO_2FIX described by Mohren and Klein Goldewijk (1990) was used for the calculations of the carbon fluxes for each system. The model describes the carbon cycle from annual growth and loss rates of the main biomass compartments of the tree component of the systems, in combination with accumulation and turnover of soil organic matter. In this simulation program, a carbon accounting procedure is used to drive carbon accumulation in the entire biomass through proportionality coefficients derived from biomass measurements. Adjustments can be made for the rotation age, thinning procedures, product use and other silvicultural factors, etc., according to the species and silvicultural system (Nabuurs and Mohren, 1993). Existing local field data and published data on growth rates, decomposition rates and amounts of biomass in the various land use types were used in the model. Calculations of the growth rates are based on the site-quality indices developed for the regions. Mean annual increment (MAI) and total height of mature trees were used as the parameters to define the quality index. The MAI in the Tojolabal region was calculated, taking the average of the yearly increase of *Pinus* trees under different growing conditions. In the Tzeltal region, where trees in general do not present yearly growth rings, the MAI was determined by using the average diameter at breast height of the ten largest trees in forest fallows of known age and dividing this average by the age of the fallow. In table 4.1 we present the parameters used for this simulation (for a detailed description and application of the model, see Mohren and Klein-Goldewijk, 1990; Nabuurs and Mohren, 1995). The net accumulated amount of carbon over the first six cycles for the Tzeltal region (25 years for each cycle) and five cycles for the Tojolabal region (30 years for each cycle) was estimated.

Table 4.1. *Parameters for the CO_2FIX simulation model to estimate carbon fluxes in potential agroforestry systems in Chiapas, Mexico.*

Parameters	Tzeltal			Tojolabal		
Rotations (years)	6 x 25			5 x 30		
Initial humus content of the soil (MgC ha^{-1})	75			75		
Basic Wood Density (kg m^{-3})	500			450		
Carbon content (% of dry weight)	50%			50%		
Dry weight increment relative to stem increment during one rotation:	years after planting			years after planting		
	0-10	*10-20*	*20-25*	*0-10*	*10-20*	*20-30*
Foliage	0.7	0.4	0.4	0.8	0.6	0.2
Branches	0.6	0.4	0.4	0.8	0.5	0.2
Roots	0.7	0.4	0.4	0.9	0.6	0.3
Turnover rates:						
Foliage	0.5			0.3		
Branches	0.05			0.05		
Roots	0.07			0.07		
Humification factor	0.1			0.05		
Litter residence time (years)	1			3		
Stable humus residence time (years)	100			200		
Product allocation for thinnings:						
First thinning after 8 years	60% fuel wood					
	40% dead wood					
Second thinning after 16 years	40% fuel wood					
	40% fence poles					
	20% dead wood					
Expected life time of products (years)						
Fuel wood	1					
Fence poles	5					
Dead wood	10					
Timber	35					

Total net carbon accumulation was thus calculated for a period of 150 years for each region and system. To account for local variation in soil fertility and varying management practices we included a 20% variation in estimated annual increment ($m^3\ ha^{-1}\ yr^{-1}$). To use the model, it was assumed that: (i) the incorporation of the tree component in the system does not affect the total carbon dynamics of the rest of the system; and (ii) the site quality remains unaltered during the cycles.

The costs of carbon sequestration ($MgC\ ha^{-1}$ sequestered) for each system were calculated by summing the costs of: (i) the establishment and maintenance of the tree component within the system; and (ii) opportunity costs of lost benefits from the alternative system. The cost factors include labor, equipment, site preparation, site protection and planting stock for the first rotation of the trees. Costs for additional rotations are expected to be covered from the sale of the tree products of the first rotation. Labor costs are based on local minimum wages. The costs are represented on a per-hectare basis, as are the carbon sequestration potentials. Interest rates used to calculate costs to present date were 5 and 10%. Dividing the total costs by the net carbon-sequestration potential during the 150-year cycle determined the final cost of 1 Mg carbon sequestered in each selected system.

Results

Land available for a carbon-sequestration project

As a result of the workshops and project presentations in the communities, a total of 171.5 ha are made available for the first year of a carbon-sequestration project: coffee plantations (46.5 ha), fallow (32 ha), maize fields (8 ha), forest (5 ha), and pasture (2 ha) for the Tzeltal region, and fallow (35.5 ha), forest (25 ha), maize fields (12.5 ha), and pasture (5 ha) for the Tojolabal region. The communities possess considerable amount of communal forest areas; however, management decisions for these areas have to be taken by the whole community, which requires a general community assemblies. At this stage of the project, the members of the Pajal are undecided concerning the possible management alternatives for the communal forests and therefore prefer to start a project on an individual basis. Therefore, we have omitted the analysis of the carbon-sequestration potential of forest management. However, it is expected that, once the project has started, the rest of the communities will gain interest and will discuss the possibilities of sustainable management of the communal forests.

Table 4.2. *Forestry and agroforestry systems considered viable in Chiapas, Mexico.*

		Production level (m^3 ha^{-1} $year^{-1}$)		
System and treatment	**Planting distance (m)**	**I**	**II**	**III**
Tzeltal				
Live fence with *Cedrela odorata*	3	4.8	6.0	7.2
Coffee with *Cedrela odorata* or *Cordia alliodora* as shade trees	10 x 10	6.0	7.5	9.0
Taungya with *Cedrela odorata*: thinnings after 8 and 16 years (25% of stand)	10 x 3	11.9	14.9	17.9
Enriched fallow with *Cedrela odorata*, *Cordia alliodora* or *Calophyllum brasiliense*: thinnings after 8 and 16 years (25% of stand)	10 x 2	11.9	14.9	17.9
Tojolabal				
Live fence with *Pinus oocarpa*, *P. michoacana* or *Cypressus* sp.	3	3.2	4.0	4.8
Plantation of *Pinus oocarpa*, *P. michoacana* or *Cypressus* sp.	2 x 3	8.0	10.0	12.0
Taungya with *Pinus oocarpa*, *P. michoacana* or *Cypressus* sp.: thinnings after 8 and 16 years (25% of stand)	4 x 4	8.0	10.0	12.0
Enriched fallow with *Pinus oocarpa*, *P. michoacana* or *Cypressus* sp.: thinnings after 8 and 16 years (25% of stand)	7 x 2	8.0	10.0	12.0

Selected agroforestry systems

As a result of the workshops and field data collection, in which alternatives of forestry and agroforestry practices were discussed with the community members, a series of systems were evaluated. A total of five systems were considered viable, with local adjustments to species selection, planting arrangements and rotation. Table 4.2 shows the systems for each region that were considered technically viable, socially acceptable and economically feasible, including the planting arrangement of the tree component, the species to be used in the systems, the most important silvicultural treatments after planting and before harvesting, and the expected wood production of each system, according to three production levels: Level I for poor sites, Level II for medium fertile sites, and Level III for fertile sites. All systems require regular weed control and replacement of dead seedlings in the first year.

Carbon sequestration potential of the selected systems

The selected systems were used for carbon-offset calculations. Figure 4.1 illustrates the current annual increments used for each region during one rotation.

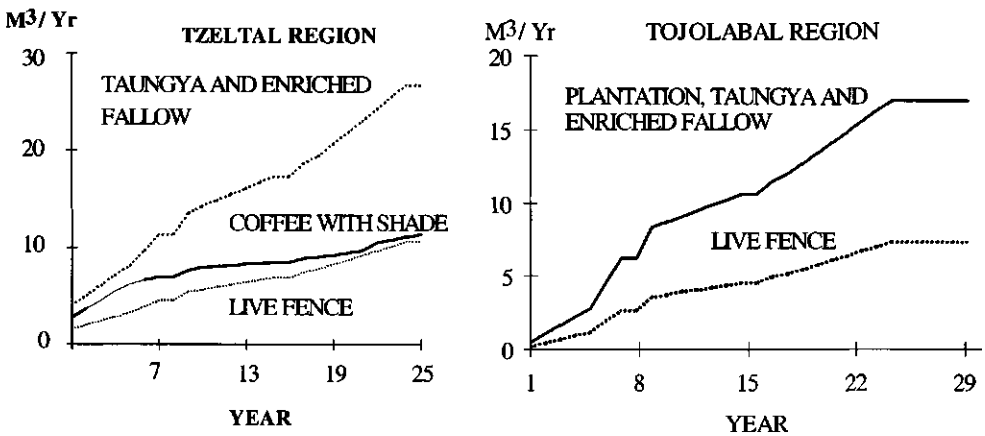


Figure 4.1. *Current annual increment of the selected systems during the first rotation.*

The estimations are based on analysis of field data on MAI and maximum heights measured in each region.

The carbon fluxes for each system and region are calculated for a 150-year period to overcome the tendency to overestimate carbon stocks in short-rotation forest systems (see also

Table 4.3. *Net accumulated carbon (MgC ha^{-1}), including product decomposition in 150 year forest rotation in Chiapas, Mexico.*

	PRODUCTION LEVEL					
	I		II		III	
System	Net carbon accumulation (MgC ha^{-1})	Net annual accumulation (MgC ha^{-1} $year^{-1}$)	Net carbon accumulation (MgC ha^{-1})	Net annual accumulation (MgC ha^{-1} $year^{-1}$)	Net carbon accumulation (MgC ha^{-1})	Net annual accumulation (MgC ha^{-1} $year^{-1}$)
Tzeltal						
1 Live fence	65.6	(1.66)	92.3	(2.19)	118.9	(2.72)
2 Coffee with shade trees	84.5	(2.11)	115.9	(2.76)	147.2	(3.41)
3 Taungya	214.6	(4.71)	276.8	(6.00)	338.9	(7.23)
4 Enriched Fallow	214.6	(4.71)	276.8	(6.00)	338.9	(7.23)
Tojolabal						
5 Live fence	26.7	(0.96)	39.1	(1.25)	51.5	(1.54)
6 Plantation	92.5	(2.58)	121.4	(3.27)	150.4	(3.97)
7 Taungya	94.5	(2.61)	123.9	(3.31)	153.3	(4.01)
8 Enriched Fallow	94.5	(2.61)	123.9	(3.31)	153.3	(4.01)

Nabuurs and Mohren, 1995). Table 4.3 presents the total accumulated carbon during the 150-year cycle and the average of the net annual carbon flux within each system during the first rotation, separated for the three production levels. The model outcome in terms of carbon allocation within the major pools are presented in Figure 4.2. Carbon-sequestration potential and allocation to the various pools varies highly between the systems and regions (26.7 to 338.9 MgC ha^{-1}), with the lowest potential for live fences in the Tojolabal region and the highest for the plantation system (*taungya* and enriched fallow) of the Tzeltal region. Differences of the systems within a region are due to differences in added tree densities, whereas the high potential of the systems in the Zteltal region, compared with the Tojolabal region, is due to the differences in current annual increments of the species to be included in the systems.

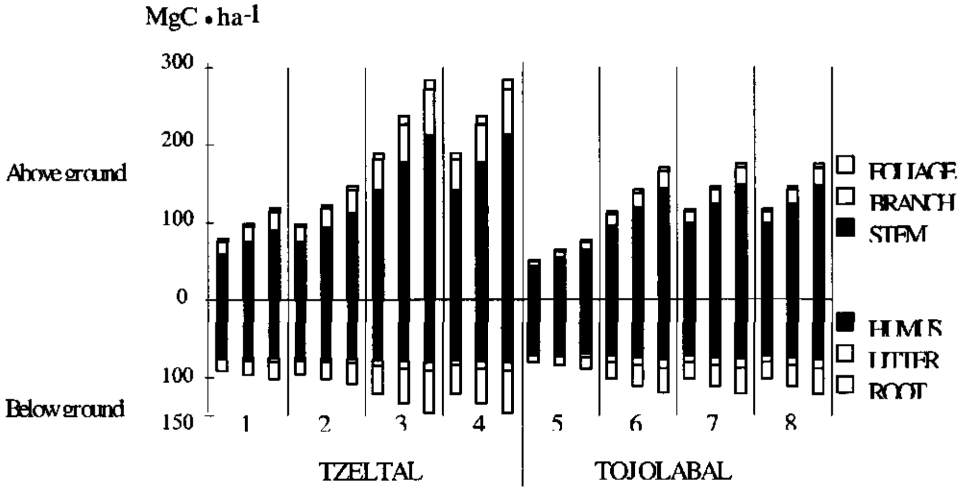


Figure 4.2. *Carbon allocation at the end of the first rotation. See text for the systems description. The left hand bar of each system represents the result of the simulation using production level I, the middle bar represents the results using production level II, and the right hand bar using production level III.*

Economic costs of the carbon sequestration

The estimated costs of carbon sequestration for each system are based on the discounted direct costs of implementing the systems, plus the discounted opportunity costs during the first rotation for those systems where land-use is diverted from agriculture or animal husbandry to forestry. It is expected that the farmer will be able to cover the costs of subsequent rotations from the income from timber production.

Table 4.4. *Estimated costs of carbon sequestration for selected forestry systems in Chiapas, Mexico, based on the total carbon accumulation for each system (production level II), calculated with 5 and 10% discount rates.*

	Live fence		Coffee		Taungya		Enriched fallow	
Tzeltal region								
Discount rates (%)	5	10	5	10	5	10	5	10
Direct costs ($US ha^{-1})	214	170	358	274	454	317	608	420
Opportunity costs ($US ha^{-1})					493	329	493	329
Total costs ($US ha^{-1})	214	170	358	274	947	646	1101	749
Total C-accumulation (MgC ha^{-1})	92.3	92.3	115.9	115.9	276.8	276.8	276.8	276.8
Costs ($US MgC^{-1})	**2.32**	**1.84**	**3.09**	**2.36**	**3.42**	**2.33**	**3.98**	**2.71**
	Live fence		Plantation		Taungya		Enriched fallow	
Tojolabal Region								
Discount rates (%)	5	10	5	10	5	10	5	10
Direct costs ($US ha^{-1})	98	58	1176	966	386	248	952	816
Opportunity costs ($US ha^{-1})	0	0	177	113	177	113	177	113
Total costs ($US ha^{-1})	98	58	1353	1079	563	361	1130	929
Total C-accumulation (MgC ha^{-1})	39.1	39.1	121.4	121.4	123.9	123.9	123.9	123.9
Costs ($US MgC^{-1})	**2.50**	**1.47**	**11.15**	**8.89**	**4.54**	**2.91**	**9.12**	**7.50**

In Table 4.4 we present the costs of carbon sequestration for each system, using the expected carbon sequestration for production Level II. Costs are calculated on a per-hectare basis and include costs for establishment, maintenance, silvicultural treatments and harvesting for year 0 to 25 for the Tzeltal region and for year 0 to 30 for the Tojolabal region. Costs are presented separately for the discount rates of 5% and 10%, which are used to calculate the costs to the present date. Total cost MgC^{-1} ha^{-1} varies between $US1.84 and $US3.98 for the systems selected for the Tzeltal region and between $US1.47 and $US11.15 $US for the systems in the Tojolabal region. The differences in costs within the same region are due to differences in costs of establishment and opportunity costs, whereas the differences between the two regions are due to differences in carbon-sequestration potential.

Summary and conclusions

In Mexican rural communities, the different models of social organization affect the outcome of a farm forestry project definition, as illustrated in this study through the experiences of a farmers´ productive organization, with individual members from various communities. The structure of decision-making, however, implies that members lack decisive power in relation to community matters. The advantage of working with such a farmers´ organization is that a project can be extended rapidly to various communities, after a successful pilot stage. The disadvantage is that the project can only work on individually based systems. Many farmers are interested in starting small-scale forestry or agroforestry activities on their own land, but consensus for managing communal areas of natural forest is much harder to achieve, since these have to pass community-level decision-making structures. Social and political divisions often require some degree of resolution before control of *ad hoc* or legal exploitation of the common resource can be achieved. In the agroforestry project outlined, individuals plant trees in their coffee plots, fallow areas, pastures and maize fields, but the participants work cooperatively to organize seed-beds, nurseries and the marketing of wood products. This use of the reciprocal work-system áyuda-por-ayuda´ takes advantage of economies of scale and strengthens the new project by embedding it in the tradional labor-exchange system.

The current project will probably benefit the male members of Pajal and possibly their immediate families. To promote a community-level forestry project, however, there would need to be a greater distribution of benefits. Women's agroforestry projects for home gardens could increase the possibilities of benefits for children and female-headed households. Community-administered work teams for tree planting; management and harvesting on communal lands could distribute salaries to families without land and a percentage of profits for all community members. In all cases, the use of traditional labor-exchange systems, decision-making for communal-land management and equitable distribution of benefits will be essential ingredients for a successful carbon-sequestration program through community forestry.

To estimate the effect of an agroforestry project on the regional balance of carbon fluxes, the simulation models tested have certain shortcomings. The landscape-based models to estimate carbon fluxes through different land-use scenarios lack a dynamic approach to calculate the fluxes through existing systems (e.g. Makundi et al., 1995), while stand-based carbon-flux models (e.g. CO_2FIX) lack a quantitative landscape component. Cairns et al. (1995) have shown that significant differences in carbon-flux calculations can occur, if the carbon dynamics in, for example, secondary and primary forests are included. Models that calculate carbon fluxes under different land-use scenarios and which consider the growth potential of all components within the landscape are required to monitor carbon fluxes on a landscape basis of community-forestry projects.

The carbon-sequestration potential of different agroforestry systems depends highly on the quantity of trees to be managed in the system. However, low-tree-density systems, such as live fences and commercial trees, combined with (perennial) crops that require shade (such as coffee and cacao) are relatively inexpensive alternatives that can contribute substantially to the carbon budget on a regional basis, since the cost of establishment and maintenance are low and opportunity costs are negligible.

Although excluded from our analysis, the improvement of seminatural communal-forest management is likely to yield additional significant net economic benefits for the communities because of: (i) the relatively low initial costs of a management project, since the system is already established; (ii) the fact that standing timber is a capital asset, which can help finance development; (iii) the lower opportunity costs (not replacing agricultural systems); and (iv) the higher conservation values (biodiversity, watershed protection, soil conservation). However, since the forests already have relatively high levels of carbon stored in the system, the forest-management options would represent a low carbon-sequestration potential but a high potential for carbon sinks. However, discussions about the economic value of conserving carbon sinks are still unresolved.

The costs and benefits associated with bringing a particular forested area under sustainable management would certainly depend upon particular circumstances at each location. Where forests in good condition exist and are currently subject to low levels of exploitation, there may be immediate short-term benefits, in terms of increased annual harvests and increased productivity, when improved forestry practices are to be implemented. However, where forests are severely damaged, the costs of reparative actions may outweigh the benefits for several years.

As forestry and agroforestry projects concern long-term investments, with final products to be obtained after more than 20 years, mechanisms are required that guarantee the financial aspects of these projects. Combining systems that provide commercial products, such as wood for timber, and long-term ecological services, such as carbon sequestration, can generate the necessary capital for the investments in the systems.

The long-term characteristics of these type of projects also require stable organizational structures that can support decision-making, sound financial management and good planning, including consulting local people and selecting appropriate species and management regimes for particular sites and social conditions. Whether the proposed project involves sustainable management of a large area of communal forest or establishment of small-farm woodlands, the quantity, quality, timing, and distribution of both inputs and outputs must be carefully considered (De Jong et al., 1995). As such, communities, groups, and individual farmers have to take responsibility for the projects, not only in planning, but also for implementation, management and monitoring of progress, which implies a structuring of the project around existing social organization patterns.

Acknowledgments

The authors thank participating institutions and organizations in their research, especially Unión de Crédito Pajal Ya Kac'tic for workshop facilities, and access to the communities, and El Colegio de la Frontera Sur (ECOSUR) for assistance. They also gratefully acknowledge the assistance of the following for the collection of data: Jerónima Alvaro, Martha Gómez, Manuela Moreno Moreno, Manuela Moreno Jiménez, Juana Moreno, Alejandra López, María de Tránsito, Angelica García, Teresa Pérez, Regina Pérez y Pascuala López, Manuel Moreno Hernández, Juan Moreno Gómez, Gilberto Alvaro Jiménez, Mariano Moreno Jiménez, Mariano Moreno Moreno, Jerónimo Gómez Pérez y Nicolas Hernández Pérez; Julian Pérez López, Pablo Santís García, Julio Jiménez Román, Antolino Pérez Hernández, Ventura Aguilar Santís, Abelardo García Pérez, Ramón Vázquez Vázquez, José R. Vázquez Vázquez, Caralampio Gullén López, Fernando López Aguilar. The study was financed by The National Institute of Ecology of the Secretary of Environment, Natural Resources, and Fisheries, the

European Community (Ref B-73014/92/412/9) and project R6320 of the UK Department for International Development, Forestry Research Programme.

References

Adams, R.M., Adams, D.M., Callaway, J.M., Chang, C.C., and McCarl, B.A. 1993. Sequestering carbon on agricultural land: social cost and impact on timber markets. Contemporary Policy Issues XI: 76-87.

Andrasko, K., Heaton, K., and Winnett, S. 1991. Estimating the costs of forest sector management options: overview of site, national and global analyses. In: Howlett, D. and Sargent, C. (eds.). Proc. Tech. Workshop to Explore Options for Global Forest Management. April 1991, Bankok, Thailand. London IIED.

Cairns, M.A., Baker, J.R., Shea, R.W., and Haggerty, P.K. 1995. Carbon dynamics of Mexican tropical evergreen forests: Influence of forstry mitigation options and refinement of carbon-flux estimates. Interciencia 20(6): 401-408.

De Jong, B.H.J., Montoya-Gómez, G., Nelson, K., Soto-Pinto, L., Taylor, J., and Tipper, R. 1995. Community forest management and carbon sequestration: A feasibility study from Chiapas, Mexico. Interciencia 20(6): 409-416.

Detwiler, R.P. and Hall, C.A.S. 1988. Tropical Forests and the Global Carbon Cycle. Science 239: 42-47.

Dixon, R.K., Andrasko, K.J., Sussman, F.G., Lavinson, M.A., Trexler, M.C., and Vinson, T.S. 1993. Forest Sector Carbon Offset Projects: Near-term Opportunities to Mitigate Greenhouse Gas Emissions. Water, Air and Soil Pollution 70: 561-577.

Forests Absorbing Carbondioxide Emission, FACE, 1994. Annual Report 1993, Arnhem, The Netherlands.

Garcia, E., 1973. Modificaciones al sistema de classificación climática de Köppen. UNAM, México D.F. 246 pp.

Gay, C., Imaz, M., Wiel, S., Wisniewski, J., Ruíz-Suarez L., and Goldberg, B. 1995. Workshop Summary: Regional Workshop on Greenhouse Gas Mitigation Strategies for Latin American Countries. Cancún, México, 10-13 July 1995. Interciencia, 20(5): 3-15.

Houghton, R. A., Unruh, J., and Lefebvre, P.A. 1991. Current land use in the tropics and its potential for sequestering carbon. In Howlett, D. and Sargent, C. (eds.) Proc. Tech. Workshop to Explore Options for Global Forest Management. April 1991, Bankok, Thailand. London IIED.

Makundi, W.R., Sathaye, J., and Ketoff, A. 1995. COPATH: A spreadsheet model for the estimation of carbon flows associated with the use of forest resources. Biomass and Bioenergy, 8 (5): 369-380.

Marland, G. 1988. The prospect of solving the CO2 problem through global reforestation. DOE/NBB-0082 Washington, DC. United States Department of Energy, Office of Energy Research.

Masera, O.R., Bellon, M.R., and Segura, G. 1995. Forest Management Options for Sequestering Carbon in Mexico. Biomass and Bioenergy 8(5): 357-367.

Mohren, G.M.J. and Klein-Goldewijk, C.G.M. 1990. CO2FIX, model listing, Research Institute for Forestry and Urban Ecology, Wageningen, The Netherlands, Report nr. 614, 96 pp.

Montoya, G., Soto, L., De Jong, B., Nelson, K., Farías, P., Taylor, J.H., and Tipper, R. 1995. Desarrollo forestal sustentable: Captura de carbono en las zonas tzeltal y tojolabal del estado de Chiapas. Cuadernos de trabajo Nº 4, INE, CIES, San Cristóbal de las Casas, Chiapas. 79 pp.

Nabuurs G.J. and Mohren, G.M.J. 1993. Carbon fixation through forestation activities: A study of the carbon sequestering potential of selected forest types, commissioned by the Foundation Face. Inst. for Forestry and Nature Research (IBN-DLO), Wageningen, The Netherlands, IBN Research Report 93/4. 204 pp.

Nabuurs, G.J. and Mohren, G.M.J. 1995. Modelling analysis of potential carbon sequestration in selected forest types. Canadian Journal of Forest Research 25: 1157-1172.

Sedjo, R.A. and Solomon, A.J. 1991. Climate and forests. In Roenberg et al., N.S. (eds.) Greenhouse warming: abatement and adaptation. Workshop proceedings. Washington, DC, WRI.

Swisher, J.N. 1991. Cost and performance of CO2 storage in forestry projects. Biomass and Bioenergy 1: 317-328.

Swisher, J. and Masters, G. 1992. A Mechanism to Reconcile Equity and Efficiency in Global Climate Protection: International Carbon Emission Offsets. Ambio 21(3): 154-159.

Trexler, M.C. 1991. Minding the carbon store: weighing US Forestry Strategies to slow Global Warming. World Resources Institute, Washington DC.

5. AN ECONOMIC ANALYSIS OF THE POTENTIAL FOR CARBON SEQUESTRATION BY FORESTS: EVIDENCE FROM SOUTHERN MEXICO[3]

Introduction

As concern has grown about the possible impacts of climate change due to anthropogenic greenhouse gas emissions, there has been considerable interest in the potential for increasing the storage of carbon (C) in terrestrial vegetation through forest conservation, afforestation and other methods of land management. Several studies have indicated that the global potential for enhancing C storage in forest and agricultural ecosystems may be as much as 60 to 90 Petagrams ($=10^{15}$) of C. (Dixon et al., 1991; Brown et al., 1995).

International measures to control greenhouse gas emissions are likely to include market-based mechanisms that will allow countries to trade in emission reductions in order to comply with their commitments under the UN Climate Change Convention. The question of how forests are to be included in these so-called flexible mechanisms is currently under consideration by the parties to the Convention. The technical options for sequestering carbon through forestry measures include: the conservation and management of existing closed forests; the restoration of degraded or secondary forests; and the establishment of plantations, agroforestry systems and new forests in open areas (Dixon et al., 1996; Masera et al., 1995; Sathaye and Ravindranath, 1997).

Preliminary evidence from a number of specific forestry projects that have been financed on the basis of the expected sequestration effect indicate that the cost of sequestration by forestry or other forms of land management is relatively low in comparison with many engineering solutions to CO_2 emission reductions (De Jong et al., 1995). However, since cost estimates rely to a large extent upon data relating to specific projects, and since current information about the land available for carbon sequestration takes little account of its suitability or of competing uses, doubts remain about the likely costs of sequestering large quantities of carbon. As soon as credits from C sequestration become a tradable commodity under a future emissions control regime, as now being close to implementation in The Netherlands, the supply response to changes in prices for sequestration, as expressed in \$US MgC^{-1}, would be critical in determining the total level of C uptake achieved by the system as a whole.

[3] *Published as: De Jong, B.H.J., Tipper, R., and Montoya-Gómez, G. 2000. An economic analysis of the potential for carbon sequestration by forests: evidence from Southern Mexico. Ecological Economics 33(2): 313-327.*

Since much of the land area in the tropics is effectively managed or influenced by a wide range of semi-subsistence farmers and shifting cultivators, their response to various measures will be a key factor in determining the feasibility and cost of carbon sequestering initiatives.

In this paper we present the results of a study to estimate the response of small farmers and communities in southern Mexico to switch from current land use to forestry and agroforestry. Based on the estimation of the level of required incentives we calculate the potential supply and cost of C sequestration of a forestry program to be implemented in a land area of about $0.6 * 10^6$ ha. We assume that farmers will switch to forestry and agroforestry from the point where the incentives are higher than the net present cost (NPC) to implement the alternative land use systems. Experiences to date with the Scolel Té Pilot Project indicate that farmers are generally eager to enter a forestry program, even with lower incentives than estimated (Scolel Té, 1997).

Key questions that are dealt with in this paper are:

- What is the biological potential for carbon sequestration of forest management and agroforestry systems preferred by farmers in developing countries?
- What are the costs and benefits of adopting such systems for the farmers?
- What are these costs in terms of the carbon that can be sequestered?

Methods

Study area

The Central Highlands of Chiapas (Los Altos, 607,500 ha, 1500-2900 m a.s.l., Figure 5.1), southern Mexico, contain various forest formations with a very high biodiversity resulting from interactions among biological, geological, edaphological, climatological and anthropogenic factors. The most extensive forest formations are pine forest, pine-oak forest and oak forest (Miranda and Hernández, 1963; Breedlove, 1981; González-Espinosa et al., 1995). The regional climate is subtropical to temperate, subhumid (Holdridge, 1967). Mean annual rainfall varies between 1100-2000 mm, of which more than 80% falls between April and November (Garcia, 1981). The soils are predominantly derived from calcareous rocks, and include cambisols, leptosols, luvisols, ferrosols, nitisols, lixisols, acrisols, and feozems (De Jong et al., 1999).

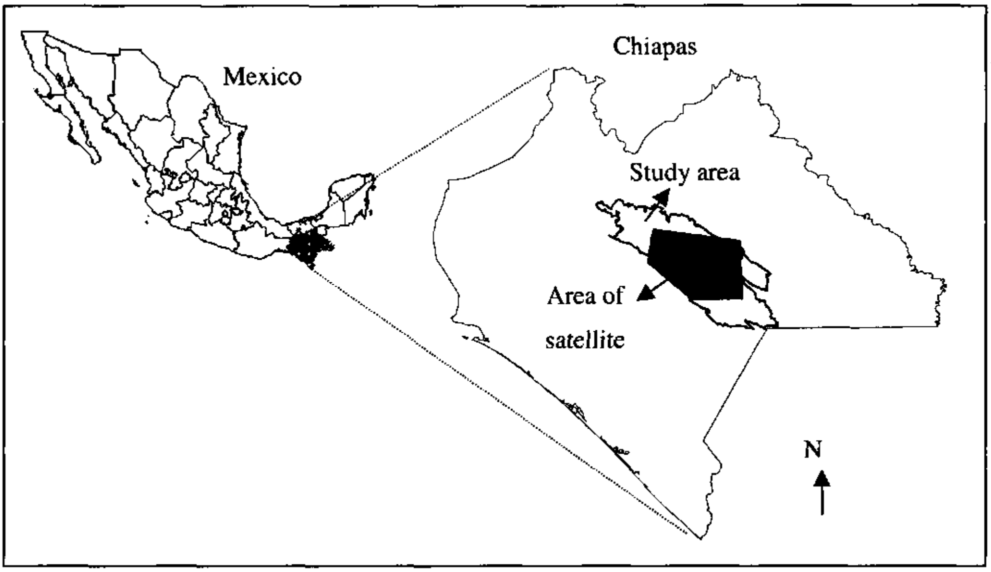


Figure 5.1. *Location of the study area.*

The historical process of land-use change has been complex, involving the expansion of cultivated land, extraction of selected high value forest products, and grazing of sheep and cattle (González-Espinosa et al., 1995; De Jong and Ruíz-Díaz, 1997). About 80% of the territory is under a communal form of tenure known as the ejido. Within the ejido, families - members of the ejido - hold agricultural land in private usufruct, whereas forestland, pastures and barren areas are generally kept and managed as common resources.

Maize is by far the most important crop in the region, accounting for about half of the total value of agricultural production (INEGI, 1993). The generic Mayan term for the maize field is milpa. Today the milpa systems of Chiapas are changing rapidly due to a combination of market forces, land tenure, and population pressure. In general, land has become increasingly scarce whereas capital and agricultural inputs have become more readily available.

Baseline Land Use / Land Cover (LU/LC) change dynamics

The C-flux impact of a given intervention must be compared with a baseline or non-intervention scenario to provide a correct estimate of the effect of the project. In the case of existing forests, we are generally concerned with conserving or enhancing the current C stock through forest conservation and management. In the case of forest restoration, afforestation or agroforestry measures, we are concerned with increasing the stock of C on a site.

To understand the historical trends in land-use change and associated carbon fluxes we used a series of land cover maps developed by Ochoa-Gaona, derived from various satellite images, aerial photographs and surveys (Tipper et al., 1998; Ochoa-Gaona and González-Espinosa, 2000). We compared a LU/LC map of the 1970s (INEGI 1984, 1987, 1988), MSS images from around 1974, 1984 and 1990 and Landsat TM images from 1996. The following LU/LC classes could be distinguished in the images: Closed oak and montane forest, pine-oak forest, pine forest, open pine forest, tree fallow, shrub fallow (thicket), open grassland, agriculture (in which we included bare soil and settlements). We assigned C densities, collected in the field, to each LU/LC class and multiplied the surface area of each LU/LC class with their respective average C density value for the years 1974, 1984, 1990, and 1996. In projecting rates of carbon fluxes into the future, it should be noted that not all the carbon in the system is vulnerable to loss. Whereas most of the aboveground portion of the carbon stores, plus some of the root matter and leaf litter are susceptible to rapid loss, a large proportion of soil carbon remains in situ a long time when land-cover changes from forest to open land. The amount of stable humus may vary due to soil type, land use history, precipitation and vegetation, among others. Comparing total C pools for each interval we estimated the historical rate of C storage depletion, assuming a stepwise process, in which C densities within each LU/LC class remained constant during the period analyzed. Default baseline scenarios were established as a fixed frame of reference by extrapolating the yearly rates of C loss into the future for a 50-year period.

The costs of transferring current land use to forestry and agroforestry

We constructed income-expenditure profiles for 12 alternative interventions for forests, agricultural, pasture and fallow lands, based on the recent experiences of the Scolel Té Pilot Project (Scolel Té, 1997; De Jong et al., 1998). Details of the inputs required for forest management were collected from various sources, including forestry organizations in the states of Oaxaca and Campeche (Tipper et al., 1998).

The predisposition of farmers to switch land use from the current one to the cultivation of trees for timber or other purposes is determined by a mixture of economic, social and cultural factors (Tipper et al., 1998). These include costs of implementation, lost opportunities, socio-technical implications, and expected benefits from product sales. The cost of carbon sequestration was calculated as follows for a period of 70 years (expressed in Mex. Pesos and converted into $US using a 7.70 pesos $\$US^{-1}$ exchange rate):

$$C_c = C_i + C_m + C_o - B_p$$

where:

C_c = cost of carbon sequestration, discounted to present value

C_I = implementation cost (initial establishment of the forestry system)

C_m = cost of management and services (including project promotion and training), discounted to present value

C_o = opportunity cost, (land rent value) discounted to present value

B_p = revenue from timber sale and labor savings, discounted to present value

Project monitoring is considered as a continuous assessment of the functioning of project activities, and as such the costs of monitoring is included in the implementation and management costs (De Jong et al., 1997). In our analysis we excluded costs for verification of the projects' performance (Swisher, 1992), as these will depend on measurement standards and allowable limits of error, which have not yet been agreed upon internationally (MacDicken, 1997). We also excluded transaction costs, as these will largely depend on how the Kyoto Protocol will be implemented internationally and in Mexico.

To calculate implementation costs we estimated the inputs necessary to establish and maintain the systems and the operational costs of each management option, based on the experiences of the inputs required in the Scolel Té Pilot Project, the only project implemented since 1997 in Mexico (Scolel Té, 1997), and as such our sole possible reference.

An important economic determinant of farmers' predisposition to change land use is the rent foregone by converting current land use to forestry (opportunity cost). To estimate the variation in net income per hectare from maize production, we interviewed 53 farmers from 12 communities to obtain a range of values of inputs and outputs (Tipper et al., 1998), which we divided into four quartiles, each with an equal number of farmers.

Currently, the use of communal forests and secondary vegetation is not restricted. To convert communal land use, such as extensive grazing, timber and fuel wood extraction, to sustainable levels implies a resource opportunity cost to communities. Hellier (1996) and Konstant (1997) tried to assess the utilities derived by farmers from secondary vegetation and forests management in several communities within the study area but encountered considerable problems in precisely calculating the level of extraction of different non-timber forest products. Based on their results we estimated this constraint for secondary vegetation on the assumption that the benefits lost by controls on their current exploitation will be around 60% of the opportunity costs associated with transferring land out of subsistence agricultural production to farm forestry. We assumed that unmanaged oak, montane and pine-oak forests will face high opportunity costs due to the additional value of fuelwood, charcoal and bromeliads that are currently extracted at significantly higher rates than the apparent level of "sustainable yield" in these forest types (Golicher, pers. comm., 1997). The calculation of rent foregone from cattle ranching is primarily based on the estimated annual weight gain per calf ha^{-1} for a typical ranching system with around 1 head of cattle ha^{-1}, and a reproduction rate of 0.6 calves yr^{-1} (INEGI, 1993).

The costs of building community management skills related to forest management are difficult to assess *ex-ante*, therefore we resorted to estimates of the time and effort required to develop the necessary level of organization in communities representing four quartiles within a difficulty spectrum. Our estimated values are again based on the experiences of the Scolel Té project (Scolel Té, 1997;Tipper et al., 1998). At the upper end of the difficulty spectrum are communities where it is virtually impossible to establish a forest management program given the apparently intractable nature of internal communal divisions and conflicts. At the bottom end of the difficulty spectrum are communities that already have considerable positive experience of community managed projects - such as communal stores and transport co-operatives.

Large-scale investment in forestry to sequester carbon will face rising cost functions, when lands with higher productivity and/or opportunity costs increasingly enter the program (Moulton and Richards, 1990), or when project promotion and forestry training are increasingly required (Tipper et al., 1998). As such, we used four cost levels for both opportunity and socio-technical costs.

Carbon sequestration estimation

We developed a dynamic model to estimate the C fluxes through the proposed land management systems via an accounting procedure, similar to the CO_2FIX model (Nabuurs and Mohren, 1993). The model can accommodate growth of up to three species aggregates within a specific site, such as fast growing secondary species, slow growing primary species and understory species (De Jong et al., 1998). Expected growth curves (Cannell and Milne, 1995) and standing biomass drive the growth expectation of the species aggregates (De Jong et al., 1998). The values of the variables of the model were based on field data of maximum volume observed at any one site (dependent on site quality) and measured growth of the species groups (dependent on site quality and species characteristics). Data on C densities of the current LU/LC types have been gathered in the study area, with numerical data for the main C pools: soil organic matter, roots, herbaceous plants, shrubs, trees, woody debris and litter. The average C content of the various pools in each land-use class was used for the initial C values in the simulation model. For each current land management system, up to three alternative land management options were designed, according to systems developed in the Scolel Té Pilot Project (Scolel Té, 1997; De Jong et al., 1998). Soil C dynamics were simulated with data from comparable areas, available in the literature (Nabuurs and Mohren, 1993, 1995).

Farmers, participating in the Scolel Té project, are applying a variety of land management techniques, which provided the basis for the alternative options. For example, according to the farmers the most viable method of increasing biomass in shrub fallow is the establishment of plantations via enrichment planting. In the case of high tree fallow, they consider that the existing vegetation can be managed through liberation thinning and weeding with relatively little additional planting. The following management options were used for modeling (see also De Jong et al., 1998; Tipper et al., 1998):

- **Oak and montane forest**: Conservation areas and extraction of non-timber forest products.
- **Pine-oak and pine forests** (closed and open): Integrated community management, including forest conservation and restoration; selective harvesting in compartments.
- **Tree fallow and thickets:** Sustainable oak coppicing for firewood and charcoal; restoration by natural regeneration or with interplanting, depending on seed bank and genetic quality of current stand; selective harvesting.
- **Agriculture and pasture:** Agroforestry systems, such as living fences, fruit orchards, fodder banks, mixing with N-fixing trees (cf. Alnus sp., Leguminosae spp); plantations; organic agriculture.

Carbon fluxes in each alternative management option were approximated over 100 years (Nabuurs and Mohren, 1995). The 100-year average C-stock increase was calculated according to the following formula:

$C_{acc} = (\Sigma_{100} (C_i - C_0)) / 100$ (MgC ha^{-1})

where:

C_{acc} = Long-term average accumulation of C of the alternative system

C_i = C density of the alternative system in year i (i = 1 to 100)

C_0 = Initial C density of the LU/LC class.

The net sequestration potential of each management option was calculated, adding the expected low, medium or high baseline carbon emission of the same LU/LC class to the long-term average increase in C of each LU/LC type.

Model of adoption of forestry systems
A spreadsheet model was designed to compile the areas of each LU/LC class in ha, average C storage in Mg ha^{-1}, and economic inputs and outputs of the alternative management options for each vegetation type (Figure 5.2).

Cost-benefit flows were discounted to present value to provide an estimate of the net present cost ha^{-1} (NPC) of implementing the alternatives for each quartile of each vegetation type. The discount rate used in this assessment is the farmer's own rate of time preference, which may differ considerably from the commercial rate. We used a default discount rate of 10%, but explored the effect of varying this rate from 5% to 40% through a sensitivity analysis. Varying the input values of other parameters, such as labor cost and product prices did not influence the model output significantly and the results of this sensitivity analysis are not reported here (Tipper et al., 1998). We assumed that if the sequestration purchase price were higher than the NPC for a particular quartile - management system - vegetation type combination then farmers would choose to enter the scheme and implement the new system.

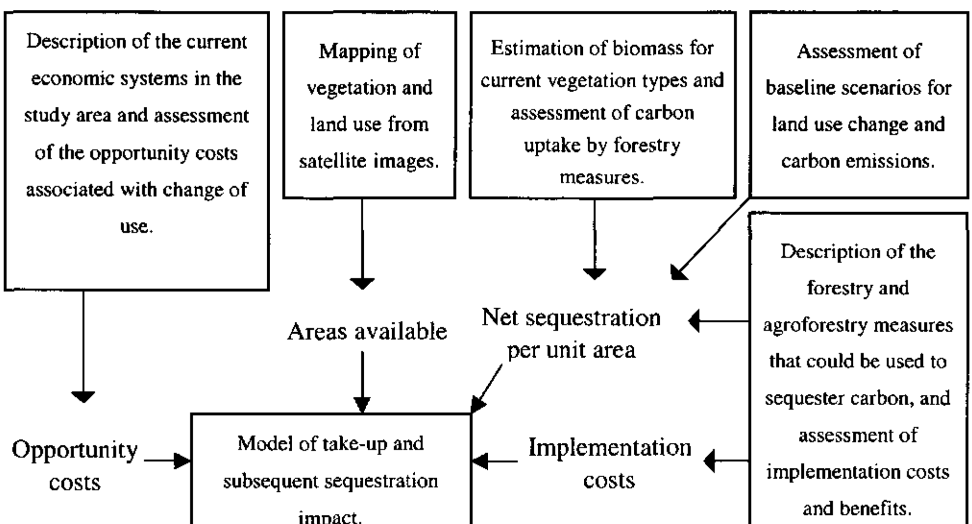


Figure 5.2. *Outline of the information flow to calculate the sequestration potential of an incentive/service payment-based forestry program.*

Results

Baseline

Comparing the LU/LC statistics obtained from the satellite image interpretation of a sub-area (308,000 ha = 49% of the whole study area, Figure 5.1), we found that during the late 1970's and early 1980's the total C stocks were depleted at a rate of approximately 1.7% yr^{-1}. This rate decreased to around −0.1% yr^{-1} during the late 1980's, and then increased again to about 2.5% yr^{-1} in the 1990s (Figure 5.3). If we assume that LU/LC change in this sub-area will proceed at the rate observed during the period between 1984 and 1990, then the C stocks will rise slightly. However, if changes proceed at the rate experienced between 1990 and 1996, then the future carbon stock will decline sharply. The overall average annual C depletion for the 1974 to 1996 period was estimated at 1.4%. Comparing the vegetation cover of 1996 with the *Uso de Suelo y Vegetación* maps of 1975 (INEGI, 1984, 1987, 1988) of the whole area (624,600 ha) resulted in virtually no change in carbon content. Either the sub-area we used in the satellite image comparison was not representative for the whole area, or the data of the *Uso de Suelo y Vegetación* maps were not consistent with those obtained from the interpretation of the satellite images from the 1970s.

Given this uncertainty, we used a conservative range of baseline emissions through LU/LC change dynamics of 0.5 to 1.5% yr^{-1}, from "Low (0.5%)" to "Medium (1%)" and "High (1.5%)" future C depletion. The medium rate we used in our baseline estimations matches the average decline observed during the first 16 years, whereas the high rate corresponds more or less to the 22-year average C decline (Figure 5.3).

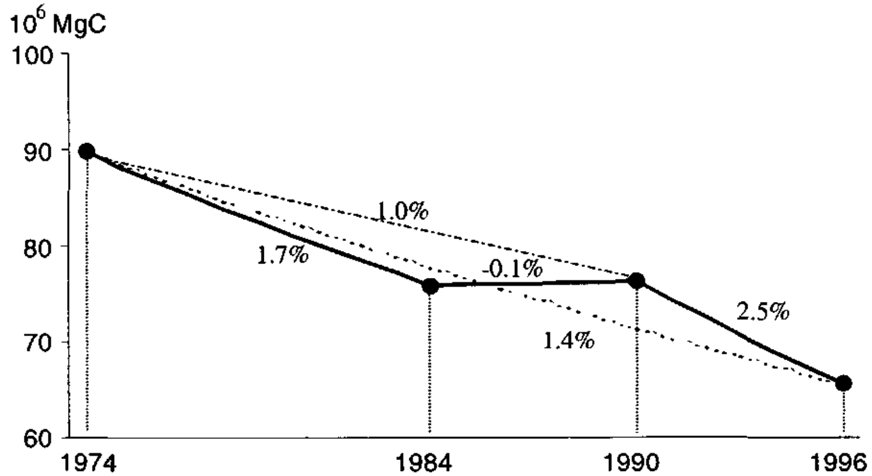


Figure 5.3. *Historical carbon depletion (in 10^6 MgC and % annual change) in a sub-area of 308,000 ha, based upon data from Landsat MSS (1974, 1984, 1990) and Landsat TM (1996) images.*

Costs and benefits

The main elements in the cost of establishing community forestry systems are costs for inventories, management planning and stock protection. Their sum was estimated to range from \$US186 ha^{-1} for oak and montane forests to \$US217.5 ha^{-1} for open pine forests (Table 5.1).

It is thought that open forests reduce the capacity for natural regeneration, and therefore requires more intensive re-planting. The direct opportunity costs for forests were estimated to be lower than those for agriculture and pastoral land and ranged from \$US0 yr^{-1} for the lowest quartile of open and closed pine forests to \$US130 yr^{-1} for the highest quartile of oak and pine-oak forests (Table 5.2).

Table 5.1. *Costs and benefits of the management options.*

Land use / Land Cover Types	Establishment including labor	Operational and maintenance costs including project monitoring			Timber harvest 100 year cycle
	($US)	Costs ($US yr^{-1})	Labor input (d ha^{-1} yr^{-1})	Total ($US ha^{-1} yr^{-1})	(in m^3)
Oak and Mountain Forest	186	38.3	10	64.3	300
Pine-oak Forest	208.5	37	10-15	63 - 76	282.5
Pine Forest	192	48.7	15-20	87.7-100.7	280
Open Pine Forest	217.5	48.7	20	100.7	227.5
Tree Fallow	223.4	36.4	15-25	75.4 - 101.4	235
Thicket	285.7	37.7	15-25	76.7 - 102.7	305
Pasture	282.5	13.1	10-20	39.1 - 65.1	267.5
Agriculture	212.2	10.1	10-15	36.1 - 49.1	221.5

The socio-technical costs associated with developing new community-based social structures for forest management were expected to add significantly to the real cost of establishment. In some communities (the lower quartile) these costs were thought to be only $US52 ha^{-1}, but in the third quartile this cost was estimated at $US325 ha^{-1}. The fourth quartile represents the most difficult communities, where establishment of communal management systems was considered unfeasible (Table 5.2). On individual land holdings, the main input required to establish (agro)-forestry systems is labor for weeding, land preparation and planting. The estimated cost of establishment of farm forestry systems on the individual land holdings varied between $US212 on agricultural land to $US286 to convert thickets. The annual maintenance in agriculture-based systems is expected to be low as labor inputs to trees can be combined with those to annual crops, varying from $US36 to 49 yr^{-1}.

Table 5.2. *Annual opportunity costs (in $US yr^{-1}) to convert current land use practices into C-sequestration management alternatives and one-off socio-technical costs for community capacity building in forest management (in $US).*

Production System	Opportunity Costs ($US yr^{-1})			
	1st Quartile	2nd Quartile	3rd Quartile	4th Quartile
Milpa Agriculture	0	140.2	250	358.5
Cattle Ranching	39	78	107	152
Thicket	0	85.5	150	215
Tree Fallow	0	85.5	450	215
Oak and montane forest	6.5	13	65	130
Pine-oak forest	6.5	13	65	130
Pine forest	0	6.5	26	65
Open pine forest	0	6.5	26	65
Socio-technical costs for community capacity building in forest management ($US).	52	104	325	Not feasible

Maintenance of the management systems for the remaining individual land holdings varied between $US39 yr^{-1} to convert pasture to $US103 yr^{-1} to transfer thickets to forest plantations (Table 5.1).

To calculate the annual opportunity cost to replace agricultural production systems, farmers provided estimates of average yields and costs of production, from which we derived the net income per hectare. In the lowest quartile of the sample, the net income was less than zero; i.e. maize was produced at a loss. We therefore assumed that the rent foregone for this quartile was zero. We also assumed that the new systems would vary in tree planting intensity - from low and medium intensity planting in "living fences" and windbreaks, agroforestry systems such as taungya, where agricultural crops are grown for part of the forestry rotation (before the trees canopy closes), - to intensive plantations, where agricultural activity is completely replaced by forestry on each unit of land. The opportunity costs for successive quartiles are therefore lower than in the case where all agricultural activity would be entirely replaced, varying from $US0 to 358.50 yr^{-1} (Table 5.2).

To estimate opportunity costs for pasture conversion, we calculated that typical ranching systems have around one head of cattle ha^{-1}, and a reproduction rate of about 0.6 calves yr^{-1}. The annual weight gain per calf is around 100 kg, at a value of approximately 8 pesos kg^{-1} on the hoof. The rent foregone per hectare was therefore estimated to vary from $US39 to 152 yr^{-1} (Table 5.2). Despite the poor returns, cattle and sheep continue to play an important role in farming systems as they represent a form of savings that requires only a modest amount of maintenance (Parra-Vázquez, 1989).

The expected benefits of timber harvesting for the 100-year simulation varied between 221 m^3 ha^{-1} for systems replacing agriculture to 305 m^3 ha^{-1} for systems replacing thickets (Table 5.1). The expected timber benefits from the forest management options varied between 280 and 300 m^3 for closed forests and between 227 and 235 m^3 for open pine forest and tree fallow respectively. The differences between these options were mainly due to differences in the harvesting potential for the first 20 years. While harvesting can start from the onset of the project in existing closed forests, the forest options for open land or disturbed forests only start producing wood 15 to 25 years after tree planting.

The overall average NPC of viable land management options for C sequestration was less than $US15 MgC^{-1} (Figure 5.4). Forest management options represent the lowest costs, excluding the 4th quartile, where socio-political constraints are insurmountable (Table 5.2). They can be implemented at less than $US15 MgC^{-1} for the first three quartiles (Figure 5.4).

Carbon sequestration by replacing current agricultural practices, thickets and pasture represents the highest costs. To replace agriculture, only marginally profitable systems from quartile 1 and 2 are expected to enter in a C sequestration forestry program (Figure 5.4). The early product revenues from closed forest management offset much of the opportunity and implementation costs, giving a total cost of between $US2 and 13 MgC^{-1}, with lowest costs associated with oak forest management and highest with open pine forest. While open pine forests have low opportunity costs, the implementation costs are high and revenue from product sale is low (Table 5.1).

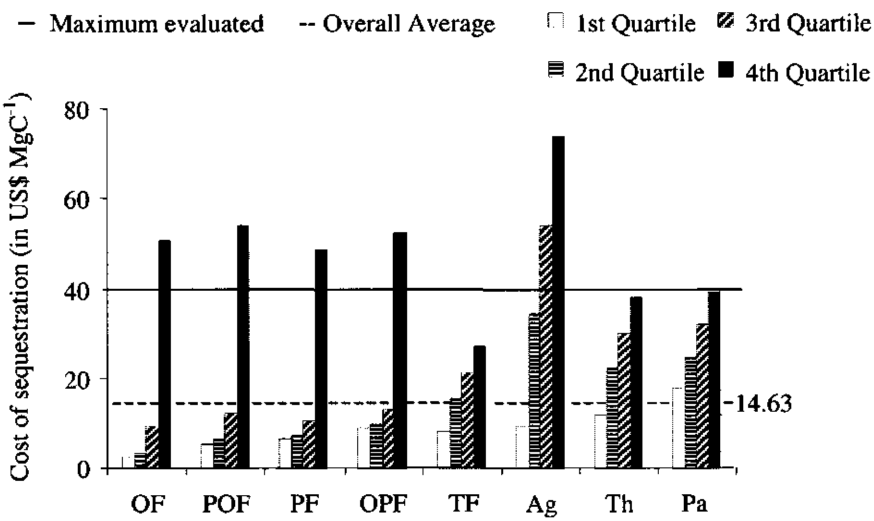


Figure 5.4. *Costs of Carbon sequestration in $US MgC^{-1} for the four quartiles of the Land use / Land Cover classes. OF = Oak and montane forest; POF = Pine-oak forest; PF = Pine forest; OPF = Open pine forest; TF = Tree fallow; Ag = Agriculture; Th = Thicket; Pa = Pasture.*

Carbon sequestration potential of the management options

The expected long-term average increase in C stock for the range of management options varied between 16 and 104 MgC ha^{-1} (Table 5.3).

If we take into account the future C decline over a 50 year period as calculated in the three baseline scenarios, the expected net sequestration potential varied from between 60 and 122 MgC ha^{-1} for the low baseline scenarios to between 60 and 174 MgC ha^{-1} for the high baseline scenarios. The net sequestration potential of the forest management options at the medium baseline scenario varied little, from 134 to 139 MgC ha^{-1}. Pasture and agriculture have the lowest net C sequestration potential of all scenarios, estimated at 60 MgC ha^{-1} (Table 5.3). A sensitivity analysis to test the effect of varying the input variables of the soil C dynamics with 25% around the default values produced a maximum of only 5% variation in the average long-term C storage estimation, when maintaining other variables constant.

Table 5.3. *Areas of Land Use / Land Cover types within the study area, the average total initial C-density, the estimated long-term increase in C stock and the net sequestration potential for each ha entering the C sequestration program under low, medium and high baseline scenarios.*

Land Use / Land Cover Types	Area (1996) (10^3 ha)	Total initial C-density (MgC ha^{-1})	Long-term average increase in C stock (MgC ha^{-1})	Net sequestration under low, medium and high baseline scenarios		
				Low (0.5%) (MgC ha^{-1})	Medium (1%) (MgC ha^{-1})	High (1.5%) (MgC ha^{-1})
Oak and Mountain Forest	14.9	503	16	83	134	174
Pine-oak Forest	190.7	341	61	103	135	159
Pine Forest	75.1	318	72	110	139	161
Open Pine Forest	36.2	236	104	122	135	146
Tree Fallow	115.6	315	77	104	124	140
Thicket	57.8	212	69	80	89	95
Pasture	59.2	153	60	60	60	60
Agriculture	75.1	153	60	60	60	60

Model outputs and sensitivity analysis

Under all baseline scenarios, the supply of sequestration was expected to be negligible with incentives below or equal to \$US5 MgC^{-1}, but rose sharply when increasing the incentives from \$US5 to 15 MgC^{-1}.

According to Fankhauser (1997), projects that pass a cost-benefit test within the range of \$US5 to 20 MgC-1 are worth undertaking. Within this cost range, forestry and agroforestry measures in our study area could mitigate from 1 to 42 * 106 MgC, with a maximum economic supply of carbon sequestration of around 55 * 106 MgC at \$US40 MgC-1.

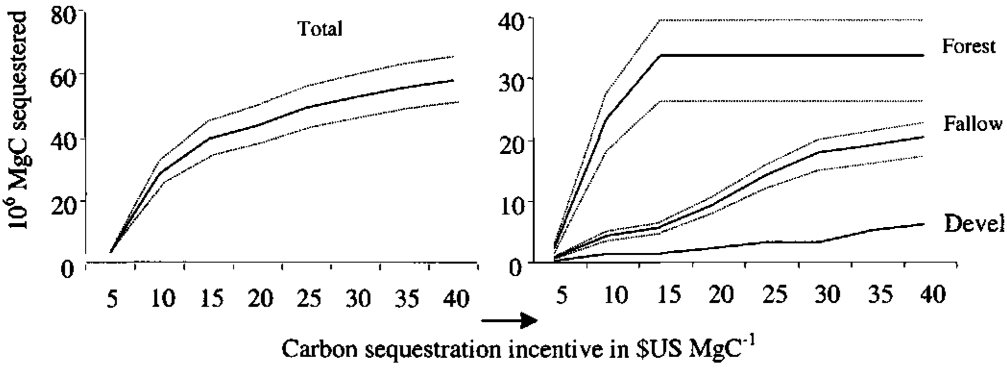


Figure 5.5. *Predicted carbon sequestration supply curves (in 10^6 MgC) for total, forest, fallow and development (Agriculture + Pasture) management options, based on low, medium and high baseline assumptions.*

The maximum supply through the management of communal forests was reached at \$US15 MgC^{-1}, the highest supply response from the improvement of fallow vegetation occurred between \$US15 and 30 MgC^{-1}. The response for agriculture and pasture was expected to rise slowly along the whole incentive gradient (Figure 5.5).

The effect of varying the discount rate was tested at 5, 10, 20 and 40%. A rate of 5% gave high present value to the medium- and long-term opportunity costs, while increasing the rate implied that future income and costs were less important. The supply response at 40% was dominated by the rate of incentive versus the opportunity, implementation and maintenance costs over the first five years. At low discount rates relatively low levels of sequestration supply were predicted for fallow and development options, where opportunity costs are relatively high (Figure 5.6 and Table 5.2). At a 20 to 40% discount level, the maximum supply of C sequestration for all systems could be obtained at an incentive level of below or around \$US20 to 25 MgC^{-1} (Figure 5.6).

Discussion

The model of the uptake of forestry incentives is based upon the assumption that farmers will react in an economically rational way to price signals, and there is evidence to suggest that this would hold for southern Mexico. Tipper (1993) and Javier Anaya (pers. comm. Unión de Crédito Pajal, 1994) both found that farmers in the northern highlands of Chiapas switched labor and capital inputs from coffee production to maize and bean production in response to a fall in coffee prices in the late 1980's. Experiences to date with the Scolel Té Pilot Project also indicate that farmers are generally eager to enter such program, even with lower incentive levels than predicted.

The issues associated with discounting for projects with long time horizons have been extensively discussed from the perspective of the public sector (e.g. Livingston and Tribe, 1995). One approach conventionally applied to calculate the social discount rate is the marginal rate of return on capital investment, since this represents the "opportunity cost" of capital. In the case of developing countries, where capital resources are judged to be scarce, this discount rate has frequently been set as high as 15% or more. However, in this study the purpose is not to make an objective decision about the value of a project in public accounting terms, but to estimate the probable reaction of a population of farmers to a package of financial and technical inputs that are distributed in time. From a simple test, we obtained generally high estimates of farmer's own rate of time preference, in the range of 20 to 40% after adjusting for inflation. As it is difficult to assess whether such rates would apply to decision making about long-term investments, we tested the model's sensitivity to a range of 5 to 40% discount rates.

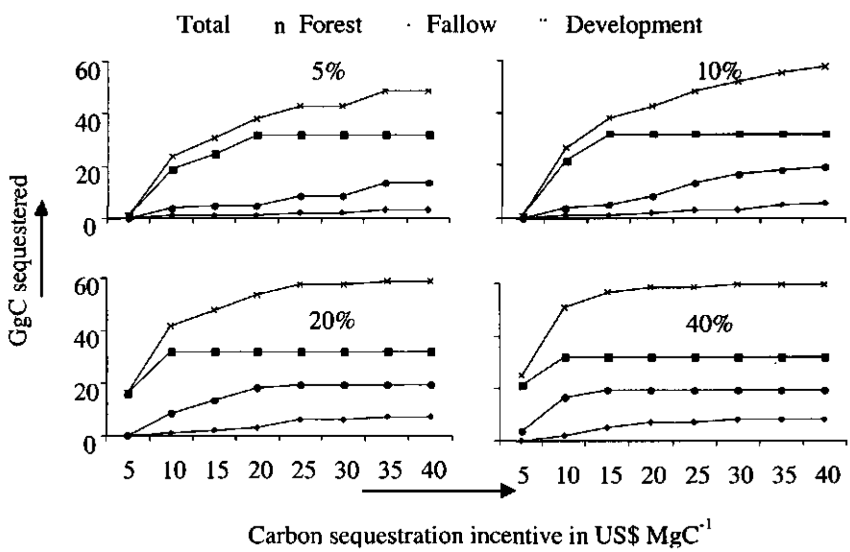


Figure 5.6. *Predicted carbon sequestration supply curves (in 10^6 MgC) for total, forest, fallow and development (Agriculture + Pasture) management options using 5%, 10% (default), 20%, and 40% discount rates.*

We have not taken into account the inevitable time lags that would be involved in the promotion and start-up pathways. In the first years of such a scheme, the start-up is likely to be tentative, with farmers assessing the benefits and costs by entering small areas of land on a "pilot" basis. The full scale of a program as predicted for a given rate of incentives might take 10 or more years to achieve (see also De Jong et al., 1998). Even if individual farmers are convinced that the scheme is worthwhile, there will be time lags associated with the mobilization of resources, the building of consensus for the management of communal areas, and capacity building. Farmers prefer to try out farm forestry on their own plots of land before committing themselves to organized activities at a communal level (De Jong et al., 1996). Other time lags will be caused by the need to build the administrative capacity in the organizations responsible for managing the scheme, and in particular the requirement for foresters trained in the social skills required to develop community forestry projects.

The choice of a baseline rate of biomass loss remains an important area of uncertainty when calculating the net sequestration effect of forestry activities. Differences in the order of half a percent in the assumed annual loss of carbon stock projected into the future can alter the expected long-term sequestration of a given area by tens of percentage points (Table 5.3, Figure 5.5). Future development policies may differ from previous initiatives, leading to quite different outcomes, in terms of land use. A key question is also whether high rates of forest conversion should be considered acceptable as a null case scenario, or to what extent governments are responsible for controlling deforestation through internal policies, without the use of resources specifically allocated to reduce greenhouse gas emissions. Setting a voluntary emission ceiling over a specific region and/or sector could address this problem (Tipper and De Jong, 1998). Other political concerns relate to the additionality effect of proposed forestry initiatives. Should programs and policies that conserve other service functions of forests (such as biodiversity, watershed integrity and amenity values) be considered part of the baseline or part of the project scenario?

In the current study, carbon dynamics from biomass changes over time were represented as long-term average changes in carbon stocks. However, carbon dynamics in terrestrial systems are a complex process with continuous, variable, and bi-directional fluxes. In addition, soil carbon dynamics remain important areas of uncertainty (Malhi et al., 1999). Our simulations estimated only a slight increase in average soil C stock from the alternative management options. However, there is evidence that differences in cultivation practices can have a significant effect on the soil C storage in certain cases (Ewel et al., 1981; Buyanovsky and Wagner, 1998).

In the case of sequestration projects, regulatory authorities might establish procedures to take into account the time differences between emissions and take-up by corresponding sinks. Such procedures could be based on either - application of a discount rate reflecting the estimated social cost of delaying emission reductions, or - calculations of the additional sequestration required to compensate for the radiative forcing produced by the delay. These procedures affect the merits of different forestry projects. Schemes that provide CO_2 abatement early on, for example maintaining existing forests, could be favored contrary to schemes that result in abatement in the future, such as planting new forests.

Conclusions

The experiences to date with the Scolel Té project indicate that since conventional agriculture is only marginally profitable, modest incentive payments can produce substantial shifts in land use, as was also observed in the UK (Crabtree, 1997). Given an appropriate mechanism for distributing sequestration rents to landowners, the amount of carbon sequestered would rise sharply from 1 to 38 * 10^6 MgC when incentive levels increase from $US5 to 15 MgC^{-1}, mainly due to natural forest management and fallow improvement. The management of natural forests and secondary vegetation will therefore be the most important element of any large-scale carbon sequestration program in Chiapas. Successful management requires mechanisms for adapting management plans to incorporate new information about the growth of the forest and changing social and economic circumstances. Since communities, rather than individuals or the public sector legally hold over 80% of the forests in Chiapas, any change in the management regime needs approval and consensus at the local level. Policies must therefore take into account the variation that exists both between and within communities.

Pilot projects, based on administrative models that can be increased in scale, such as the Scolel Té project (Scolel Té, 1997), could provide much of the information needed for more detailed assessments of costs and design requirements of large-scale schemes.

Acknowledgements

We thank Roger A. Sedjo, Roelof A.A. Oldeman, and four anonymous reviewers for their critical comments on earlier drafts of this paper. We also particularly wish to thank the following persons for their valuable contributions to the study: Susana Ochoa-Gaona, Lorena Soto-Pinto, Miguel-Angel Castillo-Santiago, Ignacio March-Mifsut and John Taylor. The

International Energy Agency's Greenhouse Gas R & D Programme funded acquisition and interpretation of satellite information in this document. Carbon density estimations of the land-use types were financed by the U.S. Agency for International Development / Mexico, under the Cooperative Agreement CR822200 between US-EPA and ECOSUR. The economic analysis in this publication is an output from a research project funded by the United Kingdom Department for International Development (DFID) for the benefit of developing countries. The views expressed are not necessarily those of the DFID. Project R7274 Forestry Research Programme.

References

Breedlove, D.E., 1981. Flora of Chiapas, Part 1: Introduction to the flora of Chiapas. The California Academy of Sciences, San Francisco, CA, USA.

Brown, S., Cannell, M.G.R., Kauppi, P., Heuveldop, J., Sathaye, J., Singh, N. and Weyers, S., 1995. Establishment and management of forests for mitigation of greenhouse gas emissions. IPCC 1995 Assessment.

Buyanovsky, G.A. and Wagner, G.H., 1998. Carbon cycling in cultivated land and its global significance. Global Change Biology 4(2): 131-141.

Cannell, M.G.R. and Milne, R., 1995. Carbon pools and sequestration in forest ecosystems in Britain. Forestry 68: 361-378.

Crabtree, J.R., 1997. Policy Instruments for Environmental Forestry: Carbon Retention in Farm Woodlands. In: W.N. Adger, D. Pettenella, and M. Whitby (Eds.), Climate-Change Mitigation and European Land-use Policies. CAB-International, Wallingford, UK, pp.187-197.

De Jong, B.H.J., Montoya-Gómez, G., Nelson, K., Soto-Pinto, L., Taylor, J., and Tipper, R., 1995. Community forest management and carbon sequestration, A feasibility study from Chiapas, Mexico, Interciencia 20: 409-416.

De Jong, B. H.J., Soto-Pinto, L., Montoya-Gómez, G., Nelson, K., Taylor, J., and Tipper, R., 1996. Forestry and Agroforestry Land-Use Systems for Carbon Mitigation: An Analysis from Chiapas, México. In: W.N. Adger, D. Pettenella, and M.C. Whitby (Editors), Climate Change Mitigation and European Land Use Policies. CAB International, pp. 269-284.

De Jong, B.H.J. and Ruíz-Díaz, M. de J., 1997. La investigación forestal y su contribución al manejo de los bosques. In: M.R. Parra-Vázquez and B.M. Díaz-Hernández (Editors), Los

Altos de Chiapas: Agricultura y crisis rural; tomo 1: los recursos naturales. ECOSUR, San Cristóbal de las Casas, México, pp. 149-165.

De Jong, B.H.J., Tipper, R., and Taylor, J., 1997. A Framework for Monitoring and Evaluation of Carbon Mitigation by Farm Forestry Projects: Example of a Demonstration Project in Chiapas, Mexico. Mitigation and Adaptation Strategies for Global Change 2(2-3): 231-246.

De Jong, B.H.J., Ochoa Gaona, S., Soto-Pinto, L., Castillo-Santiago, M.A., Montoya-Gómez, G., Tipper, R., and March-Mifsut. I., 1998. Modelling Forestry and Agroforestry Opportunities for Carbon Mitigation at a Landscape Level. In: G.J Nabuurs,. T. Nuutinen, H. Bartelink, and M. Korhonen (Editors). Forest Scenario modelling for ecosystem management at landscape level, EFI Proceedings No 19, pp. 221-237.

De Jong, B.H.J., Cairns, M.A., Haggerty, P.K., Ramírez-Marcial, N., Ochoa-Gaona, S., Mendoza-Vega. J., González-Espinosa, M., and March-Mifsut, I., 1999. Land-use change and carbon flux between 1970s and 1990s in Central Highlands of Chiapas, Mexico. Environmental Management 23(3): 373-385.

Dixon, R.K., Schroeder, P.E. and Winjum, J.K. (Editors), 1991. Assessment of Promising Forest Management Practices and Technologies for Enhancing the Conservation and Sequestration of Atmospheric Carbon and Their Costs at the Site Level. EPA/600/3-91/067, US-EPA, Washington D.C.

Dixon, R.K., Sathaye, J.A., Meyers, S.P., Masera, O.R., Makarov, A.A., Toure, S., Makundi, W., and Wiel, S.: 1996. Greenhouse Gas Mitigation Strategies: Preliminary Results from the U.S. Country Studies Program. Ambio 25: 26-32.

Ewel, J., Berish, C., Brown, B., Price, N., and Raich., 1981. Slash and burn impacts on a Costa Rican wet forest site. Ecology 62: 816-829.

Fankhauser, S., 1997. The Economic Costs of Climate Change and Implications for Land-use Change. In: W.N. Adger, D. Pettenella and M. Whitby (Editors), Climate-change Mitigation and European Land-use Policies. CAB-International, Wallingford, UK, pp. 59-70.

García, E., 1982. Modificaciones al sistema de clasificación climática de Köppen. México D.F., México.

González-Espinosa, M., Ochoa-Gaona, S., Ramírez-Marcial, N. and Quintana-Ascencio, P.F., 1995. Current Land-Use Trends and Conservation of Old-Growth Forest Habitats in the Highlands of Chiapas, Mexico. In: M.H. Wilson and S.A. Sader (Editors), Conservation of Neotropical Migratory Birds in Mexico. Los Tuxtlas, Veracruz, Nov 5-7, 1993, pp 190-198.

Hellier, A., 1996. The potential of local knowledge for monitoring changes in vegetation and biodiversity: a case study from Chiapas, Mexico. MSc Dissertation, University of Edinburgh.

Holdridge, L.R., 1967. Life zone ecology. Tropical Science Center, San José, Costa Rica.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica), 1984. Carta Uso del Suelo y Vegetación 1:250 000. Tuxtla Gutierrez E15-11. INEGI, Mexico, D.F.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica), 1987. Carta Uso del Suelo y Vegetación 1:250 000. Villahermosa E15-8. INEGI, Mexico, D.F.

INEGI (Instituto Nacional de Estadistica Geografia y Informatica), 1988. Carta Uso del Suelo y Vegetación 1:250 000. Las Margaritas E15-12. INEGI, Mexico, D.F.

INEGI, 1993. Anuario Estadístico del Estado de Chiapas. Instituto Nacional de Estadística Geográfica e Informática, México D.F., México.

Konstant, T., 1997. The use of secondary vegetation in the highlands of Chiapas, Mexico. MSc Dissertation, University of Edinburgh.

Livingston, I. and Tribe, M., 1995. Projects with long time horizons: their economic appraisal and the discount rate. Project Appraisal 10(2): 66-76.

MacDicken, K.G., 1997. Project specific monitoring and verification: State of the art and challenges. Mitigation and Adaptation Strategies for Global Change 2(2-3): 191-202.

Malhi, Y., Baldocchi, D.D., and Jarvis, P.G., 1999. The carbon balance of Tropical, Temperate and Boreal Forests. Plant Cell and Environment, in press.

Masera, O.R., Bellon, M.R., and Segura, G., 1995. Forest Management Options for Sequestering Carbon in Mexico. Biomass and Bioenergy 8: 357-367.

Miranda, F. and Hernández-Xolocotzi, E., 1963. Los tipos de vegetación de México y su clasificación. Boletin de la Sociedad Botánica de México 28: 29-179.

Moulton, R.J. and Richards, K.R., 1990. Costs of sequestering carbon through tree planting and forest management in the United States. USDA-Forest Service. Gen. Tech. Rep. WO-58.

Nabuurs, G.J. and Mohren, G.M.J., 1993. Carbon fixation through forestation activities: A study of the carbon sequestering potential of selected forest types. Inst. for Forestry and Nature Research (IBN-DLO), Wageningen, The Netherlands, IBN Research Report 93/4.

Nabuurs, G.J. and Mohren, G.M.J., 1995. Modelling analysis of potential carbon sequestration in selected forest types. Canadian Journal of Forest Research 25: 1157-1172.

Ochoa-Gaona, S. and González-Espinosa, M. 2000. Land use patterns and deforestation in the highlands of Chiapas, Mexico. Applied Geography 20: 17-42.

Parra Vázquez, M. P.(Coord.), 1989. El subdesarrollo agrícola en los Altos de Chiapas. Colección cuadernos Universitarios, Serie Agronomía No. 18. Universidad Autónoma Chapingo / Centro de Investigaciones Ecológicas del Sureste.

Sathaye, J. and Ravindranath, N.H., 1997. Policies, Measures and the Monitoring Needs of Forest Sector Carbon Mitigation. Mitigation and Adaptation Strategies for Global Change 2(2-3): 101-115.

Scolel Té, 1997. Scolel Té: International pilot project for carbon sequestration and community forestry in Chiapas, Mexico. http://www.ed.ac.uk/~ebfr11.

Swisher, J.N., 1992. Cost and performance of CO2 storage in forestry projects. Biomass and Bioenergy 1: 317-328.

Tipper, R., 1993. Technological Change in Contemporary Highland Mayan Agriculture. PhD Thesis, University of Stirling, Scotland, UK.

Tipper, R., De Jong, B.H.J., Ochoa-Gaona, S., Soto-Pinto, M.L., Castillo-Santiago, M.A., Montoya-Gómez, G. and March-Mifsut, I., 1998. Assessment of the cost of large scale forestry for CO2 sequestration: Evidence from Chiapas, Mexico, report PH2/15, IEA Greenhouse Gas R & D Programme.

6. UNCERTAINTIES IN ESTIMATING THE POTENTIAL FOR CARBON MITIGATION OF FOREST MANAGEMENT[4]

Introduction

The Kyoto Protocol, signed by the Parties to the United Nations Framework Convention on Climate Change in December 1997 in Kyoto, Japan, allows the use of a number of market mechanisms to enable countries to achieve greenhouse gas (GHG) emission reduction targets cost-effectively. These new mechanisms include international emission trading and joint implementation among countries with binding emission ceilings, the so called annex-1 countries, and the clean development mechanism, which aims to enhance co-operation among annex-1 and non-annex-1 countries to achieve sustainable development and at the same time reduce GHG emissions (UNFCCC, 1997). The clean development mechanism is intended to be a vehicle to harness funding for clean development projects in tropical countries in exchange for certifiable emission reduction credits.

Since the Kyoto protocol has been established, there has been heightened interest in the potential for using forests as a means of mitigating GHG emissions. The potential for enhancing carbon (C) storage or avoiding emissions by forestry and agroforestry may be as much as 60 to 90 x 10^6 Gigagrams of C (GgC = 10^9 gC; Dixon et al., 1991, 1993; Winjum et al., 1992; Trexler and Haugen, 1994; Brown et al., 1995). Forestry and land-use mitigation measures can serve other environmental, economic, and social interests simultaneously, and may offer some of the most cost-effective ways to climate change mitigation. GHG-offset projects in the land-use and forestry sector are particularly attractive if these can be tied to local social and economic goals (Trexler, 1993).

Using forests as a means of mitigating climate change can be achieved by maintaining or increasing existing stocks of C in forests that are currently threatened, by creating new stocks in growing trees, and by substitution of energy-demanding materials by renewable natural resources (Schlamadinger and Marland, 1996).

In Mexico, the forestry sector is considered a key element in the national GHG-mitigation plan (CICC, 1999). Currently, land use/land cover (LU/LC) change accounts for an estimated 35% of total national CO_2 emissions. However, if effective policies to reduce deforestation

---

[4] *Re-submitted as: De Jong, B.H.J. 2000. Uncertainties in estimating the potential for carbon mitigation of forest management. Forest Ecology and Management, May 2000.*

and increase afforestation are implemented, Mexican forests could become large carbon sinks. This path also offers two other important advantages. First, carbon savings in the forest sector will help buy time to develop mitigation alternatives in the energy sector, because growing forests accumulate carbon fast and in large amounts. Second, following the path in the policy scenario suggested by Masera et al (1997), could result in tangible additional benefits for the rural population. They consider that no intrinsic obstacles hinder the sustainable management of forest resources in the country. Population growth is not the leading factor in the deforestation process and food demand can be largely accommodated within the existing areas open to cultivation through either a combination of a better crop mix and technological improvements (Masera et al., 1997) or smart ecological land use (Van der Wal, 1998).

A key question still to be resolved is how to measure the reduction of emissions from a given project. There are serious complications in estimating C-dynamics in forest ecosystems because fluxes among vegetation, soil, and atmosphere are complex, bi-directional and persistent. Hence, a number of methodological questions need to be addressed before forestry C-

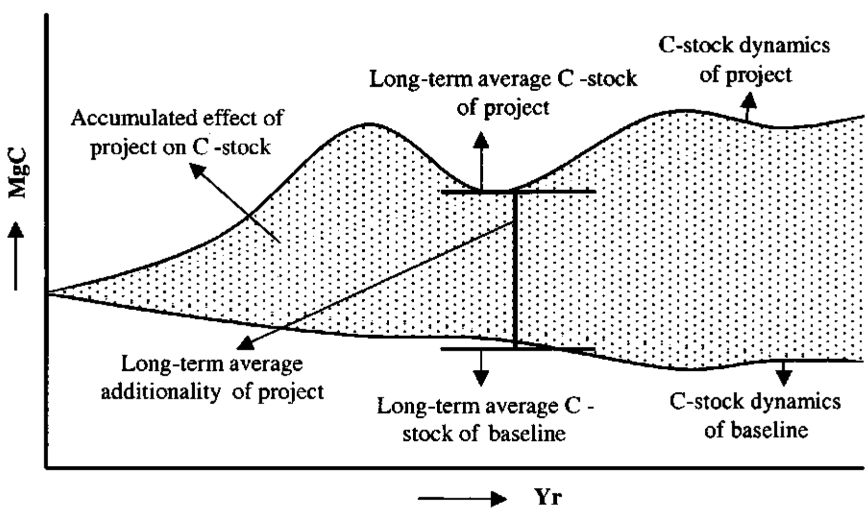


Figure 6.1. *Hypothetical effect of a GHG-mitigation project, compared to a "business-as-usual" scenario.*

offset trading can reliably provide verifiable emission reductions (Tipper and De Jong, 1998). The emission reduction of any given project rests upon the argument that the environmental performance of the project, once implemented, exceeds the performance of the expected scenario without that project, i.e. in a "business as usual" or baseline situation (Figure 6.1).

The baseline scenario describes the past, present and expected future set of GHG fluxes in the case the project is not implemented (Trines, 1998). This means that not only the project outcome has to be estimated in terms of greenhouse gas fluxes, but also the expected fluxes of the most probable future. The current convention for baseline setting is to assess the most likely outcome at a given site, based on historical precedent or "business as usual" scenarios.

It is the primary purpose of this study to assess the sources and levels of uncertainties that may occur in the quantification of the CO_2-mitigation potential of forestry projects. Data collected in the field are used to assess the levels of uncertainties in classification methods, C-stock quantification in LU/LC classes and major biomass pools, and baseline assumptions, and how to take these uncertainties into account. The proposed forest management project has been developed as part of the Scolel Té project and includes a set of actions to increase forest productivity in existing forest, the use of selected genotypes to restore degraded forests, and to increase soil carbon by leaving dead wood, litter, and slash from harvests.

Methodology

The data presented in this study were collected and analyzed at different scales and intensity. Carbon densities were measured at two levels and scales: all major C-pools were measured in a set of LU/LC classes at a regional scale, whereas the tree-C densities were measured in a set of LU/LC classes in a single community forest. Land use data were also analyzed at these two geographic scales (Figure 6.2): (1) sub-regional (part of the Highlands of Chiapas) and (2) community (Juznajab La Laguna).

Study area

The Highlands of Chiapas (607,000 ha, Figure 6.2), located at 1500 to 2900 m elevation in southeast Mexico, contain various forest formations, including pine, pine-oak, oak, and evergreen cloud forests (Breedlove 1981; González-Espinosa et al., 1995b). Regionally, a tropical lower montane, sub-humid climate predominates, with summer rains and winter droughts. Mean annual rainfall varies between 1100 and 2000 mm. The soils are dark brown, clayey loam, derived from calcareous rocks (Parra-Vázquez et al., 1989), primarily rendzinas, luvisols, acrisols, feozems, regosols y lithosols (INEGI, 1985).

The community Juznajab La Laguna (Juznajab, 16°22' N, 92°13' W) is located near Comitán de Domínguez (Comitán). Its territory of 4,004 ha lies within the municipalities Comitán and Las Margaritas. Mean annual rainfall reported for Comitán varies between 650 and 1550mm, with a long-term average of 1030 mm.

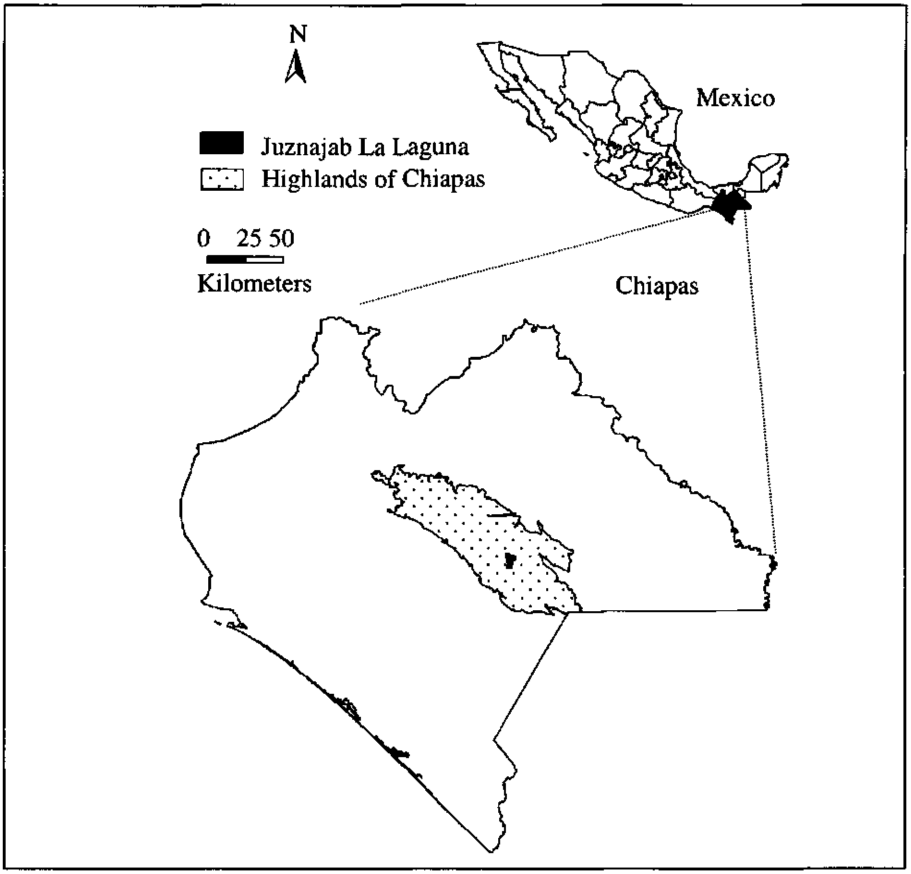


Figure 6.2. *Location of the Highlands of Chiapas and Juznajab La Laguna.*

Juznajab was founded in 1880 and currently has around 700 inhabitants (Montoya et al., 1995). The following LU/LC classes occur within the community: well-developed forests, secondary forests, tree fallow, shrubland and open areas. The latter include grassland, cropland, bare soil, and settlements. The community territory also comprises fresh-water lagoons, which provide drinking water to various surrounding communities. In 1996, 66 farmers started reforesting part of their private pastureland within the context of the Scolel Té project (Scolel Té, 1998). In 1997 a forest inventory was carried out in about 3000 ha, as part of the requirements to obtain a forest management permit.

Until the late 1980s, the community sold the standing wood to forest concessionaires, who in turn cut all harvestable trees. Farmers of the community temporarily converted a large part of the harvested area to slash-and-burn agriculture. Once the agricultural production dropped to levels assumed to be below sustained, acceptable productivity, the area was left to recover to secondary forest (Van der Wal, 1998). The current secondary forests are the result of the natural regeneration of areas that were exploited and temporarily converted to slash-and-burn agriculture during the 1970s. The well-developed, closed forests that occur have a similar origin, except that these have been exploited earlier. Shrublands originate from (temporarily) abandoned pasture- and cropland.

From the late 1980s onward, the community decided not to permit any commercial wood exploitation. Only the collection of fuelwood and felling of construction wood for local consumption was allowed.

The community is set in a landscape where recent changes in land use have been both extensive and intensive (De Jong et al., 1999). Estimated deforestation rates in the Chiapas Highlands ranged from 3.2% (from 1974 to 1984) to 3.6% (1984 to 1990) for closed forests, and from 1.6% (1974 to 1984) to 2.1% (1984-1990) for open and closed forests combined (Ochoa-Gaona and González-Espinosa, 2000). The structure and composition of the remaining forests have been altered due to (i) selective harvesting of pine trees for local timber production, and of oak trees for fuelwood and charcoal, and (ii) extensive sheep and cattle grazing (De Jong and Montoya-Gómez 1994; González-Espinosa et al., 1995a, b). Currently, only a few small patches of old-growth forest remain. Driving forces behind the deforestation process have been government incentives for agricultural development, shifts from subsistence to market-oriented production systems, infrastructure development, and insecure land and tree tenure systems (De Jong and Montoya, 1994).

Calculation of Greenhouse Gas-mitigation benefits

The mitigation benefits of an activity are equivalent to the difference between GHG fluxes as a result of the activity and the amount of GHG fluxes that would occur without any project (Figure 6.1). In this paper, the GHG-offset potential of the project is calculated with the emission-equivalent C storage approach (MgC-eq, expressed in MgC, Chomitz, 1998; Tipper and De Jong, 1998). This approach adopts a two-dimensional measurement unit that reflects storage and time, i.e. MgC·year. The approach takes into account that an instantaneous emission of CO_2 will be absorbed again slowly by the terrestrial or oceanic biosphere. In other words, the total radiative forcing produced by an emission of 1 MgC is temporal and can be calculated by summing the CO_2 concentrations remaining in the atmosphere in the years after the emission. Calculation of this sum provides an estimate of the cumulative carbon storage that would be required to offset an emission of 1 MgC at the present time. It is estimated that at least 80% of a CO_2 emission will be taken up again within 50 to 200 years (Cook, 1995). This means that between 30 and 69 Mg·years C need to be stored to mitigate 1 MgC emitted, if we take 100 years as the time horizon, i.e. the "normal" timespan proposed by the Kyoto protocol (Houghton et al., 1990, 1997; Cook, 1995). This emission equivalent approach is attractive as carbon storage can be credited according to the time frame over which storage takes place, reducing the need for long-term guarantees. It avoids problems of how to compare projects with variable duration and how to account for temporal emissions during harvesting in for example plantation forestry. It also allows project crediting periodically determining the amount of carbon stored each period, multiplied by the equivalence factor E_f (1/Conversion factor). In this paper, a conversion factor of 50 is used to estimate the cumulative C storage that would be required to offset an emission of 1 MgC, as proposed by Chomitz (1998). The amount of MgC-equivalent emission offset (MgC-eq_i) obtained with the proposed project is calculated on a yearly basis with the following formula:

$$\text{MgC-eq}_i = (\Sigma_i (C_{a,i} - C_{b,i})) / 50 \quad (1)$$

where $C_{a,i}$ is the C accumulated in the alternative system, $C_{b,i}$ is the C accumulated in the baseline, both in year i and both compared to the C-stock in year zero, and 50 is the assumed conversion factor. Thus, each year that the project either accumulates or releases C relative to the baseline C-stock of that year, this difference is added up and the sum over the years is divided by the conversion factor. With this approach temporary emissions through harvesting or ageing of the forest do not affect negatively the total GHG-impact of the project, as long as the remaining C-stock after the emission is higher than or equal to the baseline C-stock of the year that the temporal emission occurs.

Establishment of a baseline

The baseline must be set at a level that ensures that the emission mitigation activity is additional to what would have occurred if the project had not been implemented. Baseline definitions in the land-use sector minimally need to deal with delimitation of the project domain and an estimation of LU/LC dynamics and associated GHG fluxes in case the project would not be implemented. Information about historical and future land-use policies and an estimation of the effect of these policies on LU/LC change dynamics is desirable (Puhl, 1998), but not always available. The following baseline assumptions were applied:

a) Land-use change dynamics in Juznajab La Laguna and surroundings will continue at a similar rate as in the past. The C-stock is expected to diminish at the same annual rate (in % of stock) as in the last 22 years due to the land-use change until 2010. After 2010 this process is assumed to slow down until 2020, due to improved enforcement of forestry and land-use regulations. After 2020 the C-emissions caused by LU/LC change are assumed to be counterbalanced by C-sequestration of abandoned agricultural land, pasture, or secondary vegetation.

b) Juznajab La Laguna will contract a concessionaire who will harvest all timber trees in the well-developed forests, as has been done in the past. Harvested areas will be temporarily converted to slash-and-burn agriculture, leaving only some trees standing. Historically at least 90% of the original standing volume was cut due to these two combined activities.

Project scenario

The project management option, the 3000 ha forest will be managed by the community, provided they receive financial assistance within the context of the Scolel Té project.

Members of the community will carry out the forest management activities, for which they will receive training. The basic principles of the management techniques are to convert the uneven-aged low-productive forests to a mosaic of small eco-units (sensu Oldeman, 1990) of more or less even-aged high-productive forest by means of the following silvicultural measures:

- Harvesting of well-developed forests in small groups, favoring natural regeneration of pine trees in the openings.
- Selective thinning of each group every 10-years.
- Fire and pest control.
- Saving well-formed parent trees during harvesting to allow for controlled regeneration.
- Enrichment planting in degraded and open forests.

The 3000 ha of forests contain four pine species and various oak and other broad-leaved species. These forests are sources of locally important products, such as medicinal and edible plants, wood and fibers for construction, fuel, furniture, etcetera. About 60 woody species were recorded in the inventory. In order to guarantee a continued supply of these locally important goods, the community will implement a reduced level of selective thinning, without burning of harvest residues. These measures will allow the build-up of organic matter in the forest soil as a GHG sink, and the regeneration of locally important trees, shrubs and annual plants.

The following GHG-mitigation impacts are applicable to the project scenario:

- Avoided emissions from conversion to other land use or from forest degradation
- Reduced emissions from burning
- Increased C-stock in the managed forest and soil
- C fixed by long-lived forest products
- Mitigation of peak releases due to less intensive wood harvesting

To avoid possible leakage of increasing deforestation outside the community, the remaining 1000 ha of non-forest land, including cropland and degraded pastures, will be subjected to improved agricultural techniques and silvopastoral systems. These activities are already implemented by the 66 farmers, participating in the Scolel Té project. It is assumed that these land-use activities applied to all non-forest land will guarantee a continued supply of non-forest products that to cover future demands of the community.

Land use / Land cover data

Land use / land cover data were obtained from interpreted Landsat MSS images from 1974, 1984 and 1990 and a Landsat TM image from 1996. For the 1996 image classification, field data were collected, whereas the 1974, 1984 and 1990 images were classified with the aid of aerial photographs of 1974 (scale 1:50,000), 1987 and 1991 (scale 1:75,000). Ochoa-Gaona and González-Espinosa (2000) could distinguish the following LU/LC classes in the four sets: Closed broad-leaved forest, closed pine forest, closed pine-oak forest, open pine forest, disturbed pine-oak forest, tree fallow, shrub fallow (thicket), open grassland, agricultural land, the latter including bare soil and settlements.

A second independent LU/LC classification was carried out in the Juznajab community, for which 48 georeferenced forest inventory plots and around 25 georeferenced non-forest points were used to interpret the part of the 1996 Landsat TM image corresponding to the community (Escandón-Calderón et al., 1999). The following LU/LC classes could be distinguished at this level: open areas or undefined (including pasture and agricultural land, shade and settlements), oak and broad-leaved shrub vegetation, pine, pine-oak and oak-pine secondary forest, well-developed pine and pine-oak forest.

A comparison of the two LU/LC classifications was carried out for a total of 3,536 ha, excluding non-comparable classes and shade. The following aggregate LU/LC classes were established to compare the two classification methods: (1) open areas and shrubland, (2) secondary or degraded forests, and (3) well-developed, closed pine and pine-oak forests.

C-stock and growth measurements

Carbon densities associated with the LU/LC classes were measured in 39 plots distributed in the Highlands of Chiapas, separating the following pools: soil organic matter, small roots, herbaceous plants, shrubs, trees, woody debris, stumps, and litter (Adapted from De Jong et al., 1999). Tree C-densities were measured in a total of 102 sites in Juznajab La Laguna. In each site, diameter at breast height (dbh, 1.3 m), total and stem height were measured and species determined for all trees >10 cm dbh in three circular plots of 1000 m^2 each (0.3 ha in each site). Allometric equations of biomass in relation to dbh and height, developed for the Highlands of Chiapas, were used to calculate C densities of the tree component in each site (Ayala-López, 1999). At each site, diameter increment data were collected from five to 15 pine trees (depending on species and dbh size-classes present), with a total of 900 samples. These increment data were used to develop volume increment equations in relation to diameter and age, which in turn served as the basis to estimate volume increment in relation to standing stem volume. To compare the effect of the classification methods on C-density estimation, tree C-density data were multiplied by the corresponding surface area of each LU/LC class for each classification method.

Historical evidence of C-storage depletion

Total C-densities were assigned to each LU/LC class and multiplied with the surface area of each class for each evaluation year. It is assumed that the average C-stock within a LU/LC class does not change over time and that the stock change due to LU/LC change occurred during the period between each pair of successive years of comparison. The historical rate of annual C-storage depletion was calculated for Juznajab La Laguna (4 * 10^3 ha) and the Highlands of Chiapas (306 * 10^3 ha) with the following formula:

$$RR = 1 - (1 - (C_i - C_f) / C_i)^{1/y} \qquad (2)$$

where RR is annual C-reduction rate; C_i is total C in initial year; C_f is total C in final year; y is number of years.

C-flux modeling of reference and project scenarios

To understand the role of forestry and agroforestry systems in the carbon cycle it is necessary to quantify both the net annual carbon fluxes and the total carbon content of the systems,

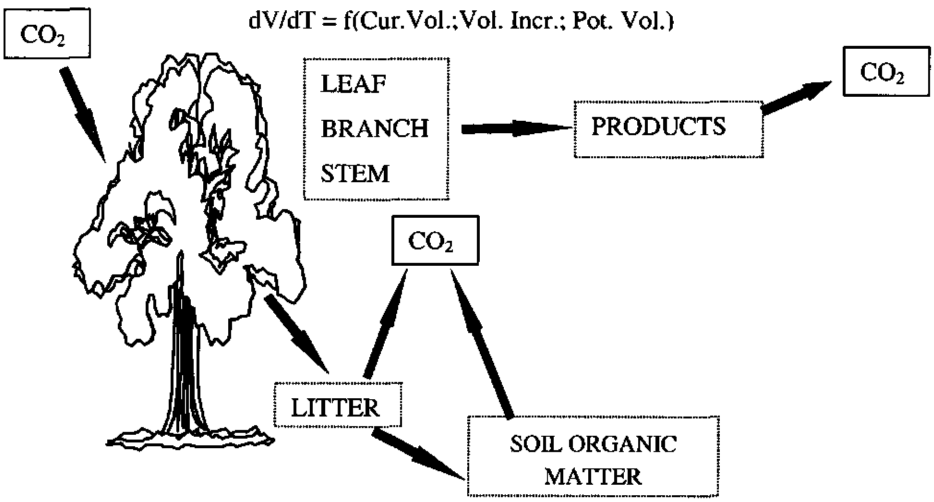


Figure 6.3. *Simplified representation of the model outline used to simulate the biomass accumulation in forest stands, with emphasis on the most important biomass pools and flows that affect these pools.*

including the carbon fluxes and stocks in the products extracted from the system (Nabuurs and Mohren, 1993). Burning and oxidation of the biomass and extracted products and mineralization of soil organic matter release large amounts of C to the atmosphere. When the living system grows, C is removed from the atmosphere and stored in the biomass and organic soil horizons.

A dynamic model, based on mathematical relations among the various biomass pools, was developed that predicts fluxes through a natural forest stand. In this model, management activities can be programmed at any time and intensity, and their impact on the various biomass pools estimated (Figure 6.3; Arp and McGrath, 1987).

In the Highlands of Chiapas, total ecosystem biomass was found to be strongly correlated with aboveground standing tree biomass (Figure 6.4). Therefore, tree volume increment data were used as the driving function to simulate biomass increase for each stand.

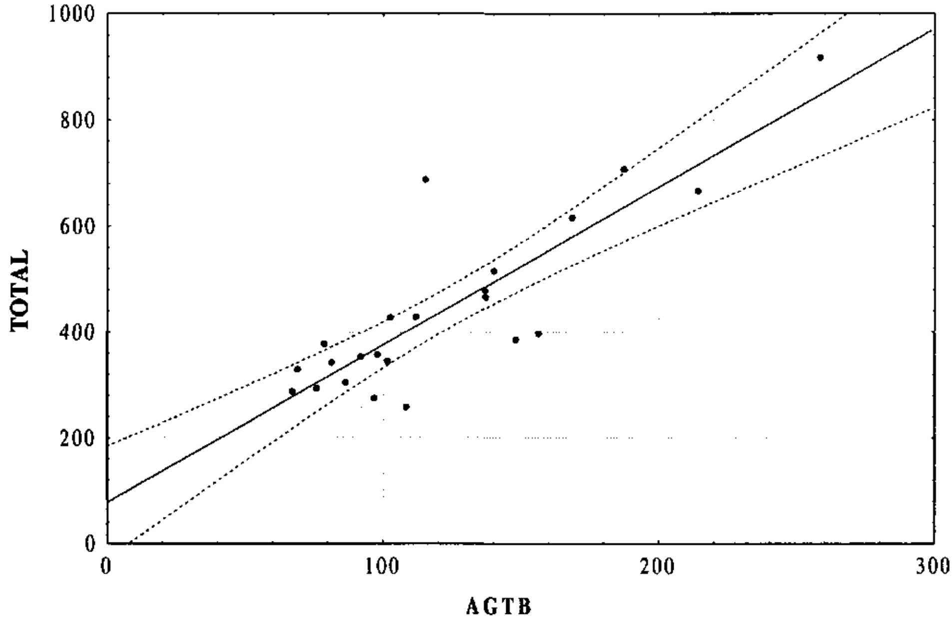


Figure 6.4. *Relation between aboveground tree biomass (AGTB) and total biomass (excluding tree biomass) for the Highlands of Chiapas (n = 23; Total = 77.7 + 1.98·AGTB; r = 0.74; p < 0.001) (Adapted from De Jong et al., 1999).*

In the model, a logistic type of growth equation is driving the tree volume increase for each year (Cannell and Milne, 1995). The parameters that determine the growth in any year are standing tree volume in that year, potential maximum standing tree volume, and expected maximum volume increment. These parameters were derived from data collected in the 102 sites in Juznajab La Laguna.

At the stand level, the input variables include an initial amount of biomass in each major pool. The carbon content, measured in the 39 plots of the Highlands of Chiapas, was used as the starting point in each simulation for each LU/LC class (See for more details De Jong et al., 1998). The change parameters were estimated based on the mathematical relations between the various C-pools of the 39 plots, or taken from the literature (e.g. Arp and McGrath, 1987; Nabuurs and Mohren, 1995; CASFOR, 1999). Tree growth, above-ground assimilate allocation, and mortality were derived from field data. Published data on leaf, branch and root turnover rates, litter humification, litter, humus and woody debris decomposition, and assimilate allocation to roots were used in the C-flux simulations (Nabuurs and Mohren, 1995). Harvesting of trees was modeled for each LU/LC class and for both reference and project forest management options (Table 6.1). The initial state variables and natural change parameters for both management options were assumed to remain the same during the flux simulation. As such, only the variation in expected C fluxes due to differences in forest management activities between the two options were considered.

Table 6.1. *Harvesting schedule used to model the expected C-fluxes in reference and project management options for each LU/LC class. Starting dates of the harvesting activities depend on initial stand condition. Pine stems are used for timber or construction wood, whereas oak stems are exploited for fuelwood.*

Activity	Reference option		Project option	
	Pine	Oak	Pine	Oak
Harvesting (Including thinning)	90% of standing volume each 40 years	10% of standing volume each year	2% of standing volume each year	4% of standing volume each two years

Estimation of the Carbon mitigation potential

A spreadsheet type model was constructed to compile the surface areas of each LU/LC class, the initial C-pool densities in each class, and the expected C-flux outputs of the reference and project management options for each LU/LC class. The expected future C-reduction rate and the conversion factor can be varied in the spreadsheet. It is assumed that the 3000 ha of forest will be either managed according to the reference or according to the project scenario. Neither the inevitable time lag that would be involved in the application of the project scenario, nor the possibility of applying parts of both scenarios, has been taken into account (See also De Jong et al., 1998).

Uncertainty analysis

The area-weighted sum of variance in total and tree C-stock for each LU/LC class was used to estimate the 95% confidence interval of the C-reduction rates, expressed in percentage of initial C-stock.

An uncertainty analysis was carried out to determine the robustness of the model and the relative importance of variation in parameter values. Populations of 100 randomly selected values of the most important input parameters were used for this analysis. To test the relative importance of the variance in parameter values, each population was assumed to have a normal distribution with a variation of about 25% around the default value, which was similar to the variation of those parameters that were directly measured (See also Van de Voet, 1993; Van der Voet and Mohren, 1994). First the model was run 100 times with a random distribution of the values of all parameters. A 95% confidence interval of the simulation results was calculated and these results were used in the spreadsheet model. Separately, the model was run in two additional series with: (i) the value of one particular parameter variable and all other parameters fixed to calculate the relative specific variance (R.S.V.) of the tested parameter and (ii) the value of one parameter fixed and all other parameters variable to calculate the relative reduction in variance (R.R.V.) of this parameter (Van der Voet, 1993). The relative specific variance represents the minimum residual variance of the estimate if the value of only the specific parameter remains unknown, whereas the relative reduction in variance gives the maximum reduction in variance that can be achieved by knowing the exact value of this particular parameter while all other parameters remain uncertain (Jansen et al., 1994). These procedures are indicative of the level of uncertainty in the flux estimation results due to observed or estimated variations in parameter values and at the same time identifies, which parameters are contributing most to this uncertainty (Jansen et al., 1994; Kleijnen, 1994; Bartelink, 1998).

The overall uncertainty of the C-mitigation estimation was calculated with the following formula (IPCC, 1996):

$$TU = \sqrt{} (\Sigma_i (U_i)^2, \quad (3)$$

in which TU is the sum of all uncertainty values, and U_i the uncertainty of independent variable i (Variance in C-reduction rate, in C-accumulation, in C-stock, etc.).

Results

Carbon stocks and allometric relations between carbon pool quantities in forests

In the highlands, the total C-densities in the LU/LC classes varied between 153.3 ± 20.5 MgC ha^{-1} in open areas and 463.4 ± 156.3 MgC ha^{-1} for oak forest, whereas tree C-densities varied between 0 to 14.8 in open areas and 152.2 ± 44.7 MgC ha^{-1} for oak forest (Table 6.2).

Table 6.2. *Estimated total and tree carbon (C) densities (95% confidence interval, CI) for the different land-use/land-cover classes, based on data collected in the Highlands of Chiapas (Adapted from De Jong et al., 1999).*

LU/LC class	N	Total-C (95% CI)	Tree-C[1] (95% CI)
Oak forest	7	463.4 (156.3)	152.2 (44.7)
Pine Oak Forest	11	340.7 (61.6)	113.5 (13.8)
Pine Forest	5	318.3 (109.1)	103.4 (48.7)
Tree Fallow	6	314.9 (50.9)	56.5 (16.2)
Disturbed Pine-Oak Forest[2]	11 + 4	276.3 (55.6)	88.4 (23.5)
Open Pine Forest[3]	5 + 6	235.8 (70.3)	50.2 (36.6)
Shrubland	4	211.8 (55.1)	19.3 (9.8)
Open area	6	153.3 (20.5)	5.9 (8.9)

[1] *All trees with diameter at breast height (1.30m) greater than or equal to 5 cm.*

[2] *Average between Pine-Oak Forest and Shrubland, with CI based on variance of plots of these classes.*

[3] *Average between Pine Forest and open area, with CI based on variance of plots of these classes.*

Linear relations between the various C-pools and tree-C density were significant, but with a low- or non-significant intercept of the linear regression (p between 0.05 and 0.65). The ratio estimates of non-tree biomass densities in relation to aboveground tree biomass were significant ($p<0.01$) for total non-tree biomass, aboveground living non-tree biomass, soil organic matter, necromass, fallen wood, litter, and small roots. Confidence intervals of the ratio estimates were less than 10% of the estimated parameter in all cases, except for fallen wood (Table 6.3). The relation between the biomass allocation of tree crowns and stems was almost linear for both pine and oak trees (Figure 6.5).

Table 6.3. *Mathematical relations between aboveground tree biomass and other major biomass pools in the forests of the Highlands of Chiapas (n = 23). Linear regression: y = a + b·x; Ratio estimate: y =a·x. x = aboveground tree biomass; y = other major biomass pool; r is goodness-of-fit statistic. Between parenthesis the p-level of the parameter.*

Biomass pool	**Linear regression**			**Ratio estimate**	
	a	**b**	**r**	**a**	**r**
Total biomass (excluding aboveground tree biomass)	77.7 ± 51.3 (0.14)	1.98 ± 0.4 (<0.001)	0.74	2.53 ± 0.14 (<0.001)	0.97
Total aboveground living biomass (excluding aboveground tree biomass)	-1.6 ± 2.2 (0.48)	0.03 ± 0.02 (<0.1)	0.36	0.02 ± 0.005 (<0.01)	0.55
Soil organic matter	19.5 ± 41.8 (0.65)	1.42 ± 0.32 (<0.001)	0.70	1.56 ± 0.11 (<0.001)	0.95
Necromass (Fallen wood, litter and tree stumps)	0.7 ± 6.0 (0.9)	0.18 ± 0.05 (<0.001)	0.65	0.18 ± 0.02 (<0.001)	0.92
Fallen wood	-6.1 ± 5.7 (0.33)	0.13 ± 0.04 (<0.01)	0.55	0.09 ± 0.01 (<0.001)	0.77
Litter	6.0 ± 2.6 (<0.05)	0.04 ± 0.02 (<0.05)	0.43	0.09 ± 0.008 (<0.001)	0.92
Small roots	0.9 ± 0.5 (<0.1)	0.009 ± 0.003 (<0.05)	0.47	0.02 ± 0.001 (<0.001)	0.92

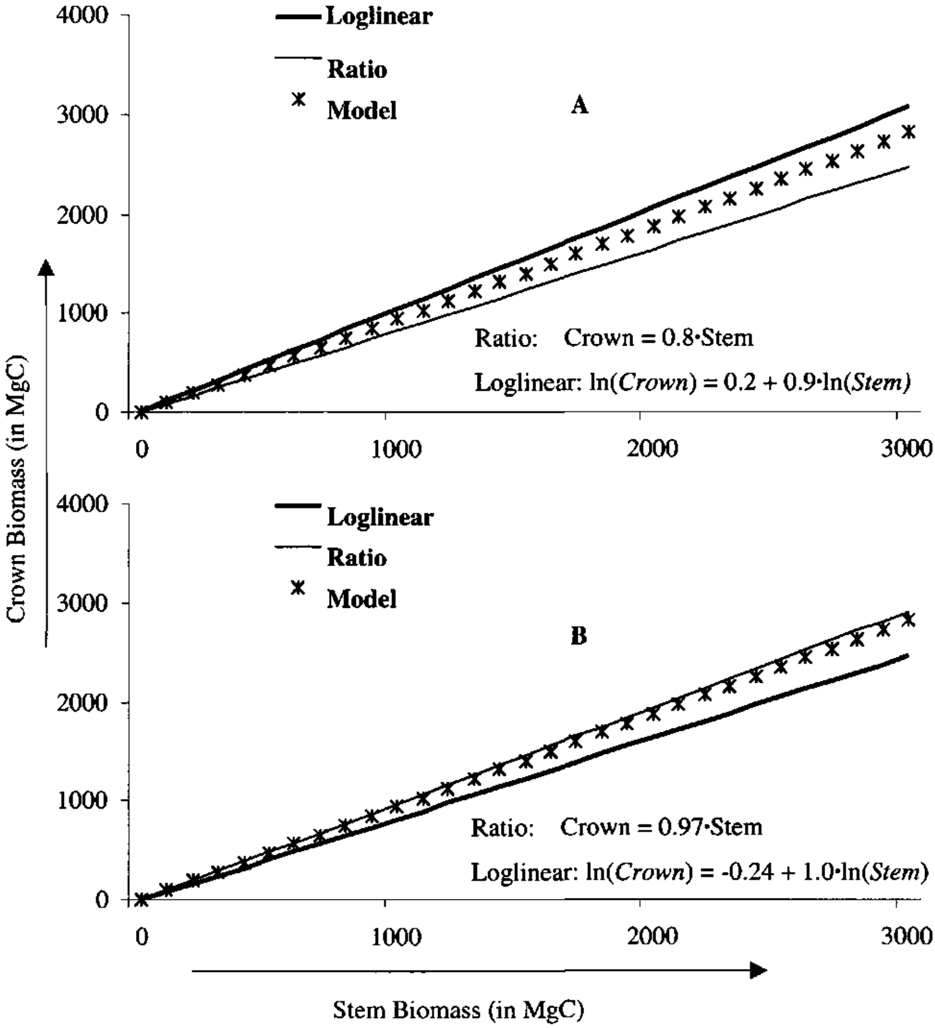


Figure 6.5. *Relation between crown biomass and stem biomass allocation, based on ratio and loglinear regression, both derived from field data, and the linear allocation ratio, as applied in the simulation model. In the simulation model, the crown allocations were separated in the following pools: non-harvestable stems (20% of total stem volume), branches (30% of total stem volume), and leaves and twigs (25% of stem volume). A = pine species; B = oak species.*

Historical land-use change and related carbon storage depletion

In the Highlands of Chiapas the area of closed forest decreased from 68 to 15% of the total area. Disturbed forests (secondary, disturbed and open forests) increased from 13 to 41%, whereas non-forest land increased from 18 to 44%. In Juznajab La Laguna only a significant shift was detected from closed pine-oak forest toward disturbed pine-oak forest, whereas the non-forest area increased from 12 to 19% (Table 6.4).

Table 6.4. *Land use/Land cover (LU/LC) from 1974 to 1996 in percentage of total area in the Highlands and Juznajab La Laguna, and estimated annual and total reductions in C-stock.*

Land use / Land cover class	**Highlands**				**Juznajab La Laguna**			
	1974[1]	**1984**	**1990**	**1996**	**1974**	**1984**	**1990**	**1996**
Pine-Oak Forest	33%	36%	29%	6%	48%	45%	20%	9%
Pine Forest	35%	17%	12%	9%	25%	19%	13%	27%
Tree Fallow	12%	13%	20%	21%	15%	13%	16%	15%
Disturbed Pine-Oak Forest	0%	0%	7%	12%	0%	0%	31%	26%
Open Pine Forest	1%	7%	5%	8%	0%	12%	6%	4%
Total Forest	82%	72%	73%	**56%**	88%	89%	86%	**81%**
Shrubland	12%	9%	10%	22%	10%	4%	10%	13%
Open areas	6%	20%	18%	22%	2%	7%	4%	6%
Total Non-Forest	18%	29%	28%	**44%**	12%	11%	14%	**19%**
Total area (ha)	$306 * 10^3$				$4 * 10^3$			
Annual C-stock reduction[2] (95% CI)	0.9% (0.1%)				0.5% (0.1%)			
Total C-stock reduction[3] (95% CI)	18.0% (3.7%)				10.6% (3.9%)			

[1] *Based on interpreted 1974, 1984, 1990 MSS and 1996 TM satellite images.*

[2] *In percentage of total C-stock, using data from 1974 and 1996 (Formula 2, see text)*

[3] *In percentage of 1974 C-stock.*

The total C-depletion due to LU/LC change between 1974 and 1996 in the Highlands was estimated at around 18% (± 3.5%), whereas Juznajab La Laguna lost an estimated 11% (± 4%) of the 1974 C-pools. The annual decrease in C-storage from 1974 to 1996 due to LU/LC change was 0.9% ± 0.1% of the 1974 C-pools in the Highlands and 0.5 ± 0.1% in Juznajab La

Laguna (Table 6.4). As a result of the LU/LC change and associated C-depletion rate, a 0.5 ± 0.1% annual C-reduction rate was applied in the reference management scenario. A sensitivity analysis was applied to the input value of the C-reduction rate, varying from 0% annual reduction (without considering future LU/LC change in the baseline) to 0.9 ± 0.1% annual reduction (future LU/LC change will be similar as the historical rate found in the Highlands).

Table 6.5. *Comparison of area (in ha) and tree biomass estimations (in MgC) of Juznajab La Laguna, based on data collected in the highlands versus data collected in Juznajab La Laguna (Escandón-Calderón et al., 1999; De Jong et al., 1999b).*

	Highland database		**Juznajab La Laguna database**	
LU/LC Class	Area (ha)	Tree Biomass (MgC)	Area (ha)	Tree Biomass (MgC)
Open areas and shrubland	645	10,583	665	6,920
Secondary and degraded forest	1585	121,289	1665	71,997
Well-developed forest	1306	134,634	1206	145,198
Total (95% CI in % of total)	3536	266,506 (9.8%)	3536	224,115 (5.6%)
Difference Highlands - Juznajab (in % ± 95% CI)		42,391 (15.9% ± 6.4%)		
Total uncertainty [1]		**19.3%**		

[1] $\sqrt{(9.8^2+5.6^2+15.9^2)}$ *(IPCC, 1996).*

The land cover estimations of the composite classes varied somewhat between the two classification methods: between 645 and 665 ha of open areas and shrubland, between 1585 and 1665 ha of secondary or degraded forest, and between 1206 and 1306 ha of well-developed closed pine and pine-oak forest (Table 6.5).

The tree C-pool estimation derived from the two classifications and the two biomass measurements varied between 224,115 MgC ± 5.6% and 266,506 MgC ± 9.8%, with a difference of

42,391 MgC ± 6.4%. The overall uncertainty between the two tree C-pool estimates was 19.3%.

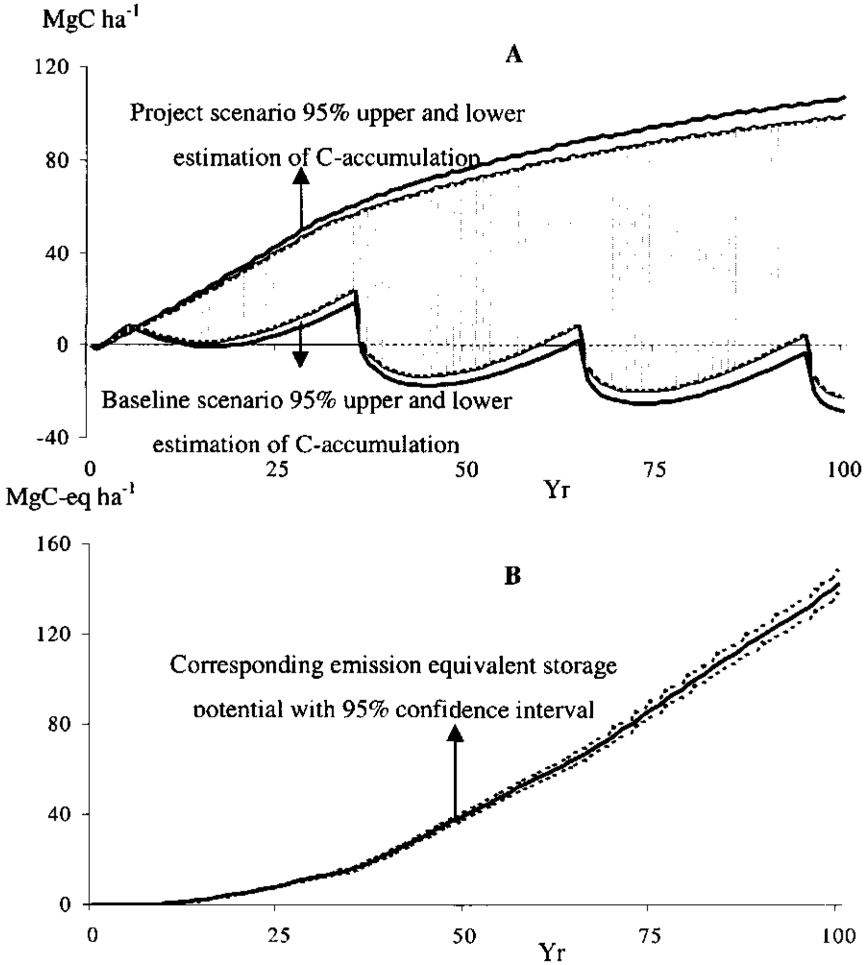


Figure 6.6. *Example of the modeling output of the baseline and project scenario for a specific forest management plot (A) and the corresponding emission equivalent storage potential (B).*

Table 6.6. *Uncertainty analysis of the major parameters used in the C-flux model to estimate the C-flux of forest management.*

Variable or parameter (unit of measurement)	Mean	Uncert. Var. [1] (% of mean)	R.R.V. [2] (% of Var.)	R.S.V. [3] (% of Var.)
Wood density (kg dm^{-3})	0.5	24.2%	38.1%	34.4%
Maximum growth (m^3 yr^{-1})	9.5	23.2%	20.5%	17.8%
Carbon content (kg kg^{-1})	0.5	23.2%	14.6%	11.3%
Litter humification fraction (yr^{-1})	0.1	22.9%	9.1%	4.0%
Humus decomposition fraction (yr^{-1})	0.01	24.3%	8.7%	6.0%
Initial growth (m^3 yr^{-1})	6.1	21.3%	7.7%	3.0%
Leaf assimilation ratio (kg kg^{-1})	0.3	26.2%	6.2%	1.1%
Root assimilation ratio (kg kg^{-1})	0.3	22.9%	4.1%	0.4%
Wood decomposition fraction (yr^{-1})	0.2	26.2%	2.7%	1.5%
Branch assimilation ratio (kg kg^{-1})	0.25	23.3%	2.3%	1.5%
Mortality fraction (yr^{-1})	0.01	30.1%	1.9%	0.8%
Turnover rate root (yr)	7.1	25.4%	0.4%	0.1%
Turnover rate leaf (yr)	2.0	22.4%	0.3%	0.3%
Fence life expectancy (yr)	5.0	21.9%	0.0%	0.0%
Litter decomposition fraction (yr^{-1})	0.8	24.8%	3.0%	0.0%
Turnover rate branch (yr)	8.0	30.1%	0.0%	0.7%
Maximum volume stand (m^3 ha^{-1})	450	23.5%	0.5%	0.5%
Timber life expectancy (yr)	25.1	22.5%	0.3%	0.3%

[1] *Uncert. Var.(in % of mean) = range of input variables / 2 * mean of input variables (in %).*

[2] *R. R. V. (in % of Var.) = Relative reduction in variance of ln (Av-C-Flux) of 100 simulations, keeping the specific input variable constant and fixed, while the other variables remain uncertain (van der Voet, 1993).*

[3] *R. S. V. (in % of Var.) = Relative specific variance of ln(Av-C-Flux), keeping the specific input variable uncertain and all other variables fixed (van der Voet, 1993).*

Uncertainties in carbon flux simulation and modeling variables

The emission equivalent storage potential as a result of the differences between the two management activities rises slowly during the first 35 years (Figure 6.6). This can be attributed to the fact that the two management options initially are not very different in terms of C-dynamics. Only the second harvest in the reference option after about 35 years will cause a steep decline in C-accumulation. Afterwards, the increased difference of accumulated C between the two options is reflected in a steeper rise of the emission equivalent C-storage potential.

Table 6.7. *Carbon mitigation (in 10^3 MgC emission equivalents) of natural forest management in Juznajab La Laguna, Chiapas, México. Calculations are based on the default baseline assumption and when changing the C-reduction rate to 0% or 0.9 ± 0.1%.*

	Year (Year 0 = 2000)				
	2012	**2025**	**2050**	**2075**	**2100**
Default	**15**	**40**	**108**	**222**	**347**
Coefficient of variance of simulation and C-depletion rate [1]	5%,8%	7%,10%	9%,9%	9%,7%	9%,6%
Total uncertainty [2]	9%	12%	13%	11%	10%
C-reduction rate at 0%	**6**	**11**	**39**	**113**	**197**
Coefficient of variance of simulation	5%	7%	9%	9%	9%
Difference with Default in % of Default	60%	73%	64%	50%	43%
Uncertainty in the difference	61%	74%	66%	51%	45%
C-reduction rate of 0.9 ± 0.1%	**21**	**61**	**158**	**301**	**455**
Coefficient of variance of simulation and C-depletion rate, in % of estimation	3%,10%	5%,13%	6%,15%	6%,12%	7%,11%
Uncertainty	11%	14%	16%	14%	13%
Difference with Default in % of Default	44%	53%	46%	36%	31%
Uncertainty in the difference	47%	56%	51%	40%	35%

[1] *Expressed in perecntage of mitigation potantial*

[2] *$\sqrt{(5^2 + 8^2)}$ (IPCC, 1996)*

The parameters that caused the highest variance in the C-flux estimate were wood density, tree growth, carbon content of the biomass, litter humification and humus decomposition (Table 6.6). These parameters either give rise to slow changes in large C-pools (tree growth and humus decomposition), rapid changes in small pools (litter humification), or influence the calculation of the C-stock of the system (carbon content of biomass or soil organic matter and wood density). These results are similar to those reported by Van der Voet (1993).

Carbon mitigation potential and levels of uncertainty.

The total C-mitigation potential of the improved forest management project was estimated at $347 * 10^3$ MgC-eq ± 10% at the end of 100 years. However, if we would apply a 0% C-reduction rate in the reference scenario due to continued land-use change, the mitigation potential would decrease to $197 * 10^3$ MgC-eq ± 9%, whereas a 0.9 ± 0.1% C-reduction rate, as observed in the Highlands area, would result in a mitigation potential of $455 * 10^3$ MgC-eq ± 13%. The overall uncertainty in C-mitigation estimation varied between 5 and 16%. The levels of uncertainty between the three estimates were high, fluctuating between 35 and 74%. The uncertainty of the flux estimation due to expected variations in parameter values varied between 3 and 9% at any time (Table 6.7).

In synthesis, the main sources of uncertainties observed in the calculation of the GHG-offset impact of the forestry project were related to:

1. Classification of LU/LC types, with observed differences of up to around 8% in land-cover estimations.

2. Estimation of C-stocks within each LU/LC type, with uncertainties varying from around 13 to 34% in total C-stock. Calculating C-stocks at a landscape level lead to uncertainties of up to around 10% using one classification method, and close to 20% when combining variations in tree C-stock and LU/LC classifications.

3. Historical evidence of LU/LC changes and related GHG fluxes applied in baselines gave rise to uncertainties of up to about 16%. Temporal and spatial variations in LU/LC change dynamics and variations in C-stock in each LU/LC class were the main factors that were causing this level of uncertainty. Varying baseline assumptions produced differences between 31 and 73% in the C-mitigation calculations, with levels of uncertainty up to 74%.

4. Variation in parameter values to calculate C-fluxes generated uncertainties of up to around 10%.

Each of these sources and/or their combinations will be discussed in the following sections.

Discussion

Notwithstanding the overwhelming literature available about potential biotic mitigation measures, there is still a large gap between accepting that C fluxes can in fact be modified to help mitigate climate change and accepting that this modification can take the form of reliable and verifiable projects (Trexler 1993). A key issue to accept that forestry projects can provide reliable, verifiable reductions of GHG-emission is to point at and reduce the main sources of error in the estimates. The present paper brings elements of proof that the uncertainties in estimating the size of fluxes related to LU/LC changes indeed exist, which they are, and that they can be reduced to acceptable levels (See also Noble, 1998).

Uncertainties in classification of land use/land cover types, carbon stock estimation and historical evidence of carbon depletion

An uncertainty of up to 16% could be attributed to the estimation of the C-stock change associated with the historical LU/LC changes, using the average stock and 95% confidence interval of each LU/LC class as the basis for the calculation. However, the LU/LC change analysis did not take into account possible errors in the LU/LC classification and variation in C-stock within each LU/LC over time. Historical C-stock variation within each LU/LC class could not be determined directly, so they had to be assessed by "best guess". As such, the C-densities measured between 1994 and 1996 in each LU/LC class were used as the "best guesses" for all the evaluated years. Any variation in C-stock densities over time will most likely have caused an underestimation of the loss of C, since the C-densities in the LU/LC classes were presumably higher in the 1970s and 1980s than in the1990s (De Jong et al., 1999). As these fixed C-stock data were only used in the baseline calculations, this error would lead to an underestimation of the GHG-offset potential. The assumption that all C will be lost or gained from one period to another due to LU/LC change is also prone to error. Especially the impact of LU/LC change on soil organic matter dynamics is unclear. Although a positive correlation was observed between the amount of soil organic matter and aboveground living biomass in forests of the Highlands (Figure 6.7), in secondary vegetation this relation was less clear, probably due to differences in management histories (Figure 6.8; Van der Wal, 1999).

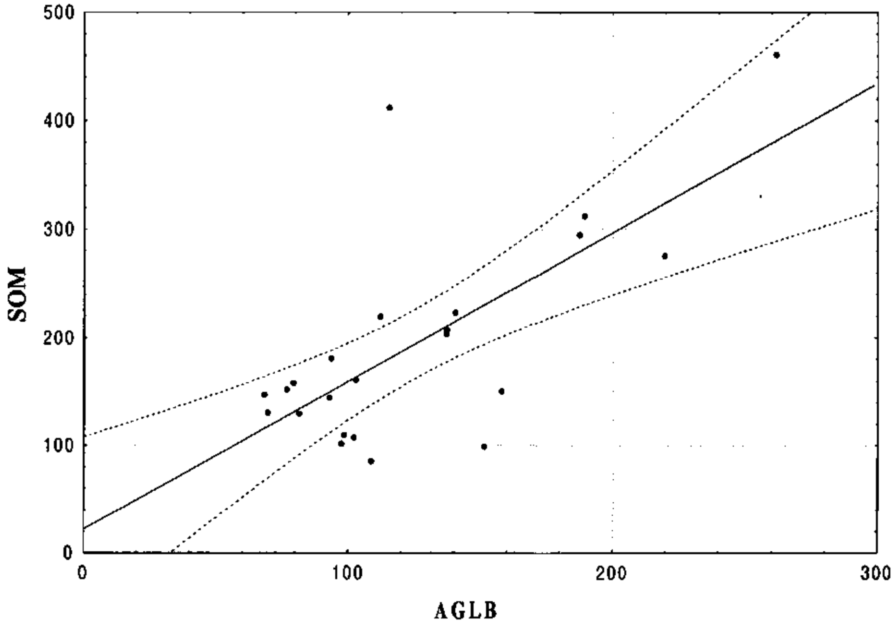


Figure 6.7. *Relation between C density (Mg ha^{-1}) in soil organic matter (SOM) and above-ground living biomass (AGLB) in forests (n = 22; SOM = 6.1 + 1.4·AGLB; R^2 = 0.67; p < 0.001 for angle coefficient).*

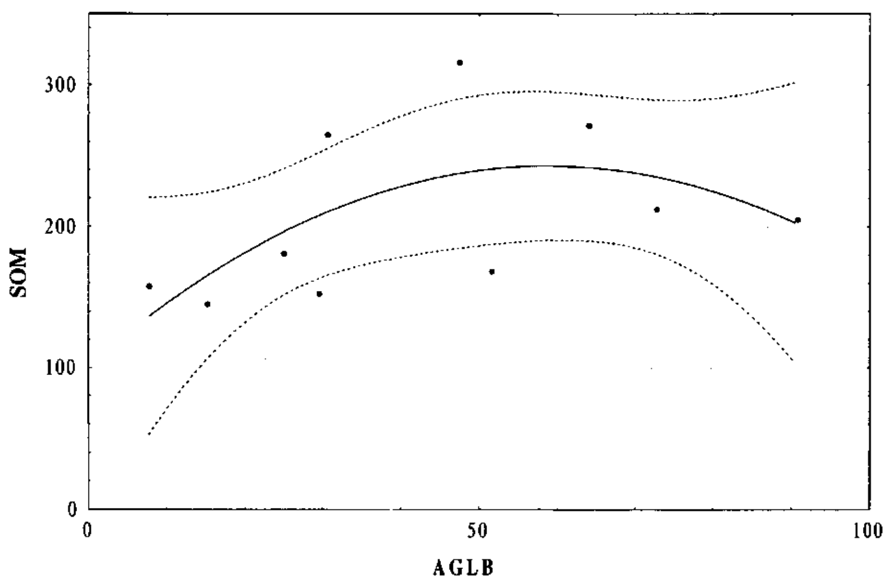


Figure 6.8. *Relation between C-density (Mg ha^{-1}) in soil organic matter (SOM) and above-ground living biomass (AGLB) in shrub and tree fallows (n = 10; SOM = 102 + 4.8·AGLB – 0.04·AGLB^2; R^2 = 0.35; p = 0.10-0.18 for all coefficients).*

Comparing the two 1996 classification databases resulted in a variation in tree C-stock estimation of around 16 ± 6% at the landscape level, with an overall uncertainty of around 19% between the two methods. The LU/LC types in the classification system in Juznajab La Laguna were stratified to reflect on the one hand dominance of species groups and on the other hand C-densities of those C-pools that are expected to change significantly in time because of management. In this case the pools that are prone to rapid change as a result of management are located in the woody biomass. In the classification applied in the Highlands, LU/LC types were stratified with the primary goal to evaluate the processes of deforestation and forest fragmentation. In the latter classification system some LU/LC classes represent a relatively high variation in C-densities, particularly the open and disturbed forest and secondary vegetation classes. As such, the tree C-densities estimated with the community level classification probably provide a better estimate of what is actually in the field, whereas the classification at the sub-regional level lead to an overestimation of the tree C-densities.

Flux estimations

The flux model that was used to estimate future C-fluxes is based on the concept of an equilibrium state, in which a forest system will tend to approach a matter balance under undisturbed development, such that nutrient uptake and mineralization occur approximately at the same rate (Ulrich, 1987; Oldeman, 1990; Jandl, 1998). Seasonal and annual variations are currently not considered but are certain to cause oscillations around the average. Ecological disasters, such as the severe forest fires of 1998, can also change drastically project outcomes.

The current model assumes that disturbances induced by humans do not change the system dynamics. There is evidence that this is true over a considerable range of ecological scales (Oldeman, 1990). This indeed facilitated the calculation of the effect of varying forest management activities on expected C-fluxes. Thus the values and variation in values of the change parameters were considered the same for both forest management scenarios, except for harvesting intensity and frequency (Table 1). However, management can divert certain ecosystem dynamics, such as species composition and nutrient cycling. This in turn will certainly influence the C-pool change dynamics, particularly of standing tree biomass, litter, and soil organic matter pools. For example, oak and pine-oak forests have the highest biomass densities per surface area (Table 6.2). Selective silviculture and management of natural regeneration could be used to steer (part of) the forest composition towards stands dominated by oaks, to increase total biomass per unit area. This shift in canopy composition in turn influences the micro-climatic conditions in the undergrowth (Vester, 1997) and thus affects undergrowth species composition and biomass decomposition. González-Espinosa et al. (1995a and b) found that oak-dominated forests are richer in understory tree and shrub species than pine- or pine-oak-dominated forests. Salamanca et al. (1998) and Kaneko and Salamanca (1999) confirmed the old forester's wisdom that pine litter decomposition is slower than oak litter decomposition and both separately decompose slower than litter mixtures of these species. The complexity of the interactions between all these processes cannot be accommodated yet in one computer model, so simplifications in the simulation were necessary. The uncertainty analysis performed to help understand which parameters of the model are causing the highest variance in the projected outcome indicated that about 38% of total uncertainty of the flux estimation is caused by varying tree growth. This agrees with architectural data (Oldeman, 1990) and indeed, a high variation in tree growth was observed in the forest of Juznajab La Laguna due, among others things, to local variation in site conditions (Figure 6.9). Needless to say that the low levels of uncertainty in the flux estimation do not decrease the levels of uncertainty due to variations in forest structure, species composition and growth.

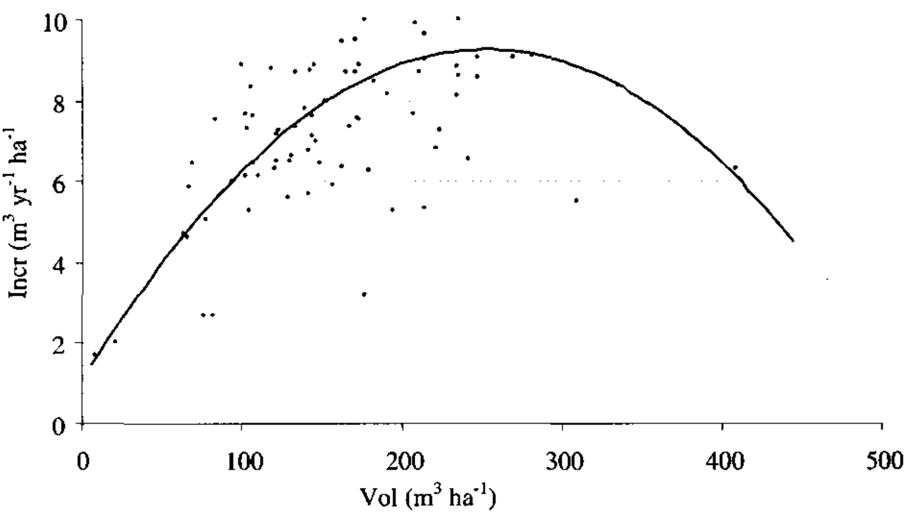


Figure 6.9. *Relation between Current Annual Increment of pine trees (Incr; in m^3 yr^{-1} ha^{-1}) and total standing tree volume (Vol; m^3 ha^{-1}), based on data collected in Juznajab La Laguna ($n = 102$; $CAI = 0.98 + 0.067 \cdot Vol - 0.00013 \cdot Vol^2$; $r = 0.49$; $p = 0.2$ for intercept and $p < 0.001$ for other coefficients).*

The data that were used to calculate the logistic-type growth curve were collected from one time measurements in plots with different stand structure and species composition. Although this type of growth models have been widely used in forestry (Cannel and Milne, 1995; Van Kooten et al., 2000), the growth assumption could not be validated for this particular site. Estimates of the level of error of the flux model itself could therefore not be measured. As validation of models and change parameters in forestry is a long-term process, the fact that the level of this uncertainty remains unknown is one of the major concerns of the monitoring program in the Scolel Té Pilot Project (Scolel Té, 1998). Therefore a set of around 40 permanent monitoring plots have been established recently in various LU/LC types and ecological conditions and as data become available from monitoring, these will be used to improve the C-flux modeling (De Jong et al., 1997).

Baseline assumptions

In the example, uncertainties of up to 74% in the offset calculations were observed between the three baseline assumptions. This level of uncertainty is about four to 10-fold higher than the uncertainty observed in C-pool quantification methods or variances in classification data. If only the land-use history data from Juznajab were used, the default baseline setting could be argued. However, the community could state that the regional trend of biomass reduction due to land-use change also applies to their community, so that their historical forest conservation measures are contemplated and furthermore this would avoid perverse competition from communities that has a high historical rate of deforestation. On the other hand, it could be argued that deforestation is a national and community problem , so that the 0% C-reduction rate applies.

This means that, even though C-pools and fluxes can be estimated within acceptable confidence intervals, we will still be faced with the problem how to define baselines that allow calculating reliably and verifiably the additional effect of the project. Baseline determination unavoidably has a judgmental component. This means that this depends not just on methodology, but on a set of criteria and indicators that keep the methodology critical and honest (Chomitz, 1998). Appropriate guidelines should set standards for baseline definitions that are acceptable and based on international agreements. These guidelines in turn do not necessarily guarantee precision of measurements, but will avoid systematic miscalculations. Part of the guidelines could indicate the levels of acceptable error in the estimates of C-pools, C-fluxes and C-offsets. The remaining uncertainties at the project level would then tend to cancel each other out, if various projects are aggregated, and thus would not affect global GHG emissions (Heister, 1997).

Concluding remarks

Mexico is very keen to explore the potential of the clean development mechanism, especially in the forestry sector (CICC, 1999). Since about 85% of the forested land is communally owned, their forest resources can only be conserved and managed when this option is economically attractive for the owners (Thoms and Betters, 1998). Combining benefits from sustainable forest exploitation with economic incentives based on ecological services is a very cost-efficient alternative for those who want to both conserve biologically important forest resources and alleviate the critical poverty of rural communities that have to make a living from these forests (Tipper et al., 1998). Farmers of Juznajab La Laguna have already demonstrated interest in such financial mechanisms, as more than 60 farmers are participating in the Scolel Té Pilot Project with farm forestry on private land (De Jong et al., 1997).

The uncertainty science described in this paper cannot eliminate all errors in the estimation of the C-mitigation of a forestry project. However, it points to the main sources of error in the calculations of GHG-offset potential. Thus, it could take away some of the doubts that currently exist about the reliability and verifiability of forestry as a GHG-mitigation measure.

Acknowledgements

I would like to thank Roelof A.A. Oldeman, Richard Tipper, Omar Masera, and Susana Ochoa-Gaona for their critical review of an earlier version of this manuscript. Thanks also to Miguel-Angel Castillo-Santiago and John A. Taylor for their comments on and assistance with the development of the community forest management plan. The land-use data were made available to me by Susana Ochoa-Gaona, for which I am very grateful. Juan López-Morales and Guadalupe Pérez-Ecobar assisted with field data collection and laboratory analysis. Collection of forest management data used in this paper was partly financed by the UK Department for International Development's Forestry Research Programme, project No.R7274, through Richard Tipper, University of Edinburgh. The International Energy Agency's Greenhouse Gas R & D Programme financed collection of land-use data under contract of University of Edinburgh. The U.S. Environmental Protection Agency financed the collection of information on carbon densities, under the Cooperative Agreement CR822200 between US-EPA and ECOSUR.

References

Arp, P.A. and McGrath, T.P. 1987. A parameter-based method for modelling biomass accumulations in forest stands: Theory. Ecological Modelling 36: 29-48.

Ayala López, R.A. 1999, Ecuaciones para estimar biomasa de pinos y encinos en la meseta central de Chiapas. Thesis. Universidad Autónoma Chapingo, Div. Ciencias Forestales.

Bartelink, H.H. 1998. Simulation of growth and competition in mixed stands of Douglas-fir and beech. Thesis Landbouwuniversiteit Wageningen. 222 pp.

Breedlove, D.E. 1981. Flora of Chiapas, Part 1: Introduction to the Flora of Chiapas. The California Academy of Sciences, San Francisco, CA.

Brown, S., Cannell, M.G.R, Kauppi, P., Heuveldop, J., Sathaye, J., Singh, N. and Weyers, S. 1995. Establishment and management of forests for mitigation of greenhouse gas emissions. IPCC 1995 Assessment. Cambridge University Press.

Cano-Capri, J. 1988. Elementos básicos del sistema de manejo regular. In: El Sistema de manejo regular en los bosques de México. Chapingo, México, pp. 17-49.

Cannell, M.G.R. and Milne, R. 1995. Carbon pools and sequestration in forest ecosystems in Britain. Forestry 68: 361-378.

CASFOR, 1999. http://www.efi.fi/projects/casfor/about.html.

CICC (Comité Intersectorial para el Cambio Climático), 1999. Programa Nacional de Acción Climática, Documento de Consulta Pública. Mexico D.F. 189 pp.

Chomitz, K. 1998. Baselines for greenhouse gas reductions: problems, precedents, solutions. Draft paper. World Bank, Carbon Offsets Unit. http://www-esd.worldbank.org/aij/-meth.htm.

Cook, E. 1995. Lifetime commitments: Why climate policy-makers can't afford to overlook fully fluorinated compounds. WRI Issues and Ideas. 16 pp.

De Jong B., Soto, L., Montoya, G., Nelson, K., Taylor, J. Tipper, R. 1997. Forestry and agro-forestry alternatives for carbon mitigation: an analysis from Chiapas, Mexico. In: Adger W., Pettenella D., Whitby M.C., (eds) Climate Change Mitigation and European Land Use Policies. CAB International. Wallingford, UK, pp. 263-284.

De Jong, B.H.J. and Montoya-Gómez, G. 1994. Sustainable management of forest resources: a proposal for the highlands of Chiapas, Mexico. Pages 7-17 in Proceedings of the 1994 symposium on systems analysis in forest resources: management systems for a global economy with global resource concerns, Sept 6-9, 1994. Pacific Grove, CA.

De Jong, B.H.J., Cairns, M.A., Ramírez-Marcial, N., Ochoa-Gaona, S., Mendoza-Vega, J., Haggerty, P.K., González-Espinosa, M., and March-Mifsut, I. 1999. Land-use Change and Carbon Flux between the 1970s and 1990s in the Central Highlands of Chiapas, Mexico. Environmental Management 23(3): 373-385.

De Jong, B.H.J., Ochoa Gaona, S., Soto-Pinto, L., Castillo-Santiago, M.M., Montoya-Gómez, G., Tipper, R. and March-Mifsut, I. 1998. Modelling Forestry and Agroforestry Opportunities for Carbon Mitigation at a Landscape Level. In: Nabuurs, G.J., Nuutinen, T., Bartelink, H. and Korhonen, M. (Eds.). Forest Scenario modelling for ecosystem management at landscape level, EFI Proceedings No 19: 221-237.

Dixon, R.K., Andrasko, K.J., Sussman, F.G., Lavinson, M.A., Trexler, M.C., and Vinson, T.S. 1993 Forest Sector Carbon Offset Projects: Near-term Opportunities to Mitigate Greenhouse Gas Emissions. Water, Air and Soil Pollution 70: 561-577.

Dixon, R.K., Schroeder, P.E. and Winjum, J.K. (Eds.). 1991 Assessment of Promising Forest Management Practices and Technologies for Enhancing the Conservation and Sequestration of Atmospheric Carbon and Their Costs at the Site Level. EPA/600/3-91/067, US-EPA, Washington D.C.

Escandón-Calderón, J., De Jong, B.H.J., Ochoa-Gaona, S. March-Mifsut, I., and Castillo-Santiago M.A., 1999. Evaluación de dos métodos de estimación de biomasa arbórea a través de datos LANDSAT TM en Jusnajab, Chiapas, México: estudio de caso. *Investigaciones Geográficas Boletin* (In press).

González-Espinosa M., N. Ramírez-Marcial, P.F. Quintana-Ascencio, and M. Martínez-Icó. 1995b. La utilización de los encinos y la conservación de la biodiversidad en Los Altos de Chiapas. Pages 183-197 *in* Memorias del tercer seminario nacional sobre utilización de encinos, 4-6 de Noviembre de 1992. Reporte Cientifico UANL, Número Especial 15. Linares, Nuevo Leon, Mexico.

González-Espinosa, M., S. Ochoa-Gaona, N. Ramírez-Marcial, and P.F. Quintana-Ascencio. 1995a. Current land-use trends and conservation of old-growth forest habitats in the highlands of Chiapas, Mexico. Pages 190-198 *in* M.H. Wilson and S.A. Sader (eds.) Conservation of Neotropical Migratory Birds in Mexico. Maine Agricultural and Forest Experiment Station, Miscellaneous Publication 727, Orono, ME.

Heister, J. 1997. Towards a Methodology for Quantifying Greenhouse Gas Offsets from Joint Implementation Projects and Activities Implemented Jointly. Draft Working Paper, World Bank, Global Climate Change Unit. http://www-esd.worldbank.org/aij/meth.htm. 19 pp.

Houghton, J.T., Jenkins, G.J., Ephraums, J.J. 1990. Climate Change: The IPCC Scientific Assessment. Univ Press, Cambridge, UK. 364 pp.

Houghton, J.T., Meira Filho, L.G., Griggs, D.J., Maskell, K., 1997. An introduction to Simple Climate Models used in the IPCC Second Assessment Report. IPCC. WGI. Technical Report no II. 51 pp.

INEGI (Instituto Nacional de Estadística, Geografía e Informática), 1985. Carta Edafológica 1:250,000. Tuxtla Gutiérrez E15-11. INEGI, México D.F.

IPCC, 1996. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories Reporting Instructions (Volume 1), Annex 1, Managing Uncertainties. 6 pp.

Jandl, R. 1998. Modeling processes in forest soils – problems, simplifications and caveats. Ecological Engineering 10: 33-51.

Jansen, M.J.W., Rossing, W.A.H., and Daamen, R.A. 1994. Monte Carlo estimation of uncertainty contributions from several independent multivariate sources. In: Grasman, J and Van Straten, G (eds.) Predictability and nonlinear modelling in natural sciences and economics. Kluwer Acad.Publ., Dordrecht, The Netherlands: 334-343.

Kleijnen, J.P.C., 1994. Sensitivity analysis versus uncertainty analysis: when to use what? In: Grasman, J and Van Straten, G (eds.) Predictability and nonlinear modelling in natural sciences and economics. Kluwer Acad.Publ., Dordrecht, The Netherlands: 322-333.

Masera, O.R., Ordoñez, M.J., and Dirzo, R. 1997. Carbon emissions from Mexican forests: Current situation and long-term scenarios. Climatic Change 35: 265-295.

Montoya, G., De Jong, B., Nelson, K., Soto, L., Farías, P.J., Taylor, J., Tipper, R. 1995. Desarrollo forestal sustentable: Captura de carbono en las zonas tzeltal y tojolabal del estado de Chiapas. Cuadernos de Trabajo No.4 Instituto Nacional de Ecología. México, DF. 79 pp.

Nabuurs, G.J. and Mohren, G.M.J. 1993. Carbon fixation through forestation activities: A study of the carbon sequestering potential of selected forest types, commissioned by the Foundation Face. Inst. for Forestry and Nature Research (IBN-DLO), Wageningen, The Netherlands, IBN Research Report 93/4. 204 pp.

Nabuurs, G.J. and Mohren, G.M.J. 1995. Modelling analysis of potential carbon sequestration in selected forest types. Canadian Journal of Forest Research 25: 1157-1172.

Noble, I., 1998. The Gross-net Issue. http://www.igbp.kva.se/grossnet.html, 6 pp.

Ochoa-Gaona, S and González-Espinosa, M. 1999. Land use and deforestation in the Highlands of Chiapas, Mexico. Applied Geography 20: 17-42.

Oldeman, R.A.A. 1990. Forests: Elements of Silvology. Springer-Verlag, Berlin. 624 pp.

Parra-Vázquez, M.R., T. Alemán-Santillán, J. Nahed-Toral, L.M. Mera-Ovando, M. de la López-Mejía, and A. López-Meza. 1989. El subdesarrollo agrícola en los altos de Chiapas. Universidad Autónoma de Chapingo, Chapingo, México. 405 pp.

Puhl, I. 1998. Status of Research on Project Baselines Under the UNFCCC and the Kyoto Protocol. OECD and IEA Information Paper, Paris, France. 26 pp.

Richards, K.R. and Stokes, C. 1994. Regional studies of carbon sequestration: a review and critique. US Department of Energy, DE-AC06-76RLO 1830. 40 pp.

Riley, R.H., Phillips, D.L., Schuft, M.J., and Garcia, M.C. 1997. Resolution and error in measuring land-cover change: effects on estimating net carbon release from Mexican terrestrial ecosystems. International Journal of Remote Sensing 18(1): 121-137.

Scolel Té, 1998. http://www.ed.ac.uk/~ebfr11; http://www.ecosur.mx.

Schlamadinger, B. and Marland, G. 1996. The role of forest and bioenergy strategies in the global carbon cycle. Biomass and Bioenergy 10: 275-300.

Thoms, C.A. and Betters, D.R. 1998. The potential for ecosystem management in Mexico's forest ejidos. Forest Ecology and Management 103: 149-157.

Tipper, R. and De Jong, B.H.J. 1998. Quantification and regulation of carbon offsets from forestry: comparison of alternative methodologies, with special reference to Chiapas, Mexico. The Commonwealth Forestry Review 77(3): 219-228.

Trexler, M C and Haugen, C. 1994 Keeping it green: evaluating tropical forestry strategies to mitigate global warming. World Resources Institute, Washington DC, USA.

Trexler, M.C. 1993. Manipulating biotic carbon sources and sinks for climate change mitigation: Can science keep up with practice? Water, Air, and Soil Pollution 70: 579-593.

Trines, E.P. 1998. SGS' carbon offset verification service. Commonwealth Forestry Review 77(3): 209-213.

Ulrich, B. 1987. Stability, elasticity and resilience of terrestrial ecosystems with respect to matter balance. Ecological Studies 61: 11-49.

UNFCCC, 1997. Kyoto Protocol to the United Nations framework convention on climate change. FCCC/CP/1997/L.7/Add.I. Geneva. 24 pp.

UNFCC, 1998a. CC:INFO/AIJ: Methodological Issues. http://www.unfccc.de/program/aij/-inf3.html.

UNFCCC, 1998b Activities implemented jointly under the pilot phase: Update on activities implemented jointly. FCCC/SBSTA/1998/INF.3. http://www.unfccc.de/program/aij/.

Van der Voet, H. 1993. Appendix K: Uncertainty analysis of the model CO_2FIX. In: G.J. Nabuurs and G.M.J. Mohren. Carbon fixation through forestation activities: A study of the carbon sequestration potential of selected forest types, commissioned by the Foundation Face. IBN-DLO Research Report 93/4, Wageningen, The Netherlands. pp.181-205.

Van der Voet, H. and Mohren, G.M.J. 1994. An uncertainty analysis of the process-based growth model FORGRO. Forest Ecology and Management 69: 157-166.

Van der Wal, H. 1998. Chinantec Shifting Cultivation and Secondary Vegetation: A case-study on secondary vegetation resulting from indigenous shifting cultivation in the Chinantla, Mexico. BOS Document No 19, BOS Foundation, Wageningen, The Netherlands. 121 pp.

Van der Wal, H. 1999. Chinantec shifting cultivation: InTERAcTIVE landuse. A case-study in the Chinantla, Mexico, on secondary vegetation, soils and crop performance under indigenous shifting cultivation. Treebook 3, Treemail Publishers, Heelsum, The Netherlands. 162 pp.

Vester, H. 1997. The trees and the forest. The role of tree architecture in canopy development; a case study in secondary forests (Aracuara, Colombia). Ponsen and Looijen, The Netherlands. 180 pp.

Winjum, J. K., Dixon, R.K., and Schroeder, P. E. 1992 Estimating the global potential of forest and agroforest management practices to sequester carbon. Water, Air and Soil Pollution 64: 213-222.

7. A FRAMEWORK FOR MONITORING AND EVALUATION OF CARBON MITIGATION BY FARM FORESTRY PROJECTS: EXAMPLE OF A DEMONSTRATION PROJECT IN CHIAPAS, MEXICO[5]

Introduction

Since the rise of international concern about climate change due to anthropogenic greenhouse gas emissions, there has been considerable interest in the potential for increasing the storage of carbon in terrestrial vegetation through forest conservation, afforestation, farm forestry and other methods of land management. Several studies have indicated that the global potential for enhancing carbon storage in forest and agricultural ecosystems may be considerable. (Dixon et al, 1991; Dixon et al, 1993; Schroeder et al, 1993, Masera et al, 1995; De Jong et al, 1995). Where these systems replace low biomass cropping or pasture systems or provide economic alternatives to the conversion of tropical forests they reduce the net flux of CO_2 to the atmosphere by: - 1) accumulating carbon (C) in new trees on agricultural land, - 2) protecting stocks of C in existing forest biomass and - 3) substituting energy intensive materials and GHG emitting fuels.

Some preliminary estimates of the potential area available for carbon sequestration in Mexico are (Trexler and Haugen, 1995): 4.5 x 10^6 ha for farm forestry (with a C-sequestration potential of 33.3 to 113.4 x 10^6 MgC), 1 x 10^6 ha for plantations (30.7 to 85.5 x 10^6 MgC), and 30 x 10^6 ha for natural regeneration (1 to 3 x 10^9 MgC). Furthermore, they consider that, by making established agriculture more productive and sustainable (e.g. by substituting slash-and-burn agriculture for sustainable permanent agriculture), forests that were once part of the slash-and-burn cycle can be allowed to recover, and agricultural expansion onto remaining forest areas can be curbed. They suggest that up to 6.1 x 10^6 ha (with a sequestration potential of 348.3 to 714.9 x 10^6 MgC) could be saved from deforestation until 2040. These estimates for C-sequestration do not consider possible displacement of fossil fuel energy by biomass (Schlamadinger and Marland, 1996).

If this potential is to be realized it will be necessary to devise practical schemes based upon appropriate economic mechanisms that will deliver GHG mitigation in objectively verifiable,

---

[5] *Published as: De Jong, B.H.J., Tipper, R., and Taylor J. 1997. A Framework for Monitoring and Evaluation of Carbon Mitigation by Farm Forestry Projects: example of a Demonstration Project in Chiapas, Mexico. Mitigation and Adaptation Strategies for Global Change, Vol. 2 (1997): 231-246.*

sustainable, and socially and environmentally responsible ways. Dixon et al (1993) and Masera et al (1995) consider agroforestry to be the most promising alternative for C-sequestration, in terms of biomass productivity and cost-efficiency. Initial studies by De Jong et al. (1995) indicated that in regions such as Chiapas, the most appropriate methods to enhance carbon storage on land managed in smallholdings are the introduction of trees within agricultural systems as crop-tree combinations or the development of small to medium scale plantations. Together, such developments are referred to as "farm forestry" (Foley and Barnard, 1984). On communally held areas of natural forest or secondary vegetation the main sequestration strategy should be to restore forest ecosystems, and to conserve and manage the tree stock in initiatives referred to as "community forestry" (Foley and Barnard, 1984). In the feasibility study carried out by De Jong et al (1995) in Chiapas, southern Mexico, five farm forestry systems were considered to be technically, socially and economically viable, including live fences, coffee with shade trees, strip plantations in abandoned pasture, tree enrichment of fallows, and taungya. The *ex-ante* estimated increase in carbon density of the systems in relation to the actual systems without the farm forestry project (See Figure 7.1) varied from 16.7 to 176.3 MgC ha^{-1} (averaged over a 150 year rotation), with the lowest potential for living fences and the highest for the plantation systems (taungya and enriched fallow).

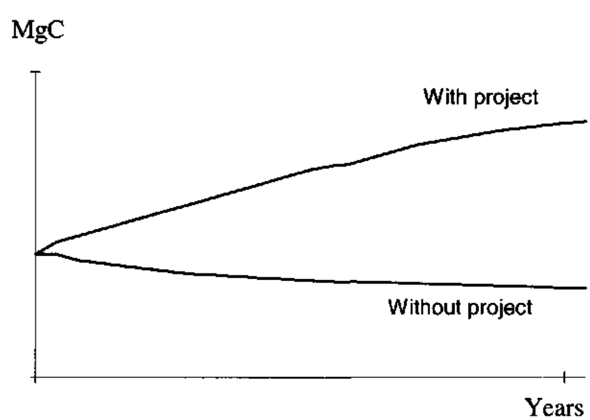


Figure 7.1. *Hypothetical effect of a farm forestry project on C-mitigation, compared to a without-project baseline*

The results of the study suggest that finance provided on the basis of C-sequestration, within the range of 5 to 15 US\$ Mg^{-1} C, can be used to initiate various farm forestry systems that will, in turn, provide a self-sustaining flow of outputs of commercial and subsistence products. This is the context of an international pilot project for carbon sequestration by forestry and agroforestry, being developed in Chiapas, southern Mexico. The objective of the project is to develop a model for carbon sequestration that will be economically viable and technically reproducible in similar regions of Mexico and Latin America, where large areas of land are managed by smallholder (campesino) farmers in both individual and communal units.

Farm forestry projects for GHG mitigation would be characterized by: - numerous participants, organized in various ways, but mainly individuals and small groups, - generally varied, small-scaled systems, replicated over large areas, and - site specific management, with individual adaptations due to personal interest, local conditions, and previous experiences. It is argued that due to these specific characteristics the methods of sequestration assessment, monitoring and evaluation proposed for large-scale forest preservation or plantation type projects will require adaptation or modification if they are to be applied to farm forestry schemes. In this paper we deal specifically with the monitoring and evaluating farm forestry developments, taking into consideration the probable social, institutional and technical constraints of such initiatives. The specific objectives of this paper are: - To clarify the main conceptual issues of assessment, monitoring and evaluation as they will relate to farm forestry carbon sequestration projects, - to describe the framework for monitoring and evaluation proposed for the pilot project, its purpose and its social, institutional and technical context, - to propose a set of criteria which monitoring and evaluation systems should fulfill in the context of farm forestry type projects.

Assessment, Monitoring and Evaluation of Forestry Projects

Carbon Sequestration Assessment

In this paper we refer to the Carbon Sequestration Assessment as the procedures used to define, at the institutional level (for the purposes of project resource allocation and of national emissions inventory) the sequestration impact of a given forestry project. As stated by Swisher (1992), "the relevant unit of measurement for carbon storage in forestry projects is the increment in CO_2 flux, expressed as tons of carbon-equivalent (MgC) out of the atmosphere, compared to existing conditions (in the case of carbon removal), or to a reference condition (in the case of prevention). The score keeping procedure should explicitly account for the carbon storage of the land-use without the project."

Various methods and models are available to assist the assessment of sequestration projects. These vary in scope and complexity according to the scale and context at which they are designed to apply (Table 7.1). For example, the relationship between climate variables and major forest types and their respective biomass provided by Brown and Lugo (1982) is useful when considering the CO_2 fluxes associated with land use changes at country or continental scales, but would not take account of variations in soil type, topography and land use practices that would be highly influential at scales of tens of km^2 or below.

The CO2FIX model by Nabuurs and Mohren (1993) and the model described by Schlamadinger and Marland (1996), provide an estimate of carbon uptake and storage at the level of a stand of trees, and is driven by an equation describing the predicted growth (which can be derived locally from inventory data). The models also account for the fractions of carbon stored in different components of biomass - soil, harvested products, litter and leaves. However, these models do not account for shrub or herbaceous biomass C-fluxes, which are typical components of agroforestry systems.

Table 7.1. *Methods used in the assessment of sequestration by forestry projects.*

Reference	Description of Method	Comments
Brown and Lugo, 1982	Carbon densities of major forest and other vegetation types, related to climate variables	Applicable to large-scale changes in terrestrial vegetation. Does not take account of management options. Mainly applicable to national or regional level assessments rather than projects.
Brown, Gillespie and Lugo, 1989; Gillespie et al., 1992	Carbon densities of tropical forest types related to inventory data	Tree size and form are related to carbon content. Mainly applicable to national or regional level assessments rather than projects.
Fearnside and Malheiros-G, 1996	Region-specific model to predict changes in C storage in Amazonia, based on land-use change modeling (using a Markov matrix of probabilities)	Similar models could be developed for other regions, and would be useful for defining regional baseline scenarios.

Reference	Description of Method	Comments
WRI's LUCS model; Faeth et al., 1994	Estimates future carbon fluxes as a result of land-use change induced by socio-economic pressures.	Is specifically designed to provide assessment of project impact. However, additional socio-economic and environmental variables are required to provide realistic outcomes from scenario assessment.
CO2FIX by Nabuurs and Mohren, 1993	Derives carbon accumulation and storage by a tree plantation over the course of a number of rotations, based on an "expected growth" curve.	Model is adaptable to local variables, such as species and increment. Has to be adapted for agroforestry systems to include interactions between system components, and for management of mixed forests.
Schlamadinger and Marland, 1996	Similar to CO2FIX, but include the possibility of to incorporate fuel and other product substitution.	Model is adaptable to local variables, but has to be adapted for agroforestry systems and for management of mixed forests. Soil carbon dynamics are poorly explained in the model.

With the exception of the WRI's LUCS model (Faeth et al, 1994), the methods described in Table 7.1 are not designed to incorporate project specific variables, such as the projected uptake of forestry techniques by farmers, or the effect of displacing the demand for food, timber or other commodities with the project. The LUCS model itself attempts to simulate the socio-economic pressure for land use change between forest and agricultural land and associated carbon fluxes. The outputs of the LUCS model would require substantial modification in order to take into account local factors such as land tenure arrangements, government intervention in markets, socio-economic stratification and cultural preferences. Therefore, while such methods are invaluable tools in the estimation of biomass changes in different scenarios, they do not provide comprehensive models for project level assessment of farm forestry projects.

The allocation of resources to a project is generally based upon *ex-ante* assessment. However, as only the *ex-post* case will be able to take account of actual project performance, drawing upon the results of monitoring and evaluation, it will be seen as more reliable and therefore used in preference to the original assessment, once projects are completed. Institutional arrangements to take account of the (inevitable) differences between successive assessments are likely to be required. Such differences may be significant in the case of farm forestry projects, whose characteristics and performance may vary considerably between locations and farmers.

Monitoring

Within the context of farm forestry programs the following definitions, adapted from Casley and Kumar (1987), for monitoring and evaluation are proposed:

Monitoring is a continuous assessment of the functioning of project activities, as compared with implementation schedules, the use of project inputs by the target populations, and the effects of the project as measured by physical, social, or biological indicators. Using the data within a management information system carries out the monitoring function. Such a system includes the basic physical and financial records, the details of inputs and services provided to beneficiaries and the data obtained from surveys and other recording mechanisms designed specifically to service the monitoring function. The objectives of monitoring are: - to inform interested parties about the performance of the project, - to adjust project development, - to identify measures that can improve project quality, - to make the project more cost-effective, - to improve planning and measuring processes (including C-Sequestration modeling), and - to be part of a learning process for all actors.

It is suggested that, in the case of forestry carbon sequestration projects, monitoring systems will be required to operate at two main levels. Firstly, to track the stock of tree biomass through periodic inventories or surveys, and secondly to track the development of social, economic and institutional structures (e.g. local trust funds, forest management plans, wood processing facilities, and training programs) that will influence the long-term viability of carbon uptake and storage.

Evaluation

Evaluation is a periodic assessment of the relevance, performance, efficiency and impact of the project in the context of its stated objectives. It usually involves comparisons requiring information from outside the project in time, area, or population. Evaluation will also draw on the management information system but, in selective cases, this will be supplemented with data from impact studies that may be designed and executed outside the project management system itself. Evaluation organizes and appraises the information collected by the monitoring procedures, compares this information with information collected in other ways, and presents the resulting analysis of the overall performance of a project at a time and place that is useful to funding agencies, shareholders, the public, and other stakeholders, so that they can make decisions about:

- whether to continue the project,
- to compare the performance of different projects,
- to make changes in the project,
- to make major changes in the projects' management.

In the case of carbon sequestration forestry projects, periodic evaluation will cover both the current performance of the project and the long-term prospects for storage and uptake of carbon. Periodic project evaluations will be used to determine the official level of sequestration that should be assigned to the project.

Monitoring and Evaluation in the Scolel Té Pilot Project

Dixon et al (1993) suggest five potential monitoring mechanisms: (i) self-reporting, with public access to findings; (ii) consensus reporting by GHG producer, regulatory bodies, and/or third party; (iii) self-reporting, combined with on-site spot-checks; (iv) satellite monitoring; and (v) private third party reporting. In the following sections we describe our approach to monitoring and evaluation that will be tested in a farm forestry C-sequestration pilot project, Scolel Té Pilot Project, to begin in Chiapas, Mexico in 1997, which resembles the self-reporting mechanism, combined with on-site spot-checks.

The project aims to develop a prototype scheme for sequestering CO_2 in sustainable forest and agricultural systems - by providing the institutional structures and organizational methods to ensure that carbon is reliably sequestered for the long term in systems that are economically viable and socially and environmentally responsible. The model should be applicable on a larger scale in similar regions of Mexico and Latin America.

A local trust fund provides Mexican farmers with up to 25 years' financial and technical assistance to implement farm or community scale forestry and agroforestry developments on the basis of the carbon that will be sequestered. Companies or institutions wishing to offset greenhouse gas emissions can purchase "proto-carbon credits" from the local trust fund. The project is managed in the field by a local farmers organization (Unión de Crédito Pajal), with technical and scientific support from national and international scientific institutions (El Colegio de la Frontera Sur and University of Edinburgh). The project is supervised by the Mexican Government's National Institute of Ecology and is registered with both the Mexican and US initiatives for "joint implementation".

Project Management

At the heart of the project's management system is a "farmer-led" planning process known as the "Plan Vivo". Local promoters help farmers to draw up these "working plans" for forestry or agroforestry systems that reflect their own needs, priorities and capabilities (Figure 7.2).

The farmers are responsible for reporting on the progress of their plan and these progress reports are periodically checked by technical staff from the local farmers' organization. At appropriate intervals the data gathered by these monitoring procedures is reviewed by an internal evaluation team to determine whether the project is "on track" in terms of carbon accumulation, whether the potential for future uptake or storage is satisfactory and whether corrective actions need to be initiated.

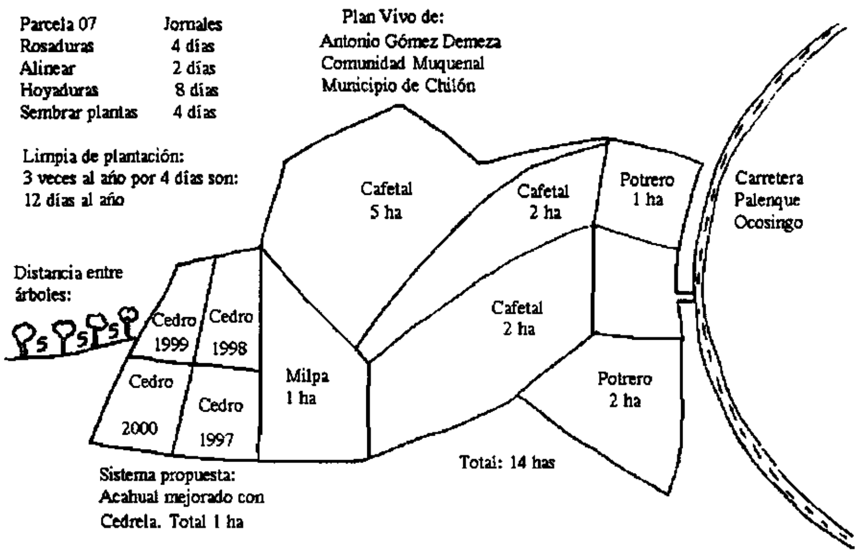


Figure 7.2. *Example of a "Plan Vivo" map.*

The working plans are presented to the Trust Fund and form the basis of discussion between farmers and the Trust, regarding the technical feasibility, the social and environmental impact and the carbon sequestration potential of each plan (Table 7.2). Once plans are judged to be viable, they are then registered with the Trust and become eligible for assistance. The level of financial support to farmers will be related to the expected carbon sequestration.

Table 7.2. *Carbon mitigation categories (MgC ha^{-1}) for three intervention intensities and two ecoregions.*

Intervention	sub-humid sub-tropical	humid tropical
low intensity agroforestry (e.g. living fences)	< 40	< 60
low intensity forestry (e.g. forest reserves) medium intensity agroforestry (e.g. strip planting in fallow)	40 - 60	60 - 100
medium intensity forestry (natural forest management) high intensity agroforestry (e.g. fallow enrichment)	60 - 80	100 - 140
high intensity forestry (e.g. plantations) high intensity agroforestry (e.g. taungya)	> 80	> 140

Once these internal procedures for planning, monitoring and evaluation are established, external evaluation and verification of these procedures can commence. At present, the main institution providing external evaluation is El Colegio de la Frontera Sur (ECOSUR). In due course it is hoped that ECOSUR and other Mexican institutions will develop a national verification and certification system in concert with the National Institute of Ecology - the official regulatory body. Box 1 describes in more detail the responsibilities of each group involved in the project, and Box 2 describes in more detail the factors monitored at each stage in the project cycle.

Box 1. *Responsibilities of Main Groups Involved in Monitoring and Evaluation*

THE FARMERS: will be responsible for (i) preparation of system proposals to be considered, (ii) implementation of the systems, (iii) maintenance of the systems, (iv) system performance reporting, and (v) farmer-to-farmer training.

THE TECHNICAL TEAM (professionals and local promoters): will be responsible for: (i) assessing the viability of proposed systems (in coordination with the farmers and research team), (ii) training of farmers in planning, implementation and performance recording, (iii) estimating C-fluxes of proposed systems, and (iv) assessing project impacts. The training of the farmers should generate farmers' abilities to identify and evaluate their agricultural technologies, to identify specific practices that can be improved, to enhance their knowledge, skills, and appreciation of farm forestry, to design alternative land-use systems, to keep records of their productive activities, including inputs and outputs. The estimation of C-fluxes of the proposed systems will be based on: tree density and spatial arrangements, site conditions, silvicultural treatments, harvesting intensity and periodicity, the management of additional system components, and the previous history of land use.

The Research Team (researchers of El Colegio de la Frontera Sur, a regional research institute, and the University of Edinburgh, UK): will (i) develop C-flux models for each system category and ecological region, (ii) train technical team in system appraisal and C-flux calculations, (iii) train technical team in project impact assessment, (iv) evaluate and refine project planning and implementation assessment procedures, and (v) assess methods for delivery of the sequestration service to potential purchasers.

The Technical and Research Team will verify the quality of the information provided by the farmers through random checks and will evaluate the proposals on their ability to meet sectorial needs where communal resources are used or where member groups are formed around shared ecological zones such as watersheds.

Box 2. *Factors Monitored and Evaluated at Each Stage in the Project Cycle.*

Stage 1. Promotion and Training

The purpose of the project and the processes involved are explained to farmers at meetings. The variety of potential farm forestry systems appropriate for given areas are discussed and the requirements for entry into the scheme are set out. Local promoters are selected and given training in the farm forestry planning procedures.

Stage 1. M&E: Progress in promotion and training, and qualitative feedback from farmers and promoters is recorded. Expected uptake by farmers can be estimated and initial insight into the socio-economic constraints on the project can be obtained. Further training requirements can be determined.

Stage 2. Planning and Processing of Applications to the Scheme

Local promoters will assist individual farmers or small groups to plan appropriate forestry systems. Working plans including sketch maps, descriptions of current vegetation and land uses, descriptions of the forestry system to be established, defined milestones and lists of inputs are drawn up and submitted to the local project management team for assessment. The local management team assess working plans for completeness and technical feasibility, according to documented criteria. Plans that are acceptable are then categorized into probable ranges for C-sequestration, according to criteria provided by the research team. The C-mitigation potential for various representative farm forestry systems applicable to the pilot project area in Chiapas has already been estimated *ex-ante*, using the CO2FIX model (Nabuurs and Mohren, 1993), calibrated with local tree growth data and adapted to the proposed systems (De Jong et al, 1996). Proposals can be either promoting C sinks or avoiding C sources (e.g. sustainable permanent agriculture as an alternative for slash-and -burn agriculture (Figure 7.3). The system proposals will be grouped in potential C-mitigation categories for each ecological zone (Table 7.2). When plans are rejected, farmers will be provided with explanations for the decision, so they will be able to adjust and re-submit the plans. Farmers whose plans are accepted are notified and offered places on the scheme. Details of the financial and technical support available are provided. Agreements are signed with farmers entering the scheme.

Stage 2. M&E: Documented working plans will be used to make estimates of expected sequestration impact. Areas to be planted, planting arrangements, species to be used, and existing vegetation types provide inputs to a C-flux model, that provides an sequestration estimate, over time for each area approved.

STAGE 3. ESTABLISHMENT OF SYSTEMS

Farmers entering the scheme will be provided with financial assistance and technical advice through local promoters, local farmers organizations/credit agencies and by qualified foresters. Farmers will be responsible for reporting on the progress of establishment and on problems, as they occur. Payment of annuities shall be contingent upon the farmer providing adequate records of progress.

Stage 3. M&E: Simple records of establishment will provide the necessary information to begin comparing the actual uptake of CO_2 with expected performance. Farmers will be primarily responsible for collecting data. However, the technical team will conduct internal verification of data collection from a random sample of participants. This process will continue in successive stages, to provide longitudinal comparisons. Problems such as pest attack will be reflected in revised sequestration estimates, and this information will also be used to initiate corrective action.

STAGE 4. MAINTENANCE, EXPANSION AND UPGRADING OF SYSTEMS

Farmers on the scheme continue to maintain their farm forestry plots and may expand or upgrade by modifying their working plans, in consultation with the technical team. This flexibility will allow farmers to gradually increase the area of tree cover on their farms whilst working within the constraints of the supply of labor and materials available.

Stage 4. M&E: Monitoring continues as in Stage 3. Estimates of biomass will be derived from measurements of tree height, diameter and absolute numbers of trees. Comparative storage of C in soils under farm forestry and conventional farming will be monitored on a stratified sample of plots.

Concluding comments

In general terms, the system for proposed for assessing the carbon sequestration effect of the Chiapas Pilot Project can be described as based upon internal monitoring and evaluation with external verification or audit, as advocated by Mitchell and Chayes (1995) in the context of national GHG inventories.

It should be emphasized that while projects are by definition time-limited interventions, they purport to initiate processes that will continue long into the future. Claims for C-sequestration by forestry must, therefore, include plausible arguments indicating the sustainability of the new land-use systems developed. Principles, criteria and indicators for sustainability of forestry systems have been developed by several international agencies, including UNCED (Heissenbuttel et al, 1992) and the Forest Stewardship Council (FSC, 1996). Table 7.3 lists some possible criteria and verifiable indicators that could be used to assess the sustainability of farm forestry systems initiated by a C-sequestration project.

With this in mind, and based on our limited experience to date, we suggest that efficient and effective systems of monitoring and evaluation for farm forestry carbon sequestration projects should fulfill the following criteria.

Integrated

Monitoring and evaluation of carbon storage parameters should be fully integrated with other aspects of project M&E, including financial accounting, input delivery and institutional development milestones. It should be recognized that the long-term viability of the storage and uptake of carbon in the project area will be affected as much, if not more, by the future capacity of local institutions to promote and support economically viable tree-based production systems than by current increments in biomass. Furthermore, the integration of biometric monitoring with other aspects of project management will both reduce costs and enhance the potential for corrective action in response to problems. Thus, in the Pilot Project the monitoring of tree stand development will be undertaken primarily to ensure that corrective measures such as pest control, beating up (re-planting), thinning, and in certain cases pruning, are carried out promptly and effectively. Biomass estimates should be obtainable from the standard mensuration data with minimal additional effort or expense.

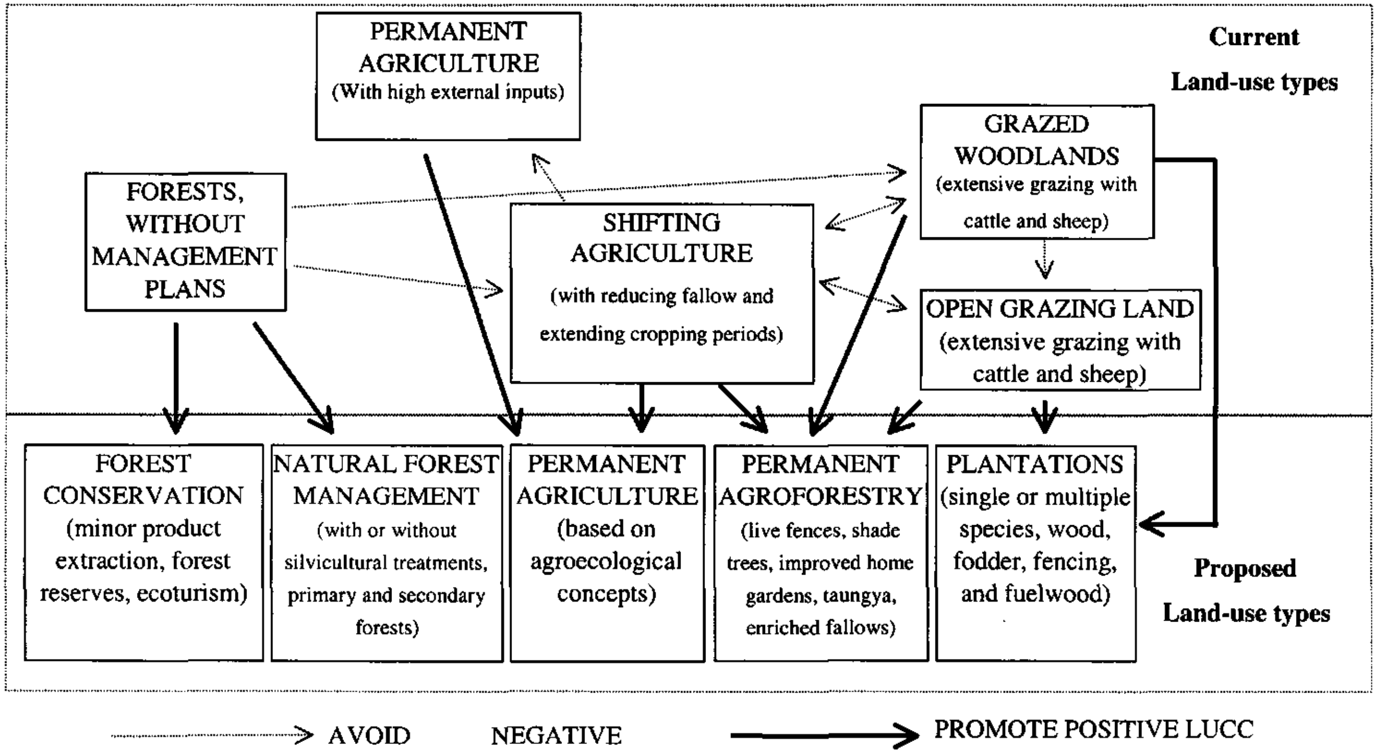


Figure 7.3. *Current and proposed land-use change strategies, based on De Jong and Montoya (1994).*

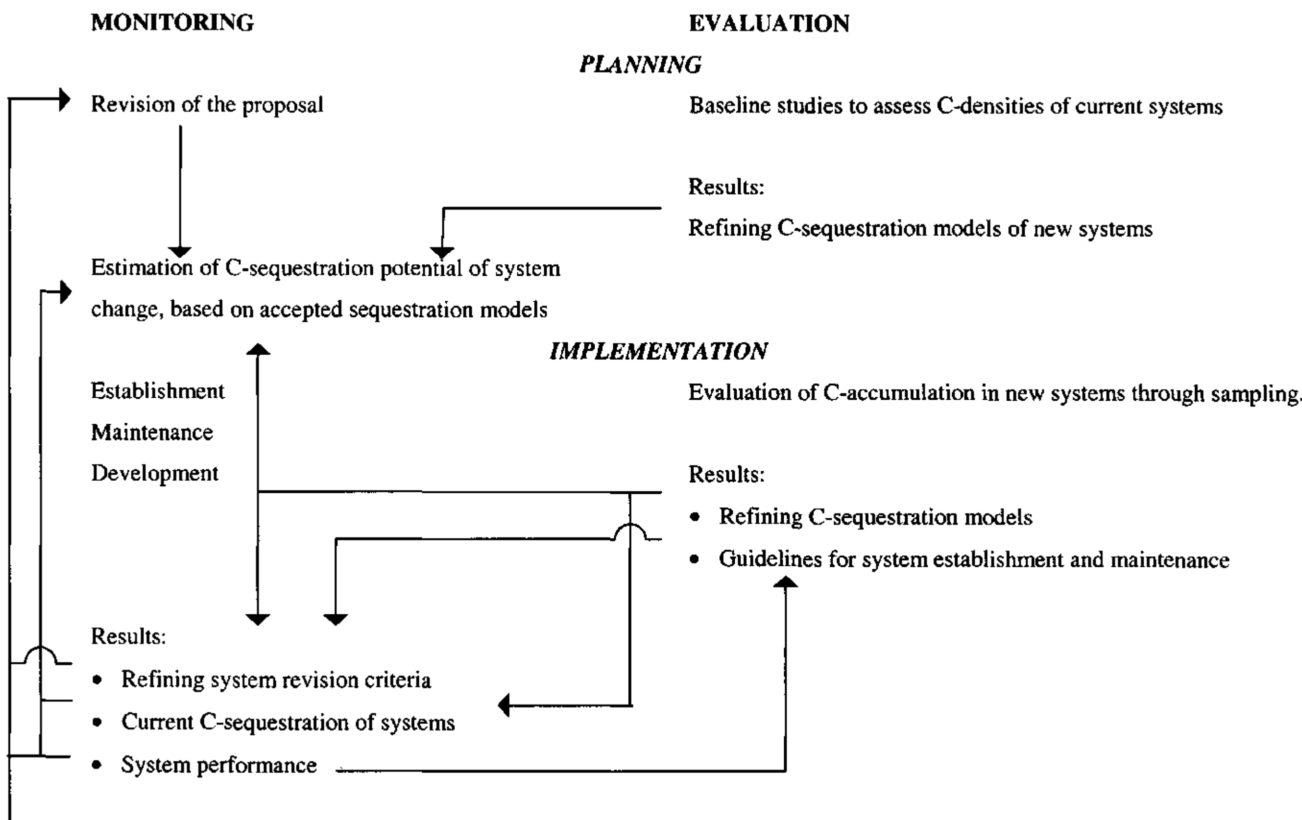


Figure 7.4. *Information flows between monitoring and evaluation results for carbon sequestration estimation.*

Table 7.3. *Sustainability criteria and indicators to be used in the farm forestry C-sequestration pilot project.*

Criteria For Sustainability	Objectively Verifiable Indicators
1. Farm forestry is established on land where ownership and usufruct rights are clearly established	1.1 Areas included within the scheme are marked clearly on maps. 1.2 Owners approve forestry plans and appropriate official tenurial documentation is available.
2. Farm forestry systems suit local needs	2.1 The scheme is voluntary. 2.2 The planning process is based upon local people's management objectives: The methods used to ensure local input into design of forestry are described and records are kept documenting the process.
3. Farm forestry design and implementation is sensitive to gender differentiation	3.1 Women participate in the design of farm forestry systems: The methods used to ensure adequate consultation are documented and progress is recorded. 3.2 The impact of forestry system on women is assessed: Details of the method of assessment are documented.
4. The farm forestry system is economically viable once support from the project is withdrawn	4.1 Economic productivity compares favorably with other economic land uses: Income per ha and per work-day should be greater than for local maize-bean system. The method used to compare farm forestry with other economic activities is documented.
5. Farm forestry systems will not disrupt important ecological processes	5.1 Soil structure and organic matter content should be maintained at acceptable levels, or improved: To be monitored by farmers. 5.2 Biological diversity at farm, village or municipal levels should not be significantly reduced. 5.3 Intra-species genetic diversity of trees used for forestry

	will be maintained: The use of native species will be encouraged; seed stocks of native species will be conserved; conservation measures will be documented. 5.4 Watercourses should not be polluted or damaged and watershed integrity should be maintained.
6. Skills and expertise required to manage farm forestry will be available locally after termination of the project	6.1 Farmers participating in the scheme will receive adequate training in farm forestry management: Details of training courses and implementation of training are documented. 6.2 Affordable specialist forestry expertise is available to farmers beyond the course of the project.

Participatory

Given the small scale, high diversity and dispersion of forestry plots in farm forestry projects, farmers themselves must be the prime source of data on growth and development of stands (measurement of stand data by technical staff would be too expensive). For this to occur successfully substantial effort must be made both to train farmers how make the required measurements and to engender positive relationships between technical staff and farmers, so as to encourage accurate record keeping. As with all well managed projects, the aim of monitoring should be to identify and address problems before they become serious; and this applies to organization and social aspects as well as technical considerations. In the case of the Pilot Project, community representatives help to liaise between technical staff and farmers. These representatives also participate on the board of the Trust Fund responsible for funding the local sequestration program. They are therefore well placed to view the system from the perspective of the offset purchaser as well as from that of the service providers (farmers).

Simple

Both the methods of data collection and recording procedures should be as simple as possible. Farmers should understand the methods they are using to collect data, and they should also understand how this data would be used. Monitoring systems have a tendency to be "greedy" for data. However, it is important to critically review the requirements of monitoring so that redundancy and irrelevancy is avoided. For example, in small-scale projects it may not be necessary to evaluate the effect of activities on biodiversity. In the Pilot Project, the Plan Vivo format is designed to be simple and relevant to farmers objectives for production improvement: most of the information required to monitor the progress of plots can be easily recorded on successive translucent paper overlays.

Reflective

As with other areas of the management system, the procedures of monitoring and evaluation should be constantly subject to improvement and refinement. The key to improvement is to reflect upon the main sources of error within the system. In the case of estimates of the carbon density of different land-uses in various ecological conditions used by the Pilot Project; these are currently based partly on direct biomass measurements supplemented by the best available data in the literature. However, the C-flux models will be adjusted periodically, as new data is gathered, showing C-densities of the pools that are likely to change rapidly and substantially, such as phytomass, necromass, and soil carbon. Figure 7.4 illustrates the information flow between monitoring and evaluation results for the C-flux estimation. Another area likely to require on-going improvement is the compliance of farmers with monitoring schedules. This will be subject to gradual improvement through modification of the Plan Vivo methodology, training of farmers, and linkage of incentive payments to fulfillment of reporting requirements.

Acknowledgments

This work is supported by the UK Overseas Development Administration's Forestry Research Programme: Community Forestry Development and Carbon Sequestration in Chiapas, Mexico: project No. R6320. The feasibility study was financed by the Mexican National Institute of Ecology of the Secretary of Environment, Natural Resources, and Fisheries and the European Community (Ref B-73014/92/412/9).

References

Brown, S., and Lugo, A.E., 1982. The storage and production of organic matter in tropical forests and their role in the global carbon cycle. Biotropica 14: 161-178.

Brown, S., Gillespie, A.J.R., and Lugo, A.E. 1989. Biomass estimation for Tropical Forests with Applications to Forest Inventory Data, Forest Science 35(4): 881-902.

Casley, D.J., and Kumar, K. 1987. Project Monitoring and Evaluation in Agriculture. The John Hopkins University Press, Baltimore MD, USA.

De Jong, B.H.J., and Montoya-Gómez, G. 1994. Sustainable Management of Forest Resources: A Proposal for the Highlands of Chiapas, Mexico. Proceedings of the 1994 Symposium on Systems Analysis in Forest Resources: Management Systems for a Global Economy with Global Resources Concerns. Pacific Grove, California. pp 7-17.

De Jong, B.H.J., Montoya-Gómez, G., Nelson, K., Soto-Pinto, L., Taylor, J. and Tipper, R. 1995. Community forest management and carbon sequestration: A feasibility study from Chiapas, Mexico. Interciencia 20(6): 409-416.

De Jong, B.H.J., Soto-Pinto, L., Montoya-Gómez, G., Nelson, K., Taylor, J., and Tipper, R. 1996. Forestry and agroforestry alternatives for carbon sequestration: an analysis from Chiapas, Mexico. Draft Proceedings of the Workshop on "Instruments for Global Warming Mitigation: The Role of Agriculture and Forestry", Trento, Italy, 22 - 25 May, 1996. pp 147-159.

Dixon, R.K., Schroeder, P.E. and Winjum, J.K. (Eds.), 1991. Assessment of Promising Forest Management Practices and Technologies for Enhancing the Conservation and Sequestration of Atmospheric Carbon and Their Costs at the Site Level. EPA/600/3-91/067, US-EPA, Washington D.C.

Dixon, R.K., Andrasko, K.J., Sussman, F.G., Lavinson, M.A., Trexler, M.C., and Vinson, T.S. 1993. Forest Sector Carbon Offset Projects: Near-term Opportunities to Mitigate Greenhouse Gas Emissions. Water, Air and Soil Pollution 70: 561-577.

Faeth, P., Cort, C. and Livernash, R., 1994. Evaluating the Carbon Sequestration Benefits of Forstry projects in Developing Countries. World Resources Institute.

Fearnside, F.M. and Malheiros-Guimaraes, W. 1996. Carbon uptake by secondary forests in Brazilian Amazonia. Forest Ecology and Management 80: 35-46.

Foley, G. and Barnard, G. 1984. Farm and Community Forestry. Earthscan, Techn. Rep. 3, IIED, London.

Heissenbuttel, J., Fox, C., Gray, G., and Larsen, G. 1992. Principles for Sustainable Management of Global Forests. UNCED, June 14, 1992, in Rio de Janeiro, Brazil. The Global Forestry Coordination. 33 pp.

FSC, 1996. Forest Stewarship Council Principals and Criteria for Natural Forest Management. Document 1-2. Oaxaca, Mexico. http://antequera.antequera.com/FSC/1-2.doc.

Masera, O.R., Bellon, M.R., and Segura, G. 1995. Forest Management Options for Sequestering Carbon in Mexico. Biomass and Bioenergy 8: 357-367.

Mitchell, R.B., and Chayes, A. 1995. Improving Compliance with the Climate Change Treaty. in: Lee, H. (Ed.), Shaping national responses to climate change: a post-Rio guide. Island Press, Washington D.C., pp. 115-146.

Nabuurs, G., and Mohren, G.M.J. 1993. Carbon Fixation through forestation activities, Institute for Forestry and Nature Research, IBN Report 93/4, Wageningen, The Netherlands.

Schlamadinger, B. and Marland, G. 1996. The role of forest and bioenergy strategies in the global carbon cycle. Biomass and Bioenergy 10: 275-300.

Schroeder, P.E., Dixon, R.K. and Winjum, J.K., 1993. Forest management and agroforestry to sequester and conserve atmospheric carbon dioxide. Unasylva 173(44): 52-60.

Swisher J.N., 1992. Cost and Performance of CO_2 Storage Projects in Forestry Projects. Biomass and Bioenergy. 1 (6): 317-328.

Trexler, M.C., and Haugen, C. 1995. Keeping it green: Tropical Forestry Opportunities for Mitigating Climate Change. WRI, EPA.

8. COMPLEMENTARY ISSUES AND CONCLUSIONS

Forestry and agroforestry can mitigate net greenhouse gas emissions by maintaining or increasing existing stocks of C, creating new stocks, and replacing energy-demanding materials (cement, fossil fuel) by renewable natural resources (Schlamadinger and Marland, 1996). Slowing deforestation and assisting regeneration, afforestation and agroforestry constitute the primary mitigation measures for carbon conservation and sequestration. Among these, slowing deforestation and assisting regeneration in the tropics (22 to 50 x 10^6 GgC) and forestation and agroforestry both in the tropical (23 x 10^6 GgC) and in the temperate zones (13 x 10^6 GgC) hold the most technical potential of conserving and sequestering carbon (Dixon et al., 1994a).

Governments in a few tropical countries, such as Brazil and India, have instituted measures to halt deforestation. For these to succeed over the long term, the enforcement has to be accompanied by the provision of economic and/or other benefits to deforesters that equal or exceed their current profits. National tree planting and reforestation programs, with varying success rates, exist in many tropical countries, for instance Indonesia. Here also, adequate provision of benefits to farmers and/or forest concessionaires will be important to ensure their sustainability.

Ongoing jointly implemented projects in Mexico, such as the Scolel Té project, combine various types of mitigation options discussed above. The lessons learned from this project and others will serve as important precursors for future mitigation projects. However, without their widespread emulation and replication, the impact of these projects by themselves on carbon conservation and sequestration is small. In the next sections, a number of supplementary issues are presented and discussed within the context of the experiences obtained to date with the International Pilot Carbon Sequestration Project Scolel Té, Chiapas. A brief outline of the most promising forest-based land management options are presented, based on the systems that farmers participating in the Scolel Té project seem to prefer. A simplified modelling approach has been developed that could deal with some of the complexities involved in estimating long-term C dynamics in the set of highly varying systems and ecological conditions, such as those in Chiapas. At the regional level, the spatial and temporal variation in LU/LC dynamics, observed in the last two decades, is placed in the context of regional reference land-use scenarios. At a local level, the spatial variation in the traditionally managed forest fragments in the Highlands of Chiapas was evident and this places methodological constraints on the calculation of the impact of management alternatives on total biomass dynamics. The Scolel Té project, in which multiple farmers participate with various small-scale systems in a variety of ecological conditions, has been a highly interesting learning experience to develop planning, implementation, and monitoring expertise necessary to design variants of this type of projects. These must be adapted to other regions; following what Central European foresters call since centuries "the iron law of the site" (*das eiserne Gesetz des Standortes*).

Forestry-based land management options for carbon mitigation

Carbon mitigation by means of forestry is a function of the amount of above- and below-ground biomass (including soil carbon) in a given land unit and given time horizon. Any activity or management practice that increases the amount of biomass in an area and/or in products over time is a potential carbon mitigation activity. Plantations, agroforestry and sustainable forest management are among the most attractive alternatives, as they can cover both global climate change concerns and local economic interests, providing the landowner with additional income (Chapter 4). Alternative land-use systems, among which the farmers of the Scolel Té project are willing to choose, were all based on their current land-use practices in which they apply locally tested or well-documented improvements (Figure 7.3). Potential options to mitigate greenhouse gas emissions that were discussed with the farmers can be broadly grouped into three categories:

1) to conserve existing pools in old-growth forest fragments;

2) to enhance pools in existing degraded or open forests;

3) to develop new pools by means of afforestation and agroforestry.

Conserving existing pools in forests

Deforestation is a response to underlying pressures, such as population growth and socio-economic development (Chapter 3). Slowing the rate of loss of existing forests reduces CO_2 emissions and has other benefits, such as maintenance of the reproduction of biodiversity in the tropics (Remmers, 1998; Rossignol et al., 1998).

Communities in the northern part of Chiapas have demonstrated interest in negotiating the long-term conservation of cloud forest fragments within the scope of the Scolel Té project and preliminary estimations of the expected C offset benefits were calculated (Jimenez et al., 1999). *Plan-vivos* (Chapter 7) have been presented to the local trust fund that manages the Scolel Té project and the implementations of the plans now await funding.

Enhancing pools in existing forests
Two management approaches can increase carbon pools in existing forests:

- management for carbon storage;
- management for carbon substitution.

Storage management
Storage management increases the amount of carbon in vegetation, soil, and durable wood products, integrated over time. This can be accomplished by various measures such as enabling secondary and degraded forests to grow up into closed mature forests by natural or artificial regeneration; reduced-impact logging practices (Van der Hout, 1999); improved silvicultural measures; and agroforestry. The resource-poor farmers of Chiapas especially consider agroforestry as an important option, since they can fit these alternative systems within their existing land-use management practices. Currently, almost all of the activities financed by the Scolel Té project fall within the category of agroforestry, including enriched shading of coffee, enriched fallow management, taungya, restoration of degraded grazed woodlands, and living fences. These agroforestry measures are highly favored by the farmers because agroforestry does not compete with their other productive activities and can be developed by each farmer individually on his own land and in his own style (Chapter 4).

Communities in the southern part of the Highlands are discussing sustainable community-based forest management plans that could follow the general pattern of the Scolel Té program quite well (e.g. Juznajab la Laguna, Chapter 6). A wide variety of forest management activities are proposed that could enhance the carbon storage per unit area, including standing stock management, fire control, soil improvements, reduced-impact logging, etc.

Substitution management
Substitution management views forests as renewable resources, yielding forest products, in particular to replace fossil-fuel demanding products (Hall et al., 1991, Marland and Marland, 1992; Marland and Schlamadinger, 1997). This alternative is difficult to conceptualize with farmers. The establishment of baseline versus project, or in other words the proof of additionality, is difficult to explain within the socio-economic context of the farmers' communities of the Highlands of Chiapas. For example, most families still depend on fuelwood as the main energy source. Electricity is scarcely available and if available, its consumption is restricted to provide light. Most building materials already come from wood resources, except in some modernizing communities, where cement is currently being introduced.

Afforestation for carbon storage

Most analyses of the potential for additional afforestation have been undertaken on a global basis (e.g. Dixon et al., 1994b). In particular, these studies have emphasized the tropics because of the high biomass growth rates, relatively cheap labor and land costs and the fact that these areas have been recently deforested. However, this option has to compete with other land-use options that are currently economically more attractive from the point of view of the farmers. This means that relatively high opportunity costs will have to be considered, which makes this option macro-economically less attractive (Chapter 5). None of the farmers or communities that are collaborating with the Scolel Té project has spontaneously submitted a proposal for afforestation of open areas.

An abstracted parametric approach to simulate C fluxes in complex forest ecosystems

In order to predict biomass dynamics in a forest, researchers have increasingly used various types of simulation models. Modelling of biomass fluxes in managed, natural forest can increase the usefulness of the existing forest-biomass information by:

(i) providing a systematic structure for summarizing existing data,

(ii) identifying information gaps,

(iii) increasing effectiveness of further data collection,

(iv) formulating and numerically checking theories about biomass accumulation, and

(v) aiding to forecast biomass yield for specific site and species conditions (Arp and McGrath, 1987; Oldeman, 1992).

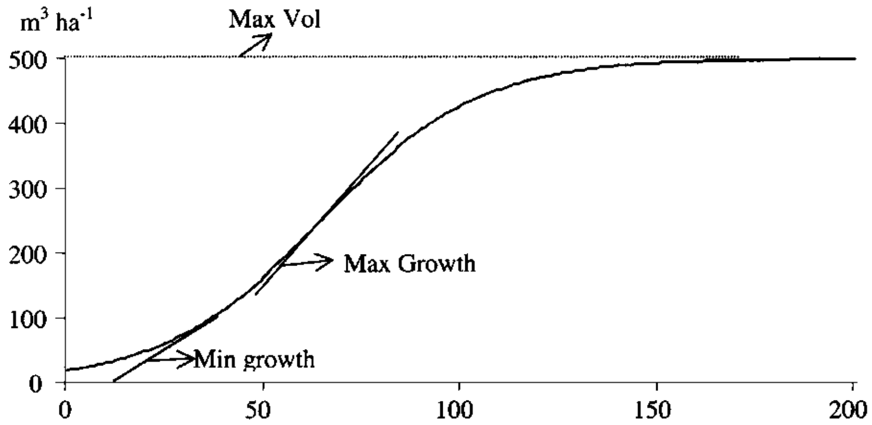
Although there is no shortage of data and models to assess stand biomass, there remains the task of examining the dynamics of above- and below-ground biomass by way of an abstracting approach. To be of general use, such an approach should use numerically expressed parameters, such as stand compartments (leaf biomass, above-ground woody biomass, roots, litter, dead wood, and soil organic matter), transition flows, rates of production, and decay processes of biomass in relation to the particular conditions determined by site and species composition. In turn, the assessment of these conditions should allow to produce efficient and realistic representations and simulations of whole-stand biomass dynamics (plant and soil) as a function of stand dynamics, silvicultural interventions, and other factors that may influence stand development.

Any model that tries to explain the dynamics of a system indeed is an abstraction of the reality. The level of abstraction used in a model depends on the purpose of the model, the information available about the functioning of the system, and the data available to test the model as well as the nature and capacity of the computer software used. Models that simulate dynamics within stands can be based on individual trees, tree guilds, more or less uniform stands, or combinations of these, all with or without information related to the three spatial dimensions. Spatial and temporal variation is strong in mixed-species forests in tropical areas, as compared to most temperate forests. This variation is reflected in a high diversity of species, complex horizontal and vertical architecture, with the horizontal distribution of trees closely linked to the behavior of each of the species and the vertical distribution to crown interaction in light interception (Vester, 1997). These factors combined result in highly complex regeneration dynamics and patterns of growth and mortality. Mixed-species stands have also more variation between individuals in shade tolerance (Vester, 1997; López-González et al., 2000) and height-growth patterns (See section on fragment variation). Defining the ecological parameters of individual trees within this complex setting is a serious problem. Architectural variation is statistically expressed by more deviation and variation from general development pathways as given by regression curves, because the type and frequency of statistical disturbance are the more stochastic, the more the spatial and temporal architectural patterns of species interactions are complex (Oldeman, 1992; Larson, 1992). Different kinds of species variables and parameters have been used in the literature, such as: species-site qualities; species response functions; tree temperaments; growth responses; case-specific responses; growth multipliers (e.g. Shugart, 1984; Oldeman and Van Dijk, 1991; Kellomäki and Kolström, 1992; Sykes and Prentice, 1996; Vester, 1997). The amount of information necessary to validate numerical models that explain the behavior of individual trees in complex multiple-species systems, which are growing under high variety of situations, has been one of the major constraints in the development of general models (Oldeman, 1990; Vanclay, 1994). The question then arises as to whether we could find a way of describing certain abstract stand dynamics, a way that takes into account the numerous situations that may occur in a typical tropical forest fragment.

The carbon cycle in a managed and unmanaged forest is only part of a more complex system where unexplained variations and uncertainties in mass dynamics, succession patterns, and internal and external processes are occurring simultaneously. A general modelling approach that has been used widely to explain nutrient dynamics, such as the carbon cycle, is to subdivide the forest ecosystem in more or less homogeneous compartments (Oldeman, 1990). This permits to simulate mass flows between the system compartments inside the system, and between those and the world outside the system. The number of compartments in the system may vary. It can range from a very simple system representation with aboveground biomass, litter, and soil organic matter as the compartments and flows between these compartments and the atmosphere and ecosphere (Figure 7.2; Grace, 1997); to several hundreds or thousands of components, compartments and fluxes, as in individual tree-based models (e.g. Symfor, 2000).

In this study, the purpose was to analyze C flows in forests on varying sites, with varying architecture and species composition, subjected to different management strategies. Although the forests in the study area demonstrated a high variability in architecture and species composition within one forest, some general patterns could be identified at the stand level (Figure 6.9; Table 6.3). A stand-based modelling approach therefore was chosen as the most appropriate starting point (Vanclay, 1994); a stand model in which an uncertainty analysis could be applied to the various input parameters to test the impact of varying their value on the C flow estimations (Chapter 6). Procedures were designed and applied that allowed for the division of a diversified forest fragment into more or less homogeneous stands, which were assumed to have similar biomass dynamics. Stand-based variation of the biomass compartments, measured by means of forest inventories, was used in the uncertainty analysis of each stand simulation.

The approach included the simulation of internal biomass dynamics by way of simple differential equations; with live carbon pool increase estimation driven by stem wood increment, with linear ratio allocations to leaves, branches, and roots. As usual, the stem wood accumulation in each stand was assumed to follow a logistic curve (Figure 8.1), the slope of which was determined by measured or estimated growth potential of species assemblages and the asymptotic maximum of the curve by maximum wood volume measured in old-growth stands. The volume increment of each year was determined by the standing volume and growth potential of the species assemblages present in that year. This approach assumed that internal stand factors remained similar or at least did not notably affect the growth potential of the stand components.



$dV / dt = a.V - b.V^2$

$a = \text{Max Growth} * 4 / \text{Max Vol}$

$b = a / \text{Max Vol}$

V = current stemwood volume

Figure 8.1. *Growth function used to calculate the increment assignment to species assemblages, according to the current volume (V), the maximum volume (Max Vol) that can be sustained at the site, and maximum growth (Max Growth) potential of the species assemblage. Minimum growth can be included as a linear function as an additional abstraction of growth when very low or zero standing tree biomass is present.*

The dynamics of the rest of the system were based on estimated or published transfer coefficients of mass between the various compartments. Knowing or estimating these coefficients and incorporating variations in the value of various parameters at once in the uncertainty analysis, permitted an analytical simulation of biomass fluxes with statistically robust confidence intervals for different site and species combinations (Chapter 6).

The model was developed as a first approximation that allowed a systematic examination of biomass dynamics at a stand level, in which the initial state, growth, resource allocation, and management strategies could vary. The model was developed as an alternative for the plantation-oriented CO_2FIX model (Mohren and Klein-Goldewijn, 1990; Mohren et al., 1999). A yearly wood increment curve and cyclic harvesting of all wood drive this latter model. It simulates plantation-type stands well, but is not very appropriate for uneven-aged multiple species forests, where trees can be harvested selectively. The conceptual approach that was used turned out to be very similar to the stand-based model, developed by Arp and McGrath (1987), so their modelling syntax was used to describe the general outline. The following compartments, flows, and compartment increments were incorporated in the model:

Table 8.1. *Biomass compartments distinguished in the model.*

Living biomass pools	Dead biomass pools
Foliage (F), including flowers and fruits	Dead wood, including stem and branches (DW)
Stem (S)	Leaf litter (LL)
Branch (B)	Dead root (DR)
Root (R)	Soil Organic Matter (SOM)
	Product biomass (P), subdivided in short-, medium-, and long-lived products

Compartments

The model only simulated the vegetative dynamics of biomass (Table 8.1). Duvigneaud and Denaeyer-De Smet (1970) reported that the sum of all vegetative biomass pools accounts for about 99% of the total biomass of a forest, whereas the animal pool within the forest system comprised the remaining 1%. None of the models in use today includes a purely microbial compartment, so it is considered here to be part of the soil compartment. There also remains a doubt as to the assessment of the animal compartment. If insects were included, the pool might be larger than the one percent found in the literature consulted.

Flows

The average annual per-hectare flow from one compartment to another was presented as:

$$x_i \rightarrow y_j \qquad (1)$$

where x_i was the ith "donor" and y_j the jth "receptor" ($i, j = 1, 2, \ldots N$ with N as the number of compartments under examination and $i \neq j$). Each transfer was considered a function of the donor and receptor compartment, resulting from natural or human-induced events (Table 8.2).

Table 8.2. *Flows between the various compartments that result from natural or human-induced events (Actual Biomass Production = ABP; ω = CO_2 to the atmosphere).*

Natural event	Flow	Human-induced event	Flow
Biomass production	ABP → S	**Harvesting of products (in the model simplified to harvesting of stem products)**	S → P
	ABP → B		S → DW
	ABP → R		B → DW
	ABP → F		F → LL
			R → DR
Tree mortality	S → DW	Burning	LL → ω
	B → DW		DW → ω
	F → LL		S → DW
	R → DR		B → DW
			F → LL
			R → DR
Branch, leaf, and root turnover	B → DW		
	F → LL		
	R → DR		
Humification of dead wood, litter, and roots	DW → SOM		
	LL → SOM		
	DR → SOM		
Humus, dead wood, and litter decay	SOM → ω		
	DW → ω		
	LL → ω		

Model abstraction was applied to focus attention upon:

1) Only stands of up to three species assemblages;
2) Specific site-dependent transfer rates, each stand being examined for a specific site;
3) Annual rates of most processes considered to be linear and 'donor' controlled (Gardner and Mankin, 1981), so that equation (1) could be written as:

   $x_i \rightarrow y_j \approx k_{ij} \cdot x_i$

in which k_{ij} values were considered to be constant. As data become available, these values can be transformed easily to other functions). This assumption of constancy of transition rates implies that environmental or genetic variation in a stand had limited effects on the rates of the processes considered here (Arp and McGrath, 1987). Attention was thus limited to stands for which the overall environmental conditions and genetic variation might be assumed to remain the same during stand development. Genetic variation was acceptable, as long as the overall composition of the species assemblages had similar rates for biomass production and photosynthate allocation on a given site.

Environmental conditions can be subdivided into factors internal in and external to a stand. In particular, external 'site conditions' since a century were generally assumed to remain constant throughout stand development, although recent research found that forest productivity may increase under changing environmental conditions, particularly under increasing atmospheric CO_2 levels and temperature (e.g. Phillips et al., 1998), or development of organic soil horizons, e.g. in re-afforested dune sands in Europe since 1850.

Internal stand factors generally change throughout stand development, such as microclimatic conditions, nutrient availability, soil moisture, etcetera. This type of variation would change equation (2) in:

$$x_i \rightarrow y_j \approx k_{ij}(\mathit{X}).x_i \qquad (3)$$

in which (X) is a variation matrix; thus the k_{ij} –coefficient would become dependent on the state of the stand architecture, i.e. the state of some out of all the compartments considered. Such qualifications could readily be incorporated into the model, but would require explicit information about the particular functional relationship for each k_{ij} (X) and would complicate the time integration of the x_i -> y_j processes by changing the linear model into a non-linear model. They would therefore complicate the first-approximation requirement for generating an abstracted overview of the development of biomass by stand age, site, species, and general stand conditions following specific interventions (Arp and McGrath, 1987). In summary, the overall model formulation and parameterization was meant to be specific as to site and species composition, so that each stand could be represented by unique k_{ij} values. The evaluation of the value of these coefficients hence was consequently of primary importance. In Chapter 6 an uncertainty analysis was employed to assess the impact of varying parameter values on expected C flux outcome.

Increment per compartment

The annual increments for each compartment were related to the transition rates according to the Principle of mass conservation (see also Thornley, 1976). For example, the annual change of wood biomass per hectare was equal to the balance of wood production, wood mortality, and woody litter fall. The annual gain in foliar biomass was defined as the amount of foliar biomass produced in the current year minus foliage loss due to foliage turnover, tree mortality and harvesting. Overall, the change in compartments were driven by a set of first-order differential equations, denoted as:

$$X' = k \cdot X + C$$

in which **X′** is a vector showing the rates of compartment increments, **k** a matrix of transfer coefficients, **X** the compartment vector and C a ′forcing′ vector that accounted for all contributions to **X′** which are independent of **X**.

This system of equations simulated **X** for a given stand, a specific combination of site and species composition, and an initial compartment setting **X(0)**. Note that the solutions generated with the simplified model should lead to steady-state values for each compartment as $t \rightarrow \infty$, as long as the logistic growth assumption was maintained. Varying the maximum amount of volume in a stand would cause fluctuations around the steady state, which indeed reflect the reality of the forest as a whole (Oldeman 1990). Asymptotic approaches toward limiting values can often be observed for individual stand compartments with advancing age. Neither variations of asymptotic values, nor possible effects of CO_2 fertilization on the biomass sink capacity of a stand were taken into account at this stage (Phillips et al., 1998).

Concluding remarks

Emphasis in the above was placed on generating an abstracted parametric method for assessing whole-stand biomass dynamics via readily identifiable compartments and growth processes. First-order approximations were used to simulate the annual quantities of biomass produced, allocated, or lost. The approach was meant to be flexible and adjustable, so as to permit the incorporation of higher-order approximations as data sets become available. For example, the stand-based litter decomposition and humification factors or root, shoot and foliage allocation coefficients were at this stage assumed to be uniform, but could be replaced eventually by other linear or non-linear functions per species or per site. Also, some of the compartments could be subdivided into smaller compartments (roots into large, medium and fine roots; litter into fresh, fermented, and humified litter layers). New compartments can be introduced too, such as biomass allocation by species in stands with two or more species, or non-tree biomass pools (e.g. agroforestry systems, soil life). Evidently, such additions would increase the amount of information required to calibrate the resulting model. In this respect, the pools used in the model may already be considered too precise, and further abstraction may still be needed. For example, the compartment for organic matter in the mineral soil could be deleted, as this compartment was found to be statistically similar in LU/LC classes of the Highlands and Selva Lacandona (Table 3.2; De Jong et al., 2000). Measuring differences in time that would be required to calibrate the model could be too complicated or too expensive. One may therefore wish to consider an abstract system consisting of two compartments only: living- and dead aboveground biomass. A dynamic description of these two compartments alone would probably assess the most important short-term changes that occur during stand development and thus would generate a useful first-order estimation of the most important annual biomass gains and losses per hectare on a stand-by-stand basis. These are also the compartments for which extensive databases are available.

Temporal and spatial variation of deforestation and forest degradation in the Highlands of Chiapas

The most important human activities that transformed the landscape of the Highlands of Chiapas from 1974 to 1996 were: (i) temporal or permanent conversion of forest to cultivated land, (ii) selective extraction of pine trees for local timber production, and of oak trees for fuelwood and charcoal, and (iii) grazing of sheep and cattle in the forests and on abandoned agricultural land (Chapter 3; González-Espinosa et al., 1995; De Jong and Ruíz-Díaz, 1997). These land-use activities created a highly complex landscape, in which small fragments of cultivated land are mixed with fragments of secondary shrub- and tree vegetation, temporary and permanent grazing lands, and degraded forests (Ochoa-Gaona and González-Espinosa, 2000). The effect of these human activities on forests cannot be labeled at this scale as deforestation, because deforestation is defined as the process of converting forest to permanent open land (Hamilton, 1991). Natural and anthropic impacts affect the forest architecture and species composition, without removing the forest cover permanently. Changing forest architecture, natural processes, and/or species composition deliberately can lead to forest degradation *sensu stricto*. Land uses that can activate forest degradation include extensive and intensive extraction of wood for timber and fuelwood, intensive grazing, and slash-and-burn agriculture. This would be considered as deforestation only, if any of these activities lead to a permanent or long-term removal of more than 90% of the forest cover (FAO, 1996), e.g. if traditional slash-and-burn agriculture with long fallow periods shifts to permanent or short-fallow agriculture.

In Chapter 5 it was assumed that the historical LU/LC change dynamics in the Highlands could be used as the baseline assumption to predict future expected emissions of C due to continued LU/LC changes. In the following paragraphs I will discuss the impact on estimated C-fluxes of the spatial and temporal variation in LU/LC dynamics within the region, specifically for the area of six municipalities that corresponded to the Highlands (>1500 m altitud).

Tracking the change of LU/LC of each fragment in the Highlands of Chiapas revealed that of the 213,000 ha of closed forest fragments that were present in 1974, 79% were eventually converted to one of the other classes in 1996, 46% to secondary vegetation (shrub and tree fallow), 24% to degraded forest (disturbed pine-oak and open pine forest), and 9% to open areas (Agricultural land, pasture and settlements). Most of the fragments that were already open area in 1974 remained as such during the known period of 22 years. Secondary shrub and tree fallow fragments were increasingly converted to open areas, whereas only between 1974 and 1984 part of the tree fallow fragments were allowed to develop as closed forests (Figure 8.2). Thus, during the last two decades three major processes could be distinguished: (i) closed forests were increasingly transformed to degraded or secondary forest, (ii) tree and shrub fallow were converted to more permanent open areas, and (iii) open areas did not change. As such, closed forests were either incorporated into the slash-and-burn system or degraded due to extraction of forest products and/or grazing of cattle. Traditional slash-and-burn systems with long fallow and short production periods were altered to permanent or short-fallow agriculture. The result of these complex LU/LC change dynamics was that the landscape that in 1974 was dominated by closed forest with scattered patches of converted LU/LC types (secondary vegetation, degraded forest and open area), changed to a landscape of only small scattered patches of old-growth forest set in a matrix of disturbed and open forests, secondary vegetation, and open areas in 1996.

However, these processes were not homogeneous during those 20 years. Between 1974 and 1990 only a relatively small proportion of the closed forest changed to other LU/LC types, but this change accelerated substantially between 1990 and 1996. In contrast, the conversion of shrub and tree fallow to open areas took place mostly between 1974 and 1984 (Figure 8.2).

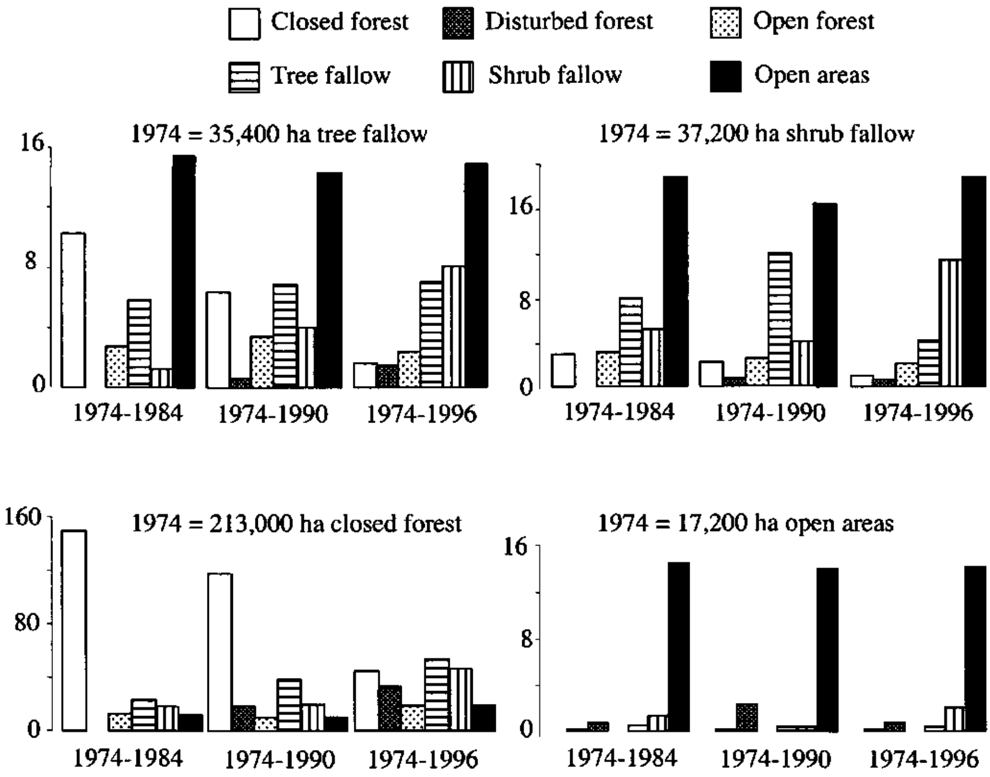


Figure 8.2. *Plotwise conversion of shrub fallow, tree fallow, closed forest and open areas from 1974 to 1996 (minimum plot size 3 ha). Data derived from overlays of maps based on remote sensing of 1974, 1984, 1990, and 1996.*

Spatial differences in land cover were already apparent in 1974. Analyzing the information for that year at the level of municipalities, the fragments of converted LU/LC types in Chanal were generally small and rather uniformly distributed throughout the municipality, whereas these were more concentrated in and around extensive valleys in the municipalities Huixtán, Teopisca, Amatenango, Comitán and Las Margaritas (Figure 8.3). Particularly Huixtán showed already large areas of converted LU/LC types in 1974, compared to the other five municipalities.

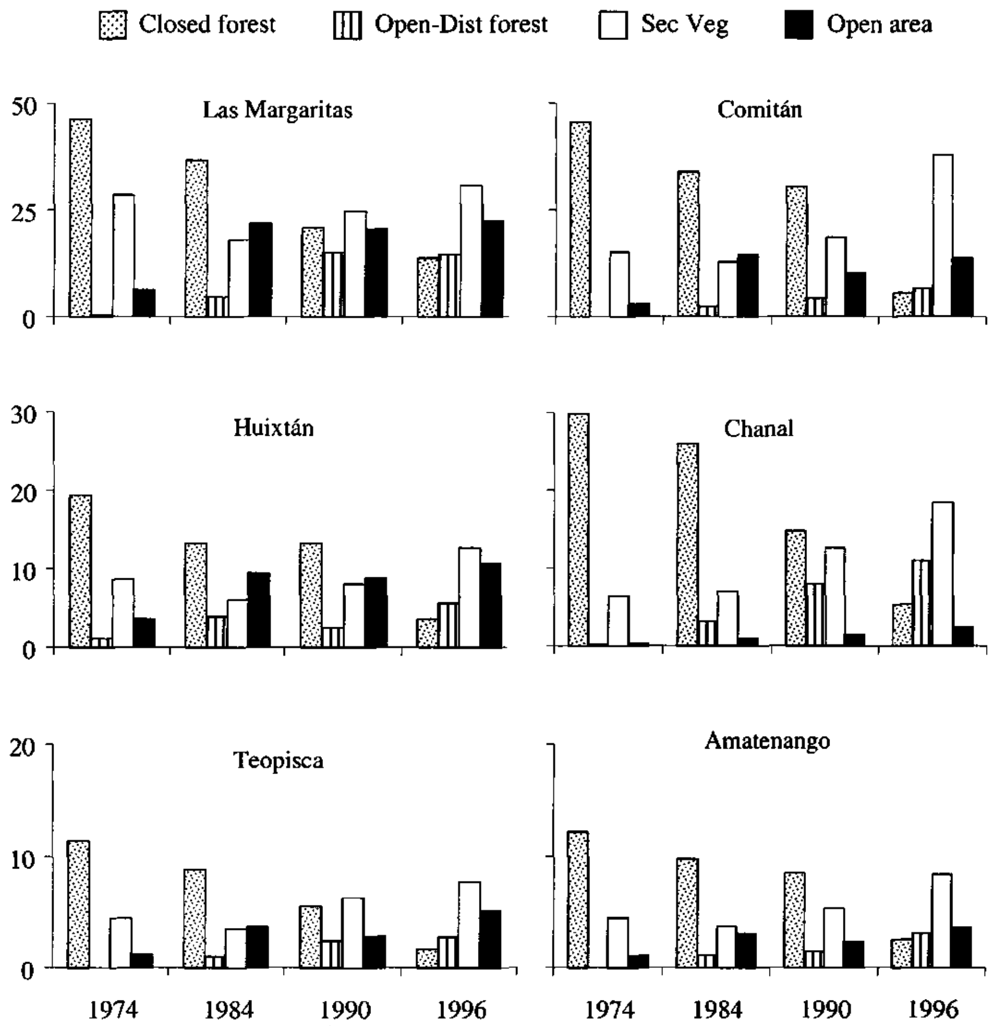


Figure 8.4. *Land use and land cover (in 1000 ha) in six municipalities of the Highlands of Chiapas in 1974, 1984, 1990, and 1996. Closed forest includes oak forest, pine-oak forest and pine forest; Open-Dist forest includes open pine forest and disturbed pine-oak forest. Sec Veg includes tree and shrub fallow. Open Area includes agriculture, pasture, settlements and other classes.*

Analyzing LU/LC change dynamics in space and time, variations could be observed between the six municipalities (Figure 8.4; Table 8.3). Driving forces that may have triggered the process of LU/LC change indeed varied among the municipalities, for example cattle grazing and forest exploitation most likely activated LU/LC change in Comitán and Las Margaritas, agriculture and cattle grazing in Huixtán, forestry in Chanal, forestry and agriculture in

Teopisca, whereas agriculture and commercial pottery were considered the most important factors in Amatenango (De Jong and Montoya, 1994; Ochoa-Gaona, 2000; Ochoa-Gaona and González-Espinosa, 2000).

Table 8.3. *Trends in LU/LC change derived from Figure 8.4 and main driving forces that most likely triggered these processes in six municipalities of the Highlands of Chiapas, Mexico.*

Municipality	Observed changes	Main driving forces
Las Margaritas	Steady decline in closed forest area with increase in degraded forest and open areas; secondary vegetation more or less constant.	Animal husbandry Forestry
Comitán	Between 1974 and 1984 a small shift from closed forest toward open area; from 1984 to 1996 a large shift from closed forests toward degraded forest and secondary vegetation.	Animal husbandry Forestry
Huixtán	Between 1974 and 1990 a gradual shift from closed forest to degraded forest and open areas; between 1990 and 1996 increasing area of secondary vegetation	Agriculture Animal husbandry
Chanal	Initially slow shift from closed forest toward degraded forest and secondary vegetation, but accelerating rapidly between 1984 and 1996	Forestry
Teopisca	Steady decrease of closed forest area with correlated increase in areas of all other classes	Forestry Agriculture
Amatenango	Increase in open area between 1974 and 1984 at the cost of closed forests; shift from closed forest toward degraded forest and secondary vegetation	Agriculture Commercial production of pottery

Impact of LU/LC change variability on C fluxes

Not all carbon in ecosystems is susceptible to disappear due to human impact, e.g. stable soil carbon will remain present a long time after clearing a forest. Assuming that the level of soil carbon in permanent agriculture or pasture land is the level of remanent C-density after disturbance, the amount of carbon released shortly after a LU/LC class changes to another class can be thus be estimated, by substracting the average soil C density of open areas from the total pools (Figure 8.5).

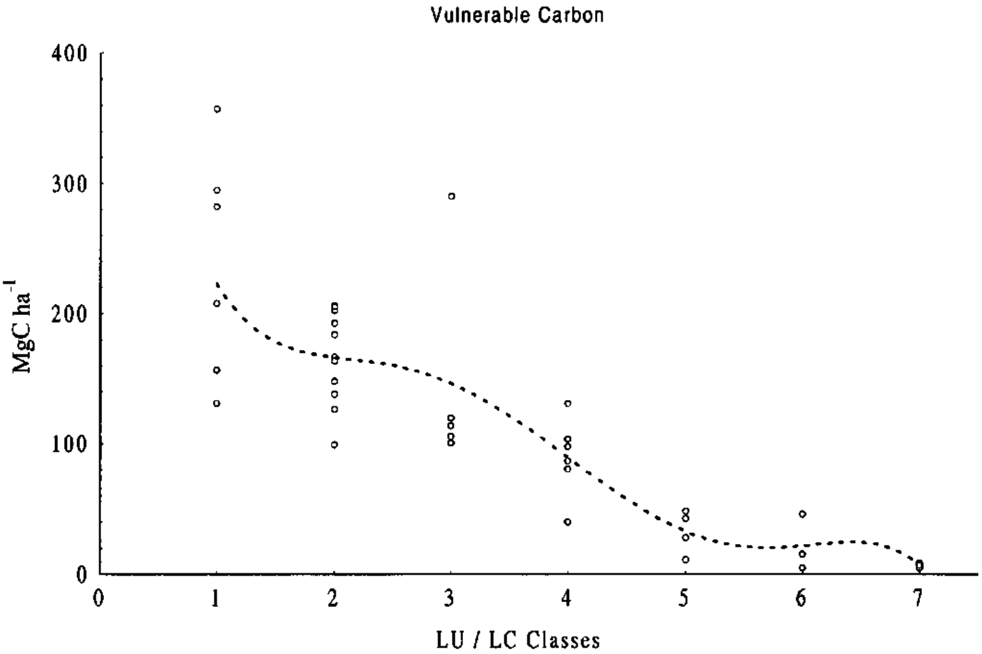


Figure 8.5. *Downward trend in vulnerable C densities in the LU/LC classes of the Highlands of Chiapas, going from Oak forest (1), Pine-Oak forest (2), Pine forest (3), Tree fallow (4), Shrub fallow (5), Pasture (6), to Agricultural land (7). N = 39.*

Assigning the vulnerable C densities to the LU/LC statistics of 1974, 1984, 1990, and 1996 would give an approximation of the amount of C that was removed or added in each municipality between each period. Thus calculated, in 1974 the vulnerable C-density per surface unit was highest in Chanal, followed by Comitán, Teopisca, Amatenango, and Las Margaritas, and lowest in Huixtán (Figure 8.6). The decrease in density between 1974 and 1996 was highest in Comitán and Teopisca, followed by Amatenango, Chanal and Las Margaritas, and lowest in Huixtán.

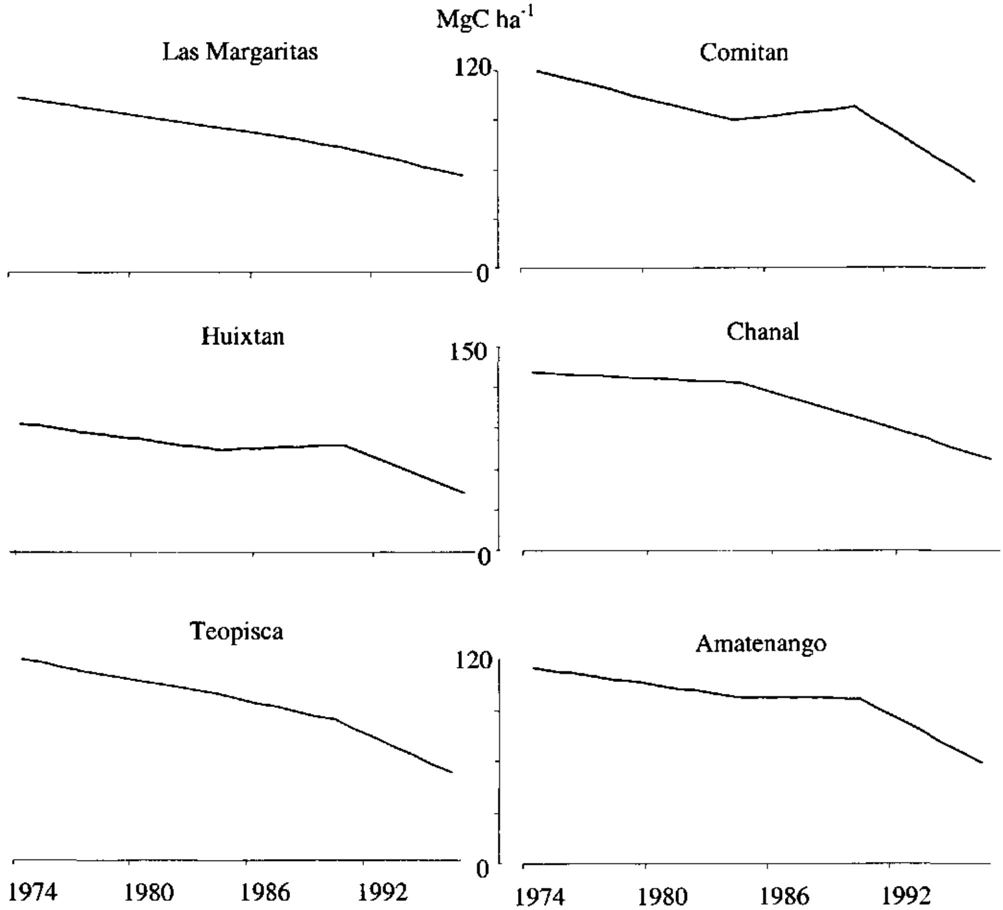


Figure 8.6. *Estimated trend in average vulnerable carbon density per land unit (MgC ha^{-1}) between 1974 and 1996 in the area above 1500 m altitud of six municipalities of the Highlands of Chiapas.*

On a yearly basis, the density decreased in Comitán and Teopisca by 3.5% yr^{-1} ha^{-1}, in Huixtán and Amatenango 3.1% yr^{-1} ha^{-1}, in Chanal 2.8% yr^{-1} ha^{-1}, and in Las Margaritas 2.7% yr^{-1} ha^{-1}. The trend also varied in time: in Las Margaritas and Teopisca the density ha^{-1} decreased uniformly downward during the 22 years, but fluctuated highly in Comitán, Huixtán and Amatenango, with high losses of biomass between 1974 and 1984, rather stable or somewhat increasing between 1984 and 1990 and declining steeply between 1990 and 1996. In Chanal the density decreased little between 1974 and 1984 but very fast during 1984 and 1996 (Figure 8.6).

Concluding remarks

This demonstrates that and how one of the problems to establish a baseline within a regional LU/LC sector relates to the prediction of future changes. This is particularly difficult, because the probable causes and driving forces behind land-use decisions are spatially and temporarily diverse, interrelated, and often discontinuous. At regional levels, factors such as demographic changes and government policies often have significant effects. At specific locations, uncertainties about land ownership, environmental characteristics, social conflicts, impact of development projects, crop failure and wildfires can cause unpredictable changes in land-use decisions among farmers (De Jong et al., 1998; Ochoa-Gaona, 2000).

The definition of a suitable baseline or reference case involves the elaboration of a hypothetical scenario. Differences in the order of a single percentage point in the assumed rate of loss of current carbon storage in the baseline assumption can halve or double the perceived net effect of a given intervention over the course of a 60 to 100 year time frame (Tipper et al., 1998). Methods used to define a baseline are either extrapolation of past rates of change of carbon stocks into the future (trend-based models; Chapter 5), or process-based models that attempt to simulate the demographic and/or economic processes driving land-use change (Brown et al., 1989; Faeth et al., 1994; Fearnside and Malheiros-Guimaraes, 1996; De Jong et al., 1998). A problem with trend-based predictions is the influence of the geographic domain and historic time frame used in the assessment, as shown earlier in this chapter. The more records show variation in the rate of LU/LC change over time and/or space, the less obvious is the most likely scenario in the future.

Another approach is to apply process-based models. Although these models may be capable of assessing the relative vulnerability of different categories of vegetation, they generally require large investments in data collection to make credible representations of land-use change processes. In densely populated areas where various factors influence local and regional land-use decisions, it is difficult to define which factor is the main driving force behind the LU/LC change dynamics. Rudel and Roper (1997) tested various factors that are thought to contribute to deforestation in the tropics for 68 tropical countries. The factors they tested were, among others, population pressure, economic growth, and national land-use policies. Based on their results, they distinguished two types of deforestation processes:

1. A frontier model, characterized by the opening up of new, still intact areas, and
2. An immiserization model (*sensu* Rudel and Roper, 1997), characterized by a continuous fragmentation and deforestation in densely populated areas, dominated by resource-poor farmers.

Both processes occur simultaneously in Chiapas. In the densely populated Highlands of Chiapas the immiserization process dominates (Ochoa-Gaona, 2000), whereas the frontier model prevails in the nearby tropical lowland forests of the Selva Lacandona (See De Jong et al., 2000). Under such conditions the same land-use policies may have positive effects in one of the regions, but negative impacts on the other. In fact, during the 1970s the Mexican government stimulated the frontier model in the Selva Lacandona region to reduce in part the immiserization process in the densely populated areas of the Highlands and Northern region of Chiapas. These kinds of regional or sub-regional positive and negative interactions reduce the reliability of a general process-based land-use model and make it very sensitive to leakages to and from adjacent regions. As such, it is most likely that process-based models that reliably predict land-use decision-making in the future will not work unless developed on a small scale first.

Project-level baseline assumptions also suffer problems of credibility and probability. For example, can forests not facing any threat claim C offset by means of forest conservation (Trexler 1993). And if forests are threatened, how to guarantee that the exploitation that threatens the forest is not simply displaced towards another area?

Also the forest architecture, site conditions and growth of trees within forest fragments can and do vary considerably, complicating baseline and project assumptions (Chapter 6), as will be highlighted in the next section.

Variability within a forest fragment

The data collected in the forest of Juznajab la Laguna showed a high spatial variability in tree biomass, species dominance, and tree species densities, due to different human activities in the forest, local differences in species composition, and topographic irregularities (Chapter 6; Escandón-Calderón et al., 1999; Konstant et al., 1999). Further analysis of the collected data also reveals that individual stands within the forest fragment function differently. Four types of stand-level variation could be distinguished:

1. Allometric relations of common species
2. Increment growth of common species

3. Stand-level species composition and stand architecture
4. Stand-level variation in volume increments

For example, estimated allometric relations between height and diameter varied highly in *Pinus oocarpa* Schiede (Figures 8.7 and 8.8) and *P. devoniana* Lindl (Figure 8.9), indicating a strong variation in site conditions. The estimated heights of *Pinus oocarpa* Schiede trees with a diameter of 50 cm varied between 14 and 25 m and of *P. devoniana* Lindl trees between 16 and 31 m among the various sites within the forest.

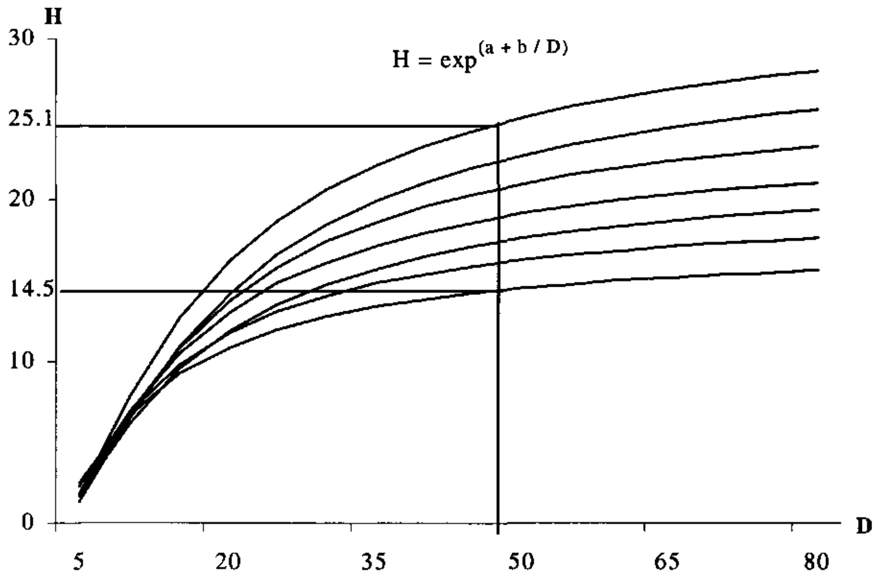


Figure 8.7. *Variations observed in the relation between height and $D_{1.30}$ in the forest of Juznajab la Laguna, Chiapas, México (with H in meters and D in cm at 1.30 m). N = 83.*

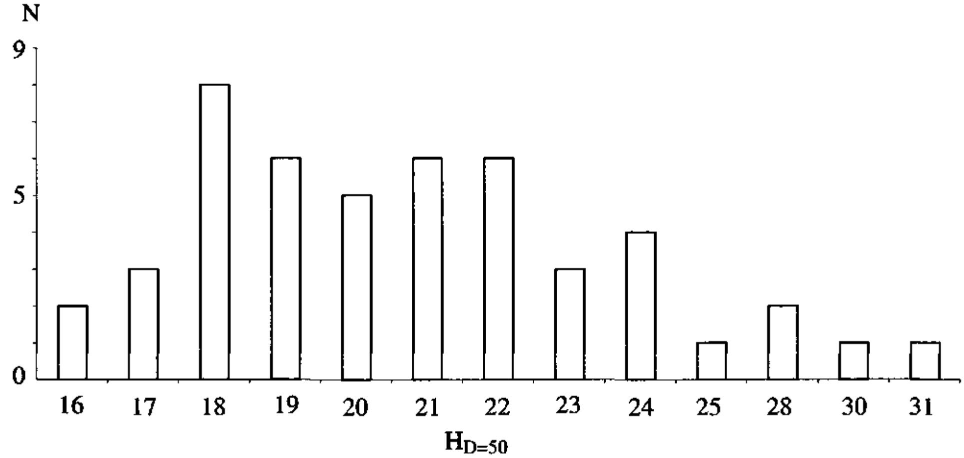


Figure 8.8. *Frequency distribution $H_{D=50}$ (H in m when $D_{1.30}$ = 50 cm) of Pinus oocarpa Schiede in Juznajab la Laguna, Chiapas, Mexico. N = 83.*

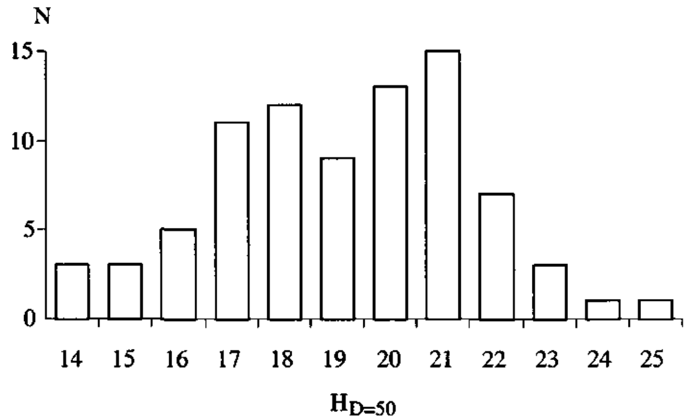


Figure 8.9. *Frequency distribution of estimated $H_{D=50}$ (H in m when $D_{1.30}$ = 50 cm) of* ***Pinus devoniana*** *Lindl in Juznajab la Laguna, Chiapas, Mexico. N = 48.*

Also the increment data of both species showed a high variability along the range of measured diameters ($D_{1.30}$), although in both cases a low but significant non-linear correlation could be observed (Figures 8.10 and 8.11).

Also stand architecture and species assemblage composition data presented a high variability (Figure 8.12).

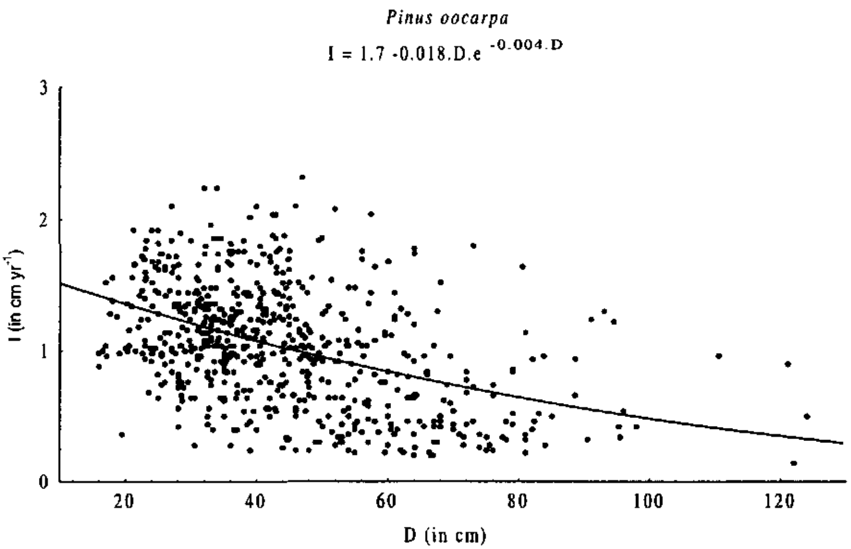


Figure 8.10. *Non-linear relation between diameter increment (I; cm yr^{-1}) and $D_{1.30}$ (in cm) for Pinus oocarpa Schiede in Juznajab la Laguna, Chiapas, Mexico. Minimum diameter is 15 cm. 19% of variation explained by the equation. N = 636.*

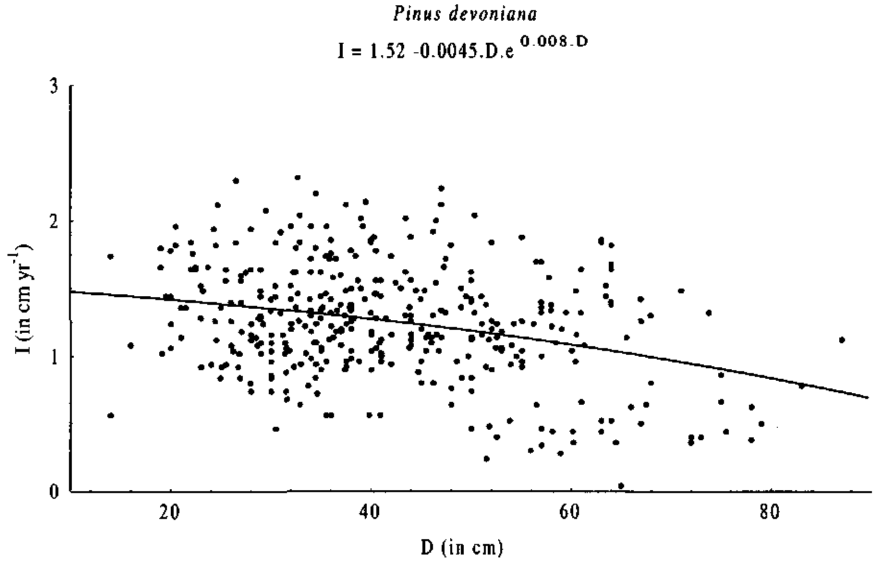


Figure 8.11. *Non-linear relation between diameter increment (I; cm yr^{-1}) and $D_{1.30}$ (in cm) for Pinus devoniana Lindl. in Juznajab la Laguna, Chiapas, Mexico. Minimum diameter is 15 cm. 10% of variation explained by the equation. N = 379.*

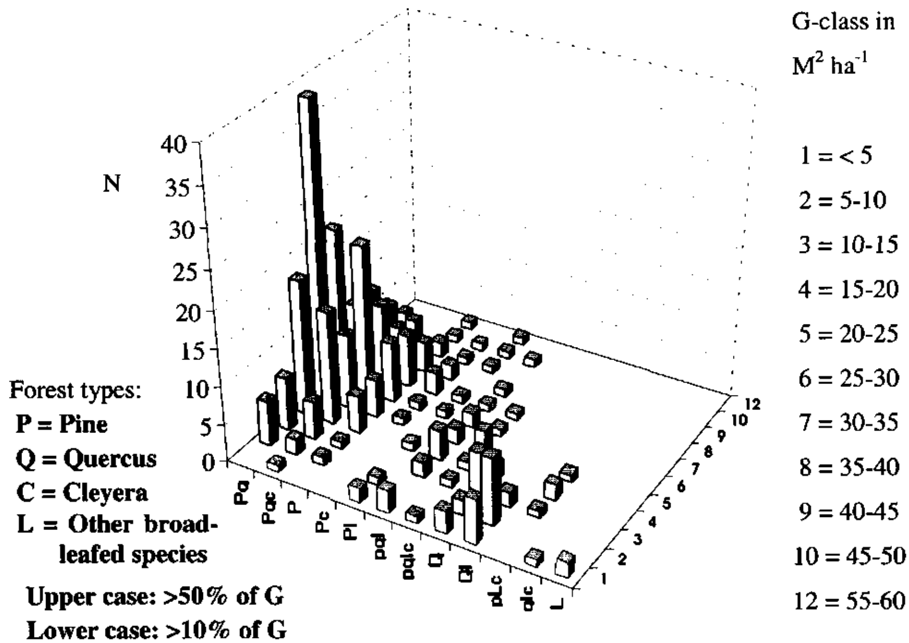


Figure 8.12. *Variability in basal area (G) of trees with $d_{1.30}$ > 10 cm (in m^2 ha^{-1}) and forest types, classified on the basis of percentage basal area of the most important species assemblages of a forest of 3000 ha in Juznajab la Laguna, Chiapas, Mexico. Number of sites was 304; total number of classes 68, with 11 basal area classes and 12 species associations.*

Finally, estimations of stand increment in relation to standing volume also varied in the forest fragment, although a polynomial relation between yearly estimated increment and standing volume could be observed (Figure 6.10).

Concluding remarks

To model carbon flows within a forest fragment, each of the 68 classes that could be distinguished should be treated as a unit and growth data should be available to each of them. However, this would be very difficult and costly, as this would not only imply collecting data on dynamics for each of the 68 classes, but also delimiting the area covered by each class. The 68 classes were therefore grouped into 8 biomass classes, for which biomass densities and confidence intervals were calculated and surface area delimited via satellite image interpretation (Escandón-Calderón et al., 1999).

For each class a separate set of initial compartment state and transfer coefficients were developed and C flux estimated including a 95% confidence interval (See Chapter 6 for the procedure used). Each class associated to a particular biomass group was assumed to behave similarly in terms of biomass dynamics, such that total variation within a group was expected not to exceed the limits of the confidence interval. Monitoring biomass dynamics of a selection of plots within each biomass group for a number of years would be required to test this latter hypothesis.

Presentation of GHG-mitigation benefits in forestry projects

The mitigation benefits of any activity are equivalent to the difference between GHG fluxes as a result of the activity and the amount of GHG fluxes that would occur in the baseline scenario, without any project (Figure 6.1). In forestry, carbon fluxes are continuous, bi-directional, and complex both in time and space (cf. Oldeman 1990, his figs. 5.75 and 6.52). To calculate the benefits of these complex fluxes is difficult and no general procedure has been accepted yet by all involved parties. Various reporting procedures have been used to calculate the GHG offset potential of forestry projects (Figure 8.13; Tipper and De Jong, 1998):

1. Yearly net flows between the vegetation and the atmosphere (expressed in MgC $year^{-1}$) or the sum of the yearly net flows at a given cut-off date (expressed in MgC; Chapter 4; Nabuurs and Mohren, 1993, 1995; De Jong et al., 1998). Annual flows are generally calculated indirectly as the net difference of total C density per unit area in two successive years.

   Long-term average increase in the carbon stock of a managed forest relative to a hypothetical baseline average (expressed in MgC; Chapter 4; Nabuurs and Mohren, 1995; De Jong et al., 1998). This approach assumes a long-term maintenance of the alternative system, calculates the yearly C flows as in procedure 1 and calculates the average effect of the system over a long time horizon, typically around 100 years.

2. Emission equivalent or emission-delayed carbon storage (MgC-eq, expressed in MgC; Chomitz, 1998; Dobes et al., 1998; Tipper and De Jong, 1998). These approaches take into account that CO_2 in the atmosphere has a limited lifetime and is eventually absorbed again by the terrestrial or oceanic biosphere (Tipper and De Jong, 1998). The latter authors base their approach on the hypothesis that the cumulative radiative forcing produced by an emission pulse of 1 Mg CO_2 can be calculated by integrating the amount of CO_2

that remains in the atmosphere in successive years after the emission impulse. The resulting amount is expressed in $MgCO_2$.years. Dobes et al. (1998) consider temporary sequestration in a forest system as a delayed emission. Both approaches are similar in the sense that they incorporate a time dimension in the calculation of the greenhouse benefit, comparable to the calculation of the global warming potential (GWP) of each greenhouse gas.

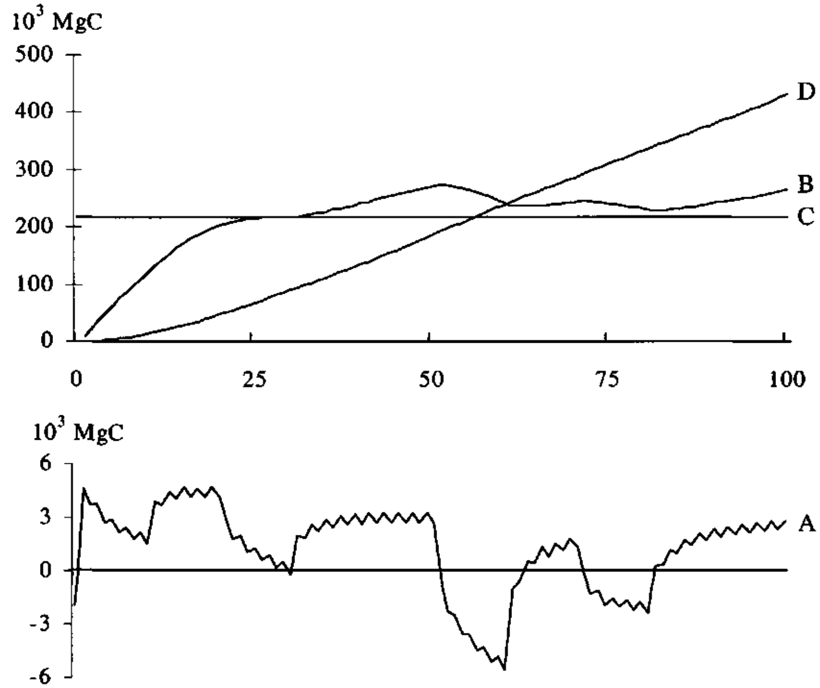


Figure 8.13. *Procedures to present carbon offset benefits. A = annual flux; B = accumulated annual flux; C = long-term average stock increase; D = emission equivalent carbon storage.*

However, one of the major problems with this approach is that the CO_2 depletion in the atmosphere is non-linear, caused by highly variable bi-directional fluxes between terrestrial biosphere- atmosphere and oceanosphere with various feedback mechanisms that are still poorly understood (Houghton et al., 1997). Nevertheless, various authors used this approach and proposed some factor to convert MgC.years cumulative storage to MgC emission equivalent carbon, varying between 42 and 50 (Tipper and De Jong, 1998; Chomitz, 1998) to 150 MgC.years MgC^{-1} (Dobes et al., 1998).

Concluding remarks

Although there are uncertainties about the various assumptions made within the context of comparative radiative forcing and GWPs of the various greenhouse gases, the approach integrated over time, applied in the emission equivalent carbon storage procedure, offers much to make it the most attractive option to calculate the GHG benefits of forestry projects (Chomitz, 1998; Dobes et al., 1998; Tipper and De Jong, 1998; Moura-Costa and Wilson, 2000). The amount of MgC-equivalent emission offset (MgC-eq_i) obtained by implementing a project could be calculated with the following formula, independently of the final value of the conversion factor:

$$\text{MgC-eq} = (\textstyle\sum_i (C_{p,i} - C_{b,i})) / E_f. \qquad (4)$$

where MgC-eq is the emission equivalent offset; $C_{p,i}$ is the C accumulated by the project and $C_{b,i}$ the C accumulated according to the baseline, both in year i and both compared to the C-stock in year zero, and E_f is the emission equivalent conversion factor.

The approach provides for a "gain-as-you go" offset system that automatically addresses risks associated with non-compliance, such as forest fire (Dobes et al., 1998), as climate benefits can be calculated for any year that a particular project runs.

Monitoring of carbon sequestration projects with multiple participants and high variety of land-use systems and ecological conditions

Notwithstanding the overwhelming literature available about potential biotic mitigation measures, there still is a large gap between accepting that C fluxes in biotic systems can in fact be modified to help mitigate climate change, and accepting that this modification can take the form of individual projects liable to be monitored and verified as part of a C emissions control system (Trexler, 1993). By definition, *monitoring* activities of C mitigation projects measure all significant flows, whereas *verification* aims at evaluating the accuracy and reliability of the monitoring set-up. Monitoring programs are important for land-use projects to increase the reliability of data and to improve project performance. To ensure that GHG mitigation data of forestry projects are transparent, reliable and verifiable, guidelines are needed to provide structure and direction for project managers (Sathaye and Ravindranath, 1997). These authors also point out that carbon pools in forest systems as well as forest products and energy should be evaluated for their significance (pool size) and vulnerability (rate and direction of change). Decisions about appropriate methodologies and intensities of monitoring depend on the relative importance of the individual pools and their expected changes (significance, speed and direction of change). Possible issues of leakage must be addressed within a regional or national context. Leakage is the term used to describe the shifting of activities with GHG implications beyond the boundaries of a project in space and/or time (Andrasko, 1997). The Scolel Té project in Chiapas, Mexico gives an example of how monitoring and leakage can be addressed in a project in which many individual farmers participate with a variety of agroforestry systems and ecological conditions (Chapter 7; Andrasko, 1997). The project is designed to increase farmers´ income by implementing a set of alternative practices chosen by the farmers that also increase carbon stocks. Each farmer presents a "Plan Vivo", i.e. a current and alternative land-use plan of his land, in which he indicates which part of his land he intends to dedicate to the carbon project. This plan is a starting point for monitoring possible land-use changes of the project and non-project plots in the future. In this way, the plan-vivo serves as a tool to monitor project performance and at the same time to check on leakage from [?] within the farmer's production unit. The project is also designed to reduce potential leakage off-site, especially to the Selva Lacandona rainforest (Chapter 7; De Jong et al. 1995; Andrasko, 1997).

The definition of the project boundary has to be clear so that questions as to whether carbon fluxes are related to the project or not can be answered. The boundaries should also be defined within a regional or national context to avoid what Andrasko (1997) calls an "Edge Effect", that is the set of issues of policy and technique emerging at the boundaries of monitoring domains. A direct linkage between individual carbon sequestration projects and regional or national services reporting on carbon flux avoids or diminishes leakage or double accounting of carbon fluxes (Table 8.4). Selecting the monitoring domain boundaries between project and regional and national accounting involves decision-making about the accuracy desired at both levels and financial resources available, seen as a cost-benefit issue. As with other management systems, the procedures of monitoring and verification of regional and project performance should be constantly subject to improvement and refinement. The key to improvement is to reflect upon the main sources of error within the system. In the Scolel Té Pilot Project, for example, the assessment of carbon densities and fluxes of the land management systems is currently based in part only on direct biomass measurements, supplemented by the best available data in the literature. New data will be used to improve C-flux modelling, as these come available from both project and regional monitoring and verification (Figure 7.3).

Project monitoring in the Scolel Té Pilot Project is also set up to improve farmers' compliance with monitoring schedules. This will be subject to gradual improvement by means of modification of the Plan Vivo methodology, training of farmers, and linkage to incentive payments for fulfillment of reporting requirements.

Assessment of offset reliability will ultimately require international agreements on a risk assessment methodology that can commonly be applied across projects and project types worldwide. Crucial to offset reliability is the time horizon of project assessment and whether all projects should have a common time horizon for their analysis (Trexler, 1993). If the carbon emission equivalent accounting approach is adopted, the time horizon will be less of a problem, since offset benefits can be calculated and credited while the project is ongoing. Only if the project performance reaches a point below the baseline expectation, non-compliance actions should be brought to bear.

Table 8.4. *Activities and requirements for a regional GHG mitigation program with individual projects;* ***activities and requirements already defined in Mexico;*** activities and instruments currently elaborated; *activities and requirements that need further definition.*

Level	**Activities**	**Requirements**
PLANNING STAGE		
Region	**Identification of critical areas (INE, 1998)** **Identification of project opportunities (INE, 1998)** **LU/LC change modeling with associated GHG flows (Tipper et al., 1998)** *Identification of government policies and short and medium term development programs* *Identification and quantification of sources of leakage*	Policy instruments to define baseline determination *Agreement on default values of major parameters* *Guidelines on data reporting* *LU/LC change detection protocols* *Leakage tracking procedures* Quantification procedures to measure GHG impact of policies **Identification and acceptance of a minimum set of credible, well designed, verifiable forest mitigation activities (Chapter 4)** **Data on carbon densities and variation in LU/LC classes (Chapter 3; Chapter 6; De Jong et al., 2000).**
Project	Elaboration of proposals of farmer and community management plans *Ex ante* estimation of GHG fluxes in management systems Identification of possible sources of leakage	Policy instruments to standardize accounting, modeling and data presentation procedures

Level	**Activities**	**Requirements**
Both	**Exchange of information to improve accuracy of data (Chapter 7)**	Standardization of data presentation Data integration procedures
IMPLEMENTATION STAGE		
Region	Periodic LU/LC change assessments	
Project	**Plot and system wise monitoring and verification of GHG fluxes in managed and control plots (Scolel Té, 2000)**	Standardization of data Data integration procedures

References

Andrasko, K. 1997. Forest Management for Greenhouse Gas Benefits: Resolving Monitoring Issues Across Project and National Boundaries. Mitigation and Adaptation Strategies for Global Change 2(2-3): 117-132.

Arp, P.A. and McGrath, T.P. 1987. A parameter-based method for modelling biomass accumulations in forest stands: Theory. Ecological Modelling 36: 29-48.

Brown, S., Gillespie, A.J.R., and Lugo, A.E. 1989. Biomass estimation for Tropical Forests with Applications to Forest Inventory Data, Forest Science 35(4): 881-902.

Chomitz, K. 1998. Baselines for greenhouse gas reductions: problems, precedents, solutions. Draft paper. WorldBank. http://www-esd.worldbank.org/aij/meth.htm.

De Jong, B.H.J., and Montoya-Gómez, G., 1994. Sustainable management of forest resources: a proposal for the highlands of Chiapas, Mexico. Pages 7-17 in Proceedings of the 1994 symposium on systems analysis in forest resources: management systems for a global economy with global resource concerns, Sept 6-9, 1994. Pacific Grove, CA.

De Jong, B.H.J. Ochoa-Gaona, S., Soto-Pinto, L., Castillo-Santiago, M.A., Montoya-Gómez, G., Tipper, R., and March-Mifsut, I. 1998. Modelling Forestry and Agroforestry Opportunities for Carbon Mitigation at a Landscape Level. En: G.J. Nabuurs, T. Nuutinen, H. Bartelink and M. Korhonen (Eds.). Forest Scenario modelling for ecosystem management at landscape level, EFI Proceedings No 19. pp. 221-237.

De Jong, B.H.J., Ochoa-Gaona, S., Castillo-Santiago, M.A., Ramírez-Marcial, N., and Cairns, M.A. 2000. Carbon Flux and Patterns of Land-Use/Land-Cover Change in the Selva Lacandona, México. Ambio (Accepted).

De Jong, B.H.J. y Ruíz-Díaz, M.J., 1997. La investigación forestal y su contribución al manejo de bosques. In: Los Altos de Chiapas: Agricultura y crisis rural (Parra-Vázquez, M.R. y Díaz-Hernández, B.M., eds). ECOSUR, San Cristóbal de las Casas, México. 149-165.

De Jong, B., G. Montoya-Gomez, K. Nelson, L. Soto-Pinto, J. Taylor and R. Tipper, 1995: Community Forest management and Carbon Sequestration: A Feasibility Study from Chiapas, Mexico, Interciencia 20(6): 409-417.

Dixon R.K., Brown, S., Houghton, R.A., Solomon, A.M., Trexler, M.C. and Wisniewski, J. 1994a. Carbon Pools and Flux of Global Forest Ecosystems, Science 263:.185-190.

Dixon, R., Winjum, J., Andrasko, K., Lee, J. and Schroeder, P., 1994b: Integrated systems: assessment of promising agroforest and alternative land-use practices to enhance carbon conservation and sequestration, Climatic Change, 30: 1-23.

Dobes, L., I. Enting and C. Mitchell, 1998: Accounting for carbon sinks: the problem of time. In: Dobes, L. (ed.) Trading Greenhouse emissions: some Australian perspectives. Occasional papers No 115. Bureau of Transport Economics, Australia.

Duvigneaud, P. and Denaeyer-De Smet, S. 1970. Biological cycling of minerals in temperate deciduous forests. In: Reichle, D.E. (Ed). Ecological Studies: Analysis and Synthesis / Analysis of Temperate Forest Ecosystems. Springer. New York. pp. 199-225.

Escandón-Calderón, J., De Jong, B.H.J., Ochoa-Gaona, S., March-Mifsut, I., Castillo-Santiago, M.A. 1999. Evaluación de dos métodos de estimación de biomasa arbórea a través de datos LANDSAT TM en Jusnajab, Chiapas, México: Estudio de caso. Investigaciones Geográficas 40: 71-84.

Faeth, P., C. Cort and R. Livernash, 1994: Evaluating the Carbon Sequestration Benefits of Forestry Projects in Developing Countries. World Resource Institute, Washington.

FAO, 1996: Forest resources assessment 1990: survey of tropical forest cover and study of change processes. FAO Forestry Paper 130, Rome, Italy.

Fearnside, F.M. and Malheiros-Guimaraes, W. 1996. Carbon uptake by secondary forests in Brazilian Amazonia. Forest Ecology and Management 80: 35-46.

Gardner, R.H. and Mankin, J.B. 1981. Analysis of biomass allocation in forest floor ecosystems of IBP. In: D.E. Reichle (Ed.). Dynamic properties of forest ecosystems. Cambridge University Press. New York. pp. 451-498.

González-Espinosa, M., Ochoa-Gaona, S., Ramírez-Marcial, N., and Quintana-Ascencio, P.F.. 1995. Current land-use trends and conservation of old-growth forest habitats in the highlands of Chiapas, Mexico. Pages 190-198 in M.H. Wilson and S.A. Sader (eds.) Conservation of Neotropical Migratory Birds in Mexico. Maine Agricultural and Forest Experiment Station, Miscellaneous Publication 727, Orono, ME.

Grace, J. 1997. Towards Models of Resource Allocation by Plants. In: Bazzaz, F.A. and Grace J. (Eds). Plant Resource Allocation. Academic Press, San Diego, USA. pp. 279-291.

Hall, D.O., Mynick, H. and Williams, R. 1991. Cooling the greenhouse with bioenergy. Nature 353: 11-12.

Hamilton, L.S. 1991. Tropical forests: identifying and clarifying issues. Unasylva 166, vol 42: 19-27.

Houghton, J.T., Meira Filho, L.G., Griggs, D.J., Maskell, K., 1997. An introduction to Simple Climate Models used in the IPCC Second Assessment Report. IPCC. WGI. Technical Report no II. 51 pp.

INE, 1998. http://www.ine.gob.mx.

Jiménez Ferrer, G., Alayón Gamboa, J.A., De Jong, B.H.J., Soto Pinto M.L., and Montoya Gómez G. 1999. Sistemas Agroforestales Y Captura De Carbono (Co_2) En La Región Norte De Chiapas, México. Ecosur. SCLC, Chiapas, México. 99 pp.

Kellomäki, S. and Kolström, M. 1992. Simulation of tree species composition and organic matter accumulation in Finnish boreal forests under changing climatic conditions. Vegetatio 102: 47-68.

Konstant, T.C., Newton, A.C., Taylor, J.H., and Tipper, R.1999. The Potential for Community-Based Forest Management in Chiapas, Mexico: A Comparison of Two Case Studies. Journal of Sustainable Forestry 9(3): 169-191.

Larson, B.C. 1992. Pathways of development in mixed-species stands. In: Kelty, M.J., Larson, B.C., and Oliver, C.D. (Eds). The Ecology and Silviculture of Mixed-Species Forests, Kluwer Academic Publ, The Netherlands. pp. 3-10.

Marland G. and Marland S., (1992), "Should We Store Carbon in Trees?" Water, Air and Soil Pollution 64: 181-195.

Marland, G. and B. Schlamadinger. 1997. Forests for carbon sequestration or fossil fuel substitution? A sensitivity analysis. Biomass and Bioenergy 13(6): 389-397.

Mohren, G.M.J. and C.G.M. Klein-Goldewijk, 1990. CO2FIX: A dynamic model of the CO2-fixation in forest stands. "De Dorschkamp", Research Institute for Forestry and Urban Ecology, Report nr. 624, 35 pp. + app. Wageningen, The Netherlands.

Mohren, G.M.J., Garza Caligaris, J.F., Masera, O., Kanninen, M., Karjalainen, T. and Nabuurs, G.J. 1999. CO2FIX for windows: a dynamic model of the CO2 fixation in forest stands. Institute for Forestry and Nature Research, Instituto de Ecología de la UNAM, Centro Agronomico Tropical de Investigación y Enseñanza (CATIE), European Forest Institute. Wageningen, The Netherlands; Patzcuaro, Mexico; Turrialba, Costa Rica; Joenssu, Finland. 27 pp.

Moura-Costa, P.H. and Wilson, C. 2000. An equivalence factor between CO_2 avoided emissions and sequestration – description and applications in forestry. Mitigation and Adaptation Strategies for Global Change (submitted).

Nabuurs G.J. and Mohren, G.M.J. 1993. Carbon fixation through forestation activities: A study of the carbon sequestering potential of selected forest types, commissioned by the Foundation Face. Inst. for Forestry and Nature Research (IBN-DLO), Wageningen, The Netherlands, IBN Research Report 93/4. 204 pp.

Nabuurs, G.J. and Mohren, G.M.J. 1995. Modelling analysis of potential carbon sequestration in selected forest types. Canadian Journal of Forest Research 25: 1157-1172.

Ochoa-Gaona, S and González-Espinosa, M. 2000. Land-use and deforestation in the Highlands of Chiapas, Mexico. Applied Geography 20: 17-42.

Ochoa-Gaona, S. 2000. Traditional Land-Use Systems and Patterns of Forest Fragmentation in the highlands of Chiapas, México. Environmental Management (Accepted).

Oldeman, R.A.A. and Van Dijk, J. 1991. Diagnosis of the temperament of tropical rain forest trees. In: Gómez-Pompa, A., Whtimore, T.C., and Hadley, M. (Eds.). Rain forest regeneration and management. MAB Series, Vol 6, UNESCO, Paris. pp 21-65.

Phillips, O.L., Malhi, Y., Higuchi, N., Laurance, W.F., Núñez, Váxquez, R.M., Laurance, S.G., Ferreira, L.V., Stern, M., Brown, S. and Grace, J. 1998. Changes in the carbon balance of tropical forests: evidence from long-term plots. Science, 282: 439-442.

Rudel, T. and Roper, J. 1997. The Paths to Rain Forest Destruction: Cross-national Patterns of Tropical deforestation, 1975-90. World development 25(1): 53-65.

Sathaye, J. and Ravindranath, N.H. 1997. Policies, Measures and the Monitoring Needs of Forest Sector Carbon Mitigation. Mitigation and Adaptation Strategies for Global Change 2(2-3): 101-115.

Schlamadinger, B. and Marland, G. 1996. The role of forest and bioenergy strategies in the global carbon cycle. Biomass and Bioenergy 10: 275-300.

Scolel Té, 1998. http://www.ed.ac.uk/~ebfr11; http://www.ecosur.mx.

Shugart, H.H. 1984. A Theory of Forest Dynamics: The ecological implications of forest succession models, Srpinger-Verlag, New York. 278 pp.

Sykes, M.T. and Prentice, I.C. 1996. Carbon storage and climate change in Swedish forests: a comparison of static and dynamic modelling approaches. In: Apps, M.J. and Price, D.T. (Eds.). Forest Ecosystems, Forest Management and the Global Carbon Cycle. NATO ASI Series I: Global Environmental Change 40; Springer; Berlin. pp. 69-78.

Symfor, 2000. http://meranti.ierm.ed.ac.uk/Symfor/Hlp/contents.htm.

Thornley, J.A.M. 1976. Mathematical Models in Plant Physiology. Academic Press, New York. 318 pp.

Tipper, R. and De Jong, B.H., 1998: Quantification and regulation of carbon offsets from forestry: comparison of alternative methodologies, with special reference to Chiapas, Mexico. Commonwealth Forestry Review 77: 219-228.

Tipper, R., De Jong, B.H.J., Ochoa-Gaona, S., Soto-Pinto, M.L., Castillo-Santiago, M.A., Montoya-Gómez, G., and March-Mifsut. I. 1998. Assessment of the Cost of Large Scale Forestry for CO2 Sequestration: Evidence from Chiapas, Mexico. Report PH12. International Energy Agency Greenhouse Gas R&D Programme. Cheltenham, Glosc. UK. 84 pp.

Trexler, M.C. 1993. Manipulating biotic carbon sources and sinks for climate change mitigation: Can science keep up with practice? Water, Air, and Soil Pollution 70: 579-593.

Vanclay, J.K. 1994. Modelling forest growth and yield: applications to mixed tropical forests. CAB International Wallingford, UK. 312 pp.

SUMMARY

Forestry measures are not yet specifically included within the current articles relating to the clean development mechanism of the Kyoto protocol on climate change. However, provisions for forestry almost certainly will be included at some stage, given the significance of developing country forests within the global carbon cycle. Forestry and land-use measures to mitigate climate change can serve other interests, and may offer some of the most cost-effective ways to combine climate change mitigation and biosphere restoration. GHG offset projects in the land-use and forestry sector can particularly be attractive if they can be attached to local social, ecological and economic goals. Forestry activities for carbon sequestration have been mostly implemented on a project-by-project basis. The lessons learned from these pilot projects will serve as important precursors for future mitigation projects.

The present study intends to answer some of the important questions that arise when translating projects that have an ecological potential to mitigate carbon excesses, into actual implementation of these projects in a farmer-dominated landscape. Farm and community forestry projects for GHG mitigation in such environments would involve numerous participants, a high variety of small-scaled systems spread over large areas, with individual adaptations of general management procedures due to personal interests, local conditions, and previous experiences.

As part of the scientific backstopping of the Scolel Té international pilot project for carbon sequestration and community forestry in Chiapas, Mexico, the following key questions are addressed: (i) what is the effect of land-use and land-cover change on carbon fluxes in highly fragmented landscapes?; (ii) which land-use systems do resource-poor farmers prefer that could contribute to the greenhouse gas problem?; (iii) what is the carbon mitigation potential of farmers' selected forestry and agroforestry systems, and what would be the cost?; (iv) what are the sources and levels of uncertainties in calculating carbon fluxes in forestry systems?; and (v) can a cost-efficient monitoring system be set up for carbon mitigation in a farm forestry project?

A brief outline is presented of some of the major conceptual constraints of forestry and climate change. Forestry typically deals with the use of a woody vegetation on a certain land surface, which can be viewed as either a type of land use or a human activity that impact a natural or human-induced land cover. The latter approach is considered more convenient if

one is concerned with estimating the impact of forestry on the carbon dynamics within a managed land unit.

Estimating the impact of a forestry project on carbon stocks and fluxes depends to a large extend on the choice of boundary settings, such as geographical delimitation and in- or exclusion of system components. Per definition, a baseline estimate needs to be established, not only to assess the project's impact on carbon but also to verify if the impact is additional to this baseline. Closely linked to the baseline concept is the issue of leakage – the positive or negative greenhouse gas impacts of a project, which occur outside the adopted geographical, temporal or subject area boundaries of the project.

In this book the results are presented of a study to estimate the flux of carbon from the terrestrial ecosystems to the atmosphere from the 1970s and 1990s of an intensively impacted and highly fragmented landscape. Carbon density values were assigned to common land-use and land-cover (LU/LC) classes of vegetation maps that were produced by Mexican governmental organizations and, by differencing areas and carbon pools, the net flux was calculated from the Central Highlands of Chiapas, Mexico. During the period of the study the total area of closed forests was reduced by about 50%, whereas degraded and fragmented forests expanded 56% and cultivated land and pasture areas increased by 8% and 30%, respectively. Total mean carbon densities ranged from a high of 504 MgC ha^{-1} in the oak and evergreen cloud forests class to a low of 147 MgC ha^{-1} in the pasture class. The differences in total carbon densities among the various LU/LC classes were mainly due to differences in living plant biomass. Soil organic matter carbon did not show any significant difference between the various LU/LC classes. About 20 x 10^6 MgC were released to the atmosphere during the period of time covered by the study. Approximately 34% of the 1975 vegetation carbon pool disappeared. The Chiapas Highlands, while comprising just 0.3% of Mexico's surface area, contributed 3% of the net national carbon emissions due to land use.

A feasibility study was carried out to (i) identify farmer preferred and ecologically viable agroforestry/forestry systems and their *ex ante* carbon sequestration potential; and to (ii) assess the economic potential of carbon offsets of such systems. A multidisciplinary team of scientists and farmers carried out the study in the Tojolabal and Tzeltal region, Chiapas, Mexico,. A total of five land-use systems were considered viable, while farmers presented local adjustments in preferred species, planting arrangements, and rotation times. The carbon sequestration potential varied highly between the systems and the regions (from 26.7 to 338.9 MgC ha^{-1}). The total cost of one MgC ha^{-1} varied between $US 1.84 and 3.98 for the systems selected in the Tzeltal region and between \$US 1.47 and 11.15 for the systems in the Tojolabal region. The differences in costs within the same region were due to differences in establishment and opportunity costs, whereas the differences between the two areas were due to differences in the carbon sequestration potential of the regions.

The carbon sequestration potential of the Highlands of Chiapas was estimated, based on an incentive program that would stimulate small farmers and rural communities to adopt biomass-accumulating measures, such as agroforestry or improved forest management.

Current vegetation types, land cover and carbon stocks were measured or estimated of an area of around 600,000 ha in southern Mexico. The carbon sequestration potential of a number of alternative land-use techniques, based on farmers' preferences, was estimated. Cost and benefit flows in \$US per Megagram ($=10^6$ g) of carbon (MgC) of each current and alternative system were developed. A model was designed to calculate the expected response to financial incentives of between \$US 0 and 40 per MgC sequestered. It was estimated that $38 * 10^6$ MgC could be sequestered for under \$US 15 MgC^{-1}, of which $32 * 10^6$ MgC by means of sustainable forest management. The choice of a baseline rate of biomass loss in the "business-as-usual" scenario remains a critical issue to estimate the carbon sequestration potential of forestry.

Uncertainties are inevitable in any estimation procedure of greenhouse gas dynamics. These may be due to distinct definitions of concepts, such as source or sink category, differences in assumptions or measurement units, the use of simplified "averaged" values, uncertainty in the basic socio-economic activity data which are driving land-use change, and lack of scientific understanding of the basic processes that are leading to carbon emissions and removals (IPCC, 1996).

The main sources of uncertainties observed in the calculations of the GHG-offset potential of a forestry project were related to: (i) classification of LU/LC types, with observed differences of up to around 8% in land cover estimations; (ii) estimation of C-stocks within each LU/LC type, with uncertainties varying from around 13 to 34% in total C-stock; (iii) historical evidence of LU/LC changes and related GHG fluxes applied in baselines, giving rise to uncertainties of up to about 16% in the estimation of fluxes, whereas varying baseline assumptions produced differences between 31 and 73% in the C-mitigation calculations, with levels of uncertainty in the differences of up to 74%; and (iv) simulation techniques used to calculate future baseline and project C-fluxes, which generated uncertainties of up to around 10% in overall C-mitigation estimations.

In Mexico, an estimated 4.5 x 10^6 ha are available for farm forestry, while up to 6.1 x 10^6 ha can be saved from deforestation if current shifting agriculture practices could be more productive and sustainable. Various farm forestry systems are technically, socially and economically viable, including live fences, coffee with shade trees, plantations, tree enrichment of fallows, and taungya, with a carbon sequestration potential varying from 17.6 to 176.3 MgC ha^{-1}. A self-reporting system with on-site spot-checks is the most appropriate method to assess the impact on carbon fluxes of a farm-forestry project, which typically include a high diversity of small-scale systems and numerous participants. The monitoring and evaluation procedure outlined in this study, facilitates the collection of field data at low cost, helps to ensure that the systems continue to address the needs of farmers, and gives the farmers an understanding of the value of the service that they are providing.

Any method to estimate carbon dynamics has to deal with sources and levels of variability and uncertainties in data. In this dissertation various approaches were used to estimate the impact of variability or uncertainty in data or assumptions. In Chapter 3, the standard deviation of collected biomass data was used as an indicator to estimate the confidence interval of the regional C flux. In Chapter 4, the uncertainty of information was dealt with by varying tree growth due to expected differences in site conditions. In Chapter 5, a sensitivity analysis was used to test the impact of baseline emission assumptions and capital interest rates on the cost of mitigating one Mg carbon. Varying baseline assumptions and carbon transfer parameters within a forest ecosystem was used in Chapter 6 to identify the most important sources of error in the C mitigation estimation of forestry projects.

RESUMEN

Aunque el mecanismo de *desarrollo limpio* del protocolo de Kioto sobre cambio climático aún no incluye medidas forestales, es muy probable su inclusión en alguna fase posterior, dada la importancia de los bosques tropicales en el ciclo global del carbono. La aplicación de políticas forestales y de uso de suelo para mitigar el efecto del cambio climático puede atender al mismo tiempo otras necesidades y así ofrecer mecanismos efectivos y baratos para combinar la mitigación del cambio climático con la restauración de la biósfera. Los proyectos de mitigación de gases de invernadero en el sector forestal son particularmente atractivos cuando se combinan con metas sociales, ecológicas, y económicas. La acreditación de las actividades forestales para la mitigación de carbono hasta la fecha han sido con base en proyectos individuales, estimando el impacto del proyecto mismo sobre los flujos de carbono a la atmósfera, con escenarios de la línea de base y del proyecto dentro los limites del proyecto mismo. Las experiencias de estos proyectos sientan bases importantes para el desarrollo de futuros proyectos.

El presente estudio trata de responder algunas de las preguntas importantes que surgen cuando se quieren aplicar proyectos con un alto potencial ecológico en áreas rurales habitadas por campesinos de bajos recursos, para mitigar excesos de carbono. La implementación de este tipo de proyectos a nivel de productor o comunidad, involucra un gran número de participantes, con una gran variedad de sistemas a pequeña escala distribuidos sobre grandes áreas, cada uno con un manejo específico para cada sitio, adaptado individualmente a los intereses personales, a las condiciones locales y a las experiencias previas de cada productor.

En éste contexto general se desarrolló el proyecto piloto internacional para el secuestro de carbono y manejo forestal comunitario Scolel Té en Chiapas, México. Como parte integral de este proyecto, las siguientes preguntas requerían una respuesta: (i)¿Cuál es el efecto de cambio de uso de suelo y cobertura vegetal sobre los flujos de carbono en paisajes altamente fragmentados?; (ii)¿Cuales de las alternativas de uso de suelo que contribuyen a mitigar las emisiones de gases de invernadero prefieren implementar los productores de escasos recursos?; (iii) ¿ Cuál es el potencial de mitigación de carbono de los sistemas forestales y agroforestales seleccionados por los productores, y cuál es el costo de su implementación?; (iv) ¿Cuáles son las fuentes y niveles de incertidumbre en los cálculos de flujos de carbono en sistemas forestales fragmentados que estan sujetos a cambios constantes?; y (v) ¿Es posible

desarrollar sistemas de monitoreo efectivos y baratos para proyectos forestales para la mitigación de carbono aplicados por campesinos?

El sector forestal típicamente se centra en el uso de la vegetación leñosa presente en un área dada. Esta actividad puede ser considerada como un uso de suelo o como actividades humanas que impactan la cobertura natural ó inducida. La última definición es más conveniente cuando el análisis se centra en la evaluación del impacto forestal sobre la dinámica de carbono. Las estimaciones del impacto de un proyecto forestal sobre los reservorios y flujos de carbono dependen de la selección del límite geográfico y de la inclusión o exclusión de los componentes del sistema. Se requiere una línea de base tanto para evaluar el impacto del proyecto sobre la captura de carbono, como para verificar que dicho impacto es adicional a la dinámica del carbono en ausencia del proyecto. El término "fuga" está muy relacionado con el concepto "línea de base", ya que indica los impactos positivos o negativos del proyecto sobre los flujos de gases de invernadero, que ocurren afuera del límite geográfico, sectorial o temporal del proyecto.

Se presentan los resultados de un estudio que se realizó en un paisaje altamente fragmentado e impactado por actividades humanas. Con el fin de estimar los flujos de carbono entre los ecosistemas terrestres y la atmósfera entre 1970s y 1990s, se aplicaron datos de densidades de carbono por unidad de superficie con base en datos colectados en el campo y mapas de uso de suelo y cobertura vegetal elaborados por organizaciones gubernamentales Mexicanas. El flujo neto de carbono fue calculado, restando los reservorios de carbono de las dos épocas que abarcan 16 años. Durante este período los bosques cerrados disminuyeron en un 50%, mientras que los bosques degradados y/o fragmentados se expandieron en un 56%, las áreas de cultivo en 8% y las de pastizales en 30%. El total de carbono presente en las clases de uso de suelo varió entre 504 MgC ha^{-1} en bosques de encino y nebliselva a 147 MgC ha^{-1} en los pastizales. Las diferencias en densidades de carbono entre las clases de hábitat se debieron a cambios en biomasa. La materia orgánica en el suelo no mostró diferencias significativas. En total alrededor de 20×10^6 MgC fueron emitidos a la atmósfera durante el periodo cubierto por el estudio. Aproximadamente 34% del reservorio de vegetación de 1975 desapareció. La región Altos de Chiapas contribuyó en 3% a las emisiones nacionales de carbono, mientras que el área representa alrededor del 0.3% de la superficie de México.

Se realizó un estudio de factibilidad para (i) identificar los sistemas agroforestales y forestales ecológicamente factibles y preferidos por los campesinos y estimar *ex ante* su potencial de secuestro de carbono (ii) evaluar el potencial económico del secuestro de carbono de estos sistemas. El estudio fue realizado por un grupo multidisciplinario de investigadores con campesinos de dos regiones ecológicas e indígenas, las zonas Tojolabal y Tzeltal de Chiapas, México. En total cinco sistemas fueron considerados viables, con variaciones locales de selección de especies, arreglos espaciales, y rotaciones de árboles. El potencial de captura de carbono varió considerablemente entre los sistemas y regiones (de 26.7 a 338.9 MgC ha^{-1}). El costo total por MgC ha^{-1} varió entre $US 1.84 y 3.98 para los sistemas seleccionados en la zona Tzeltal y entre $US 1.47 y 11.15 para los sistemas en la zona Tojolabal. Las diferencias en los costos dentro una misma región se deben a las diferencias en los costos de establecimiento y oportunidad, mientras que las diferencias entre las dos zonas se deben a las diferencias en el potencial de captura entre las regiones.

El potencial de captura para Los Altos de Chiapas se estimó bajo un programa de incentivos que estimularía a los campesinos y comunidades rurales a adoptar medidas para acumular biomasa a través del establecimiento de sistemas agroforestales y del manejo de los bosques. Para ello, se calcularon las superficies de vegetación y uso de suelo y se estimaron los reservorios de carbono presentes en un área de alrededor de 600,000 ha en el sur de México. Asimismo, se estimó el potencial de captura de carbono de prácticas alternativas de uso de suelo, basadas en la preferencia de los campesinos. Se calcularon flujos de costos y beneficios en $US por MgC para cada sistema actual y potencial. Se diseñó un modelo para calcular el potencial de captura a partir de incentivos financieros entre $US 0 y 40 para cada MgC secuestrado. El manejo de bosques naturales en áreas comunales parece el método más económico para el secuestro de carbono. Con un costo menor a $US 15 MgC^{-1}, en total se pudiera secuestrar hasta 38 x 10^6 MgC, de los cuales el 84% a través de manejo forestal. La selección de una tasa de pérdida anual de carbono para la línea de base se consideró un asunto crítico para el cálculo de la efectividad económica de proyectos forestales para la captura de carbono.

Se compararon varios métodos para estimar el impacto de sistemas forestales y agroforestales sobre los flujos de carbono. En cualquier estimación de dinámica de gases de invernadero inevitablemente existen incertidumbres, debido a las diferencias en la interpretación de los conceptos emisión o reservorio, en las unidades de medición, en el uso de valores "promedio", en las variaciones en los datos básicos de las actividades socio-económicas, y a la falta de entendimiento científico de los procesos básicos en la producción de emisiones y en la captura de carbono (IPCC, 1996). Los factores principales de incertidumbre en el cálculo del impacto de un proyecto forestal sobre los flujos de gases de invernadero, se relacionaron con:

(i) clasificación de tipos se uso de suelo y cobertura vegetal, con diferencias hasta un 8% en las estimaciones de las superficies; (ii) estimaciones de los reservorios de carbono en cada tipo de uso de suelo, con incertidumbre que variaron entre 13% y 34% del reservorio total; (iii) evidencias históricas de cambios de uso de suelo y flujos de carbono aplicados en líneas de base, dando niveles de incertidumbres hasta unos 16%, mientras que las variaciones en la definición de la línea de base mostraron diferencias entre 31 y 73% en los cálculos de captura de C, con niveles de incertidumbre hasta de 74%; y (iv) diferentes técnicas de simulación para proyectar futuros flujos de carbono para las líneas de base y proyectos generaron un nivel de incertidumbre de aproximadamente 10%.

En México, alrededor de 4.5 x 10^6 ha son disponibles para la agroforestería, mientras que 6.1 x 10^6 ha se pueden rescatar de la deforestación a través de proyectos que mejoran la roza-tumba-quema. Varios sistemas forestales y agroforestales son viables desde el punto de vista técnico, social y económico, incluyendo cercos vivos, café con sombra mejorada, plantaciones, acahual mejorado, y taungya, con un potencial de captura de carbono que varía entre 17.6 y 176.3 MgC ha^{-1}. Un sistema de control interno con muestreos aleatorios *in situ* parece el método más apropiado para monitorear el impacto sobre los flujos de carbono de un proyecto forestal realizado por campesinos, el cual típicamente incluye una gran diversidad de sistemas en pequeña escala y numerosos participantes. El método de monitoreo y evaluación que se propone en este estudio facilita la colección de información en forma efectiva y a bajo costo, asegura que el sistema atienda de manera continua las necesidades de los campesinos, y facilita la comprensión de los campesinos del valor del servicio que están ofreciendo.

En esta disertación se utilizaron varios métodos para calcular el impacto de la variabilidad e incertidumbre en los datos. En capítulo 3 se utilizó la desviación estándar como indicador para estimar los rangos de confiabilidad de los flujos de carbono en el ámbito regional. En el capítulo 4 se aplicó un rango de incremento anual del componente arbóreo debido a las diferencias en la calidad del sitio. En capítulo 5 se utilizó un análisis de sensibilidad para determinar el impacto sobre el costo de captura de carbono aplicando variaciones en la línea de base y tasas de interés. En capítulo 6 se modificaron las consideraciones de la línea de base y los parámetros de transferencia de carbono entre los diferentes reservorios de un sistema forestal, con el fin de identificar las fuentes de error más importantes en las estimaciones de captura de carbono.

SAMENVATTING

Bosbouwkundige maatregelen zijn nog niet gespecificeerd in het Kyoto protocol over klimaatsverandering. Gezien het belang van de tropische bossen in de globale koolstofhuishouding zullen deze in de naaste toekomst naar alle waarschijnlijkheid nog worden toegevoegd. Aangezien bosbouw en andere landbouwkundige maatregelen die de klimaatsverandering kunnen tegengaan tevens andere belangen kunnen dienen, worden ze beschouwd als zeer kosten-effectieve maatregelen. Projecten in de land- en bosbouwsector die koolstof uit de atmosfeer kunnen halen zijn met name aantrekkelijk als deze gekoppeld worden aan lokale sociale, ecologische en economische doelstellingen.

Bosbouwactiviteiten die tevens de koolstof uit de atmosfeer vastleggen, zijn tot nu toe alleen op projectbasis ontwikkeld, waarbij het effect van het project vergeleken wordt met een hypothetisch 'referentie scenario'. De ervaringen die in de huidige proefperiode opgedaan worden met dit soort projecten zijn van groot belang voor het ontwikkelen van toekomstige projecten. In deze dissertatie wordt antwoord gegeven op vragen die opkomen als projecten, die ecologisch gezien koolstof uit de atmosfeer kunnen vastleggen, vertaald moeten worden in projecten die toegepast worden in landschappen die door kleinschalige boeren worden beheerd. Bosbouwprojecten in boerengemeenschappen zullen te maken krijgen met talrijke deelnemers, een grote variatie van kleinschalige systemen verspreid over grote gebieden, waarbij iedere boer de voorgestelde systemen aanpast aan zijn persoonlijke interesses, lokale condities en kennis.

In het kader van de "Scolel Te international pilot project for carbon sequestration and community forestry in Chiapas, Mexico", worden in dit boek de volgende wetenschappelijke vragen beantwoord: (i) Wat is het effect van veranderingen in landgebruik op de koolstofhuishouding in een dichtbevolkt landschap? (ii) Welke landgebruiksystemen die kunnen bijdragen aan de klimaatsproblematiek worden het liefst door kleinschalige boeren toegepast? (iii) Hoeveel koolstof kan er gebonden worden met deze door boeren geselecteerde landgebruiksystemen en wat zijn de kosten die er aan verbonden zijn? (iv) Wat zijn de bronnen van onzekerheid in de berekeningsmethodes die toegepast worden om de koolstofbinding in bosbouwsystemen te schatten? (v) Kan een effectief systeem worden opgezet om de koolstofbinding van boerenbosbouwprojecten te controleren?

In dit boek wordt een korte uitleg gegeven van de belangrijkste conceptuele problemen in de relatie bosbouw - klimaatsverandering. Bosbouw betreft met name het gebruik van houtige gewassen op een bepaald stuk land, wat gezien kan worden als een soort landgebruik of als

een menselijke activiteit dat ingrijpt in natuurlijke of door mensen ontwikkelde ecosystemen. De tweede benadering lijkt aantrekkelijker, als een schatting gemaakt dient te worden van het effect van bosbouw op de koolstofhuishouding.

Het effect van bosbouw op koolstofreservoirs en -huishouding hangt in grote mate af van de ligging van de geografische- en systeemgrenzen. Om het effect van een project te schatten dient er een referentie scenario te worden ontwikkeld, waarmee het additionele effect van het project geschat kan worden. Het referentie scenario hangt nauw samen met het lekkage-probleem, dat op zijn beurt gedefinieerd wordt als de positieve en negatieve effecten van een project buiten de geografische- en systeemgrenzen, die vastgesteld zijn in het project.

Deze studie heeft als doel de hoeveel koolstof te schatten die is vrijgekomen door veranderingen in het landgebruik in een gebied van ongeveer 600 000 ha gelegen in de centrale hooglanden van Chiapas, Mexico. De hoeveelheid aanwezige koolstof werd gemeten in de meest algemene ecosystemen die in het gebied voorkomen en vermenigvuldigd met de oppervlaktes die door deze ecosystemen in de jaren zeventig en negentig werden bedekt. Door de hoeveelheid koolstof die aanwezig was in beide periodes van elkaar af te trekken, werd een schatting gemaakt van de hoeveelheid koolstof die in de atmosfeer terecht is gekomen in de vorm van CO_2. Gedurende deze periode nam de totale oppervlakte van gesloten bos af met ongeveer 50%, terwijl de oppervlakte van verstoord bos toenam met 56%, landbouwgebieden met 8% en graslanden met 30%. De hoeveelheid koolstof die aanwezig was in de verschillende ecosystemen varieerde van 504 MgC ha^{-1} in eiken en nevelbossen, tot ongeveer 147 MgC ha^{-1} in graslanden. De verschillen zijn met name toe te schrijven aan de hoeveelheid levende biomassa. De koolstof in organisch bodemmateriaal varieerde niet veel. In totaal werd naar schatting 20 x 10^6 MgC aan de atmosfeer toegevoegd. Ongeveer 34% van de koolstof die in de jaren zeventig in de vegetatie aanwezig was, verdween in de jaren negentig. De hooglanden van Chiapas droegen ongeveer 3% bij aan de nationale koolstofuitstoot veroorzaakt door landgebruik, terwijl het gebied maar 0.3% van het totale Mexicaanse landoppervlak beslaat.

Deze studie werd uitgevoerd (i) om te bepalen welke agroforestry en bosbouwkundige systemen de boeren toepassen en hoeveel koolstof door deze systemen zou kunnen worden vastgelegd en (ii) wat het economisch potentieel aan koolstofvastlegging zou zijn als deze systemen zouden worden toegepast. Een groep wetenschappers en boeren voerden de studie uit in twee etnische gebieden in Chiapas, Mexico. In totaal werden vijf systemen door de boeren geselecteerd, met lokale aanpassingen wat betreft soortensamenstelling, aantallen bomen en rotaties. De hoeveelheid koolstof die door deze systemen kan worden vastgelegd liep sterk uiteen, zowel in de systemen als in de regio's (van 26.7 tot 338.9 MgC ha^{-1}). De totale kosten per MgC ha^{-1} varieerden van $US 1.84 tot 3.98 voor de systemen in het Tzeltal gebied en van $US 1.47 tot 11.15 voor de systemen in de Tojolabal regio. De verschillen in kostprijs in een regio zijn toe te schrijven aan de verschillen in de kosten van het opzetten van de systemen en de inkomsten die verloren gaan door het veranderen van bestaande systemen. De verschillen in kostprijs tussen de twee gebieden zijn toe te schrijven aan de verschillen in de hoeveelheid koolstof die in de gebieden en systemen vastgelegd kunnen worden.

De gegevens van voorgaande studies zijn samengevoegd om een schatting te maken van de hoeveelheid koolstof die vastgelegd zou kunnen worden als de door boeren geselecteerde landgebruikmethodes zouden worden toegepast in de hooglanden van Chiapas. Een kosten–baten berekening van elk huidig en alternatief landgebruik werd opgesteld, uitgedrukt in $US MgC^{-1}. Een model werd ontwikkeld waarmee berekend kan worden hoeveel koolstof kan worden vastgelegd als financiële bijdragen van $US 0 tot 40 MgC^{-1} aan de boeren zouden worden verstrekt. Naar schatting 38 x 10^6 MgC zou in het gebied kunnen worden vastgelegd met financiële bijdragen tot $US 15 MgC^{-1}, waarvan 32 x 10^6 MgC via duurzaam beheer van bossen. Het vaststellen van een referentie scenario blijkt echter een belangrijke bron van onzekerheid in de effectiviteitberekening van koolstofvastlegging in bosbouw.

Onzekerheden in schattingen van koolstofstromen in ecosystemen zijn onvermijdelijk, deels door verschillen in interpretatie van basisconcepten, zoals bronnen en reservoirs, deels door verschillen in vooronderstellingen en door gebruik van vereenvoudigde rekenmethodes, zoals 'gemiddelde' waarden. Onzekerheden in gegevens betreffende de sociaal-economische activiteiten kunnen de berekeningen ook beïnvloeden , alsmede het gebrek aan inzicht in de basisprocessen die leiden tot koolstofuitstoot of -vastlegging (IPCC, 1996).

De belangrijkste bronnen van onzekerheid in de berekeningen van de koolstofhuishouding in bosbouw hadden betrekking op: (i) classificatie van landgebruik en vegetatie, met verschillen tot rond 8% in geschatte oppervlaktes; (ii) schattingen van de hoeveelheid koolstof per oppervlakte-eenheid en vegetatieklasse, met onzekerheden die variëren van 13 tot 34% van de gemiddelde koolstofreservoir; (iii) historische gegevens van veranderingen in landgebruik en de daaraan gekoppelde koolstofuitwisselingen, met onzekerheden die kunnen oplopen tot rond 16% van de geschatte waarden, terwijl variaties in referentie scenario's verschillen kunnen veroorzaken van 31 tot 74% in de geschatte hoeveelheden koolstof, met onzekerheden in de verschillen die kunnen oplopen tot 74%; en (iv) simulatietechnieken die de koolstofstromen in de referentie- en projectscenario's schatten, met onzekerheden van ongeveer 10% van de schattingen.

In Mexico zijn ongeveer 4.5 x 10^6 ha beschikbaar voor boerenbosbouw, terwijl 6.1 x 10^6 ha bos kan worden beschermd, als de huidige zwerflandbouwmethodes zouden worden aangepast. Er zijn verschillende landgebruiksystemen ontwikkeld die technisch, economisch en sociaal gezien zouden kunnen worden toegepast, zoals bomen in plaats van palen in omheiningen, schaduwbomen in koffieplantages, bosplantages, taungya. Deze systemen kunnen naar schatting 17.6 tot 176.3 MgC ha^{-1} koolstof vastleggen.
Om de voortgang van dit soort projecten te kunnen controleren en de koolstofvastlegging te meten, is een rapporteringssysteem ontwikkeld waarbij de boeren zelf de voortgang van hun project beschrijven, met steekproefsgewijze controles in het veld om de juistheid van de verstrekte gegevens te testen. De procedure die in deze studie wordt beschreven is kostenbesparend, verhoogt de kans dat de systemen werkelijk in de behoeftes van de boeren voorzien, en biedt de boer meer inzicht in de ecologische waarden van duurzaam landgebruik.
Elke methode die toegepast wordt om koolstofuitwisselingen te schatten, heeft te maken met variaties en onzekerheden in gegevens. In dit boek worden verschillende methodes gebruikt om het effect van de variatie en onzekerheden in gegevens te schatten. In hoofdstuk 3 wordt de standaarddeviatie in de verzamelde biomassa-gegevens gebruikt om het betrouwbaarheidsinterval van de koolstofuitstoot te berekenen. In hoofdstuk 4 wordt de variatie in groeiverwachtingen van de bomen gebruikt om verschillen in standplaats in te calculeren. In hoofdstuk 5 wordt een sensitivity analyse toegepast om het effect te berekenen van variaties in referentie scenario's en rentetarieven op de kostprijs van het vastleggen van één MgC. In hoofdstuk 6 worden tenslotte variaties in referentie scenario's en koolstofuitwisselingsprocessen binnen een ecosysteem gebruikt om de belangrijkste bronnen van onzekerheid aan te kunnen tonen.

GLOSSARY

The following are brief descriptions of the most important terms used in or related to the book. The definitions related to the Framework Convention on Climate Change are adapted from official and non-official documents published by IPCC.

Activity in the context of this dissertation is defined as an action or set of actions, operating on some aspect of a terrestrial ecosystem, within a definable physical area, that results in a net change in the flow of carbon to the atmosphere.

Additionality within the context of the Framework Convention on Climate Change means any reduction in greenhouse gas emissions, attributed to a project, and that would not have occurred otherwise. NOTE: Additionality is clear as a concept, but to assess if any given observed change in carbon stocks is "additional" is far from straightforward, because the "without project" or "business as usual" baselines are counterfactual constructs that cannot be observed, once the project is ongoing (See also Chapter 5 and 6).

Afforestation refers to the establishment of a forest or stand in an area where the preceding vegetation or land use was not forest since human memory.

Agroforestry is the underlying principle of land-use systems and technologies, in which woody perennials are deliberately used on the same land-management units as agricultural crops and/or animals, in some form of spatial arrangement and/or temporal sequence, and with ecological and/or economic interactions between the different components.

Annex-1 country is a country mentioned on the list in Annex 1, in the United Nations Framework Convention on Climate Change. NOTE: These countries are committed to adopt national policies and to take measures to mitigate climate change. The list includes the OECD countries as of 1992, the countries in transition to an open market economy, and the countries of the European Economic Community.

Anthropogenic or ***human-induced*** or ***man-made***, means resulting from human activities. NOTE: Many of the greenhouse gases are emitted naturally. NOTE: It is only the man-made additions beyond natural emissions, which may unbalance natural balances and modify the rate of climate change.

Basal area (g) is the area of the cross section of a tree stem, generally at breast height (h = 1.30 m) and inclusive bark, expressed in m^2; also referring to the sum of the cross sections of the trees present on a surface unit (G), the latter generally expressed in m^2 ha^{-1}.

Baseline refers to the collective set of economic, financial, regulatory and political conditions beyond a project, within which a project will operate during its lifetime. NOTE: There are two approaches to setting baselines. In the first, historical data sets are used to project trends forward into the future according to some model based on the best available information regarding government policies and changing economic, social, and physical conditions. An alternative approach is to develop performance standards or benchmarks for project types, adjusting the standards to 'fit' local conditions and updating them regularly as methodological refinements are made.

Baseline uncertainty is the uncertainty in baseline assumptions that can only be estimated. NOTE: Indeed, baseline determination is inherently a counterfactual exercise, based on the establishment of a series of plausible, but immeasurable assumptions. Consequently, any mitigation project is always exposed to this uncertainty.

Benchmark (or ***Performance criteria)*** is the set of recognized project performance standards, here referred to as the set of standards for environmental management that could be adapted to monitor the associated sustainable development impacts of site-specific mitigation projects, such as the ISO 14000 standards. NOTE: This latter standard is designed to monitor the impacts of a project on the quality of environmental components such as air, water and soil by using a broad assortment of standardized sampling, testing and analytical methods.

Biomass is the total weight or volume of organic material both above ground and below ground, and both living and dead. NOTE: It includes, among others, trees, crops, grasses, tree litter, roots, fauna, etc. In this study, biomass is generally limited to the weight of vegetative material per unit area.

Biomass fuel is the amount of biomass burned for energy purposes, also including gases recovered from the decomposition of organic material.

Biosphere is the spheric layer of and above the earth crust regulated by living organisms, and where these can subsist.

Canopy is the mass of foliage and branches formed collectively by the crowns of trees, including the phyllosphere.

Carbon accounting is a system of calculus designed to record, summarize, and report the quantity of carbon stored in sinks through applicable land-use change and forestry activities for a specific period of time.

Carbon allocation is the distribution of carbon and energy (photosynthates) to the various organs of a plant or components of an ecosystem.

Carbon cycle is the exchange of carbon between the atmosphere, ocean, terrestrial, biosphere and geological deposits, considered to form a cyclic flux.

Carbon density is the numerical measure of total carbon in a unit area. NOTE: In this study it is generally limited to the amount of organic carbon in the biomass and soil.

Carbon flow is the movement of carbon between living and dead components within an ecosystem or between an ecosystem and its environment.

Carbon offset is the amount of carbon withdrawn from the atmosphere for sufficient time to compensate for atmospheric warming over a period of 100 years, caused by an emission of a specified quantity of CO_2 or a CO_2 equivalent quantity of other greenhouse gases. NOTE: In this study it refers to the quantity of carbon withdrawn from the atmosphere and stocked vegetation and soil.

Carbon pool is any of the above- or below-ground biomass compartments of an ecosystem, such as stems, branches, leaves and roots of living and dead plants, litter, humus, soil organic matter, and fauna.

Carbon sequestration or ***mitigation*** is the incorporation of carbon, otherwise threatened to disappear, by growing new biomass or conserving biomass per unit area. NOTE: The carbon can be locked up in wood, leaves, roots, litter, and soil. Part of the carbon will be released again after being sequestered, due to respiration and decomposition processes.

Chiapas is the southern-most state of Mexico, bordering Guatemala.

Climate change is the change in climate beyond and above natural climate variability as observed over comparable time periods, directly or indirectly attributed to human activities that alter the composition of the global atmosphere.

CO_2 equivalent is the concentration of CO_2 exercising the same amount of radiative forcing as the given mixture of CO_2 and other greenhouse gases.

Community forestry is the use of public or communal land for tree growing in forest-like land use systems to meet community needs.

Compliance is the act of conforming or yielding to a specified norm or protocol. NOTE: In the sense of this study, the compliance issue can be thought of as verification of or adherence to established and agreed norms.

Confidence is the trust in a measurement or estimate. NOTE: Confidence in estimates helps to reach a consensus that the data can be applied to problem solving.

Confidence interval is a calculated interval for which one can assert with a given probability, called the degree of confidence or the confidence coefficient, that it will contain the true value of the variable it is intended to estimate.

Consistency (in statistics) is the probability that an estimator depending on the sample size n will assume a value arbitrarily close to the variable that it is intended to estimate approaches one, when n becomes infinite. NOTE: In the IPCC context, consistency can mean that the methods used are the same throughout the time series reported.

Correlation is a mathematical relationship of association or dependence between the values of two or more qualitative or quantitative variables.

Correlation coefficient (r) is a measure that expresses the linear relationship between two quantitative variables. NOTE: The value of r can range from -1 to +1, where 0 indicates no linear relationship, whereas -1 and +1 indicate a perfect negative (inverse) and a perfect positive relationship, respectively.

Cost-effectiveness is a criterion that specifies in how far a technology or measure delivers a good or service at equal or lower cost than current practice.

Deciduousness is regular leaf loss by perennial plants or forests during some specific season of the year. EXAMPLES: deciduous tree, deciduous forest.

Deforestation is the long-term or permanent removal of forest cover from a land surface and subsequent conversion to non-forested land-use. NOTE: This means that the common definition of reforestation in forestry as described below is not the antonym to the common definition of deforestation. That is, in forestry practice reforestation is commonly used to refer to any act of re-establishing a forest when it follows a long period of deforestation (but within human memory, i.e. 50 years says FAO), contrarily to afforestation (after more than 50 years) or forest regeneration immediately after wood harvesting, the latter not being equated with deforestation.

Diameter at Breast Height (in this study abbreviated as DBH and $D_{1.30}$) is the diameter at a fixed height above the ground, in the case of this study at the internationally accepted value of 1.30 meters (IUFRO convention).

Discounting is the mechanism by which a value for time is translated into economic decision-making.

Discount rate is the annual rate at which the effect of future events or expenditures is reduced so as to be comparable to the effect of present events or expenditures.

Ejido is a term used in Mexico for a productive grouping of people with land, given in common usufruct after the 1917 revolution.

Emission ceiling is the total allowable emission assigned to a country or group of countries within a framework of maximum total emissions.

Enriched fallow is the set of woody species planted and left to grow during the "fallow phase" of slash-and-burn agriculture (or shifting cultivation) to yield additional crops or to shorten the fallow period. (See also Fallow and Slash-and-burn agriculture).

Estimate is a quantitative assumption, based on the best scientific data available.

Evaluation combines evaluation of both impact and process of a particular project, typically entailing a more in-depth and rigorous analysis of a project using the monitored data and information. NOTE: Project evaluation in the case of greenhouse gas mitigation projects would include greenhouse-gas and non-greenhouse-gas impacts, and could include repeated estimation of the baseline, and leakage.

Fallow is the phase of the ***Slash-and-burn agriculture*** when the land is left to rest. NOTE: Usually on the land in rest a shrub and/or tree vegetation develops, which often provides products, such as fodder, fuelwood, and fruits.

Farm forestry is the practice of farmers to grow tree stands and/or woodlots on parts of theire farmland.

FCCC or ***UNFCCC*** is the United Nations Framework Convention on Climate Change.

Flux or ***Flow*** is the transfer of (part) of a compartment to another compartment either within the system boundaries, or between the system and its environment.

Forest is an area of land of more than 0.5 hectares, occupied by trees with a total crown cover of more than 10 percent (IPCC definition). NOTE: The trees should be able to reach a minimum height of 5 meters at maturity. Forest may consist either of ***closed forest*** formations where tree crowns of various heights and undergrowth together cover a high proportion of the ground (often defined as more than 40%); or ***open forest*** formations with a continuous vegetation cover in which tree crown cover exceeds 10 percent (in our case less than 40%). Young natural stands and all plantations established for forestry purposes which have yet to reach a crown density of 10 percent or tree height of 5 m are included in the definition of forest, as are areas temporarily without trees as a result of human intervention or natural causes, but which are expected to revert to forest soon. In other words, all development phases of a forest area are included.

Global warming is the predicted increase in the earth's mean temperature due to the use of fossil fuels and certain industrial and agricultural processes leading to a buildup of "greenhouse gases" in the atmosphere. NOTE: these gases are principally carbon dioxide, methane, nitrous oxide, chlorofluorocarbons, and water vapor.

Greenhouse gas allows the shorter wavelengths of radiant energy (such as visible light) to pass through it, but absorbs some of the longer wavelengths of radiant energy (such as infrared radiation). NOTE: Visible sunlight readily passes through the greenhouse gases to reach the earth's surface, which it warms. The earth's surface, much cooler than the sun, emits radiant energy in the form of longer infrared waves. The greenhouse gases absorb some of these infrared waves emitted by the earth's surface. When greenhouse gases absorb infrared energy, they transfer this energy to other gases and the atmosphere warms up.

Highlands of Chiapas or ***Central Plateau*** comprises a limestone mass with volcanic rocks at the highest peaks (See also Fig 3.1). NOTE: In this study only the area above 1,500 m altitude of the Central Plateau is considered.

Human memory, according to FAO covers a timespan of fifty years.

Implementation cost is the sum of all costs involved to establish and maintain a certain human enterprise, here a land-use system, including capital, labor and operating costs.

Increment (i) is the increase of individual trees (i) or stands (I) within a given time period, the former usually expressed in cm yr^{-1} increase in diameter and the latter in m^3 yr^{-1} ha^{-1} increase in wood volume (international IUFRO definitions).

IPCC is the Intergovernmental Panel on Climate Change, a special intergovernmental body established by UNEP and the WMO to provide assessments of the results of climate change research to policy makers.

Joint Implementation refers to the mechanism in which greenhouse-gas emission reductions or offsets in one country counterbalance emissions generated in other countries, in order to achieve reductions agreed under the United Nations Framework Convention on Climate Change.

Land cover is the observed physical and biological cover of the earth's lands, such as vegetation or man-made features.

Land use is the total of arrangements, activities and inputs that people undertake in a certain land cover type.

Landscape is considered the landform of a region together with its associated habitats at scales that range from some hectares to many square kilometers. NOTE: a rural landscape may include forests, cropland, pasture, rivers, and human settlements.

Landscape fragmentation is the process in which rather homogeneous eco-units within a landscape are divided into a mosaic of smaller patches of distinct eco-units. NOTE: for a more fundamental definition see also Rossignol et al., 1998.

Leakage is the shifting of activities with greenhouse gas implications beyond the project boundaries or other types of system boundaries. NOTE: Leakage can be induced by several different mechanisms, such as activity displacement, demand displacement, or investment crowding (See also Chapter 2).

Lifetime is the average period that a given object spends in a given reservoir. NOTE: lifetime is not to be confused with the response time of a perturbation in a concentration. CO_2 itself has no single lifetime.

Marginal cost is the cost due to one additional unit of effort. NOTE: in terms of reducing emissions, it represents the cost of reducing emissions by one unit.

Market-based incentive is a measure intended to directly change relative prices of products or services to overcome market barriers.

Mature forest is a forest that is fully grown and developed. NOTE: in the context of this book it refers to a forest that has reached an equilibrium state between biomass production and biomass decomposition.

Measure is any action that can be taken by a government or a group of governments, often in conjunction with the private sector, to accelerate the use of technologies or other practices that reduce greenhouse gas emissions (definition for this book).

Model is an artificial abstract or physical entity that is constructed to represent in some way the form and/or the function of real-world systems, entities or processes.

Monitoring is a continuous assessment of processes, so that these processes can be corrected. NOTE: in this book the processes are the functioning of project activities, e.g. the monitoring of carbon mitigation projects typically measures all significant carbon flows in time with a view to adjust project activities if something goes wrong.

Monitoring domain is the area that needs to be monitored, based in the present book on the characteristics of a particular *mitigation activity*.

Mortality is the number, percentage or volume of trees that died in a given time and a given forest or forest area. NOTE: the cause of mortality has to be specified, e.g. small, regular natural impacts such as squalls, parasites or thunderbolts, "catastrophes" e.g. storms, fires, or earthquakes, or anthropogenic impacts, e.g. land conversion, mechanical clearcuts, or application of chemicals. In the present book mortality refers to natural causes.

Normal distribution is the classical average distribution of values that, when plotted on a graph, resembles the shape of a bell; the amount by which values vary from each other defining the shape of the normal curve. NOTE: if most of the values are similar, the normal distribution is a tall and thin bell shape, whereas a short and wide bell shape means there is high variation among the values. One measure of the variation among values is called the standard deviation.

Old-growth forest is the collection of stands of trees or forest mosaics, in areas untouched by humans for many centuries.

Opportunity cost generally refers to the cost of an economic activity foregone by the choice of another activity; in the case of this study it is the lost benefit or rent foregone from converting current land use to an alternative system.

Pajal Ya Cak´tik is the farmers´ credit union that started with the Scolel Té pilot carbon sequestration project in Chiapas, Mexico.

Patch is a discrete eco-unit, surrounded by a matrix of differing eco-units. NOTE: patches in a landscape mosaic are coupled by fluxes of organisms, biotic and abiotic energy, and nutrients.

Perturbation is a discrete event that disrupts ecosystem dynamics, substrate availability, or the physical environment.

Phyllosphere is the interface between the leaves and the atmosphere.

Photosynthesis refers to the conversion process of carbon dioxide and water to carbohydrates by green plant cells in the presence of light, whereby oxygen is liberated as a by-product.

Plan-vivo is a planning tool for farmers and a monitoring tool for the project and its performance within the farmer's production unit.

Policy is a procedure developed and implemented by a government, in this book regarding the goal of mitigating climate change by means of of technologies and measures.

Polygon is a closed plane figure bounded by straight lines.

Precision (of an estimator) is its tendency to have its value cluster closely around the expected value of its sampling distribution. NOTE: precision is related inversely to the variance of the sampling distribution - the smaller the variance, the greater the precision.

Project is a large or major undertaking with discrete actions and clear temporal and geographical boundaries, involving objectives, money, personnel, and equipment.

Project cost is the sum of all financial costs of a project, including capital, labor, and operating costs.

Quality assurance is an indicator of the sum of activities that are implemented to ensure the collection and presentation of high quality, reliable data. NOTE: In experimental programs, audits with standard instruments and standard measures are used to establish the reliability of the experimental procedures. For example, in the biomass inventories of this study, quality control standards were developed internally and approved by external auditing.

Quality control is the set of procedures and tests that can be performed during the planning and development of an inventory to ensure that the data quality objectives are being met. NOTE: Quality control may include criteria tests for data on operations, completeness criteria, or averaging techniques for use in developing default parameters.

Radiative forcing is the perturbation of the energy balance of the earth-atmosphere system following changes in compartments of that system or in outside factors. Examples are a change in the concentration of CO2 or a change in the energy output of the sun. The climate system responds to the radiative forcing so as to re-establish the energy balance.

Reforestation is the establishment of trees on land that has been cleared of forest in the relatively recent past (less than fifty years ago, says FAO). NOTE: Clearing can be caused by all kinds of natural and anthropic disasters (see Oldeman 1990). Regeneration after wood harvest is no reforestation, and can be implemented by artificial (plantation) or natural (spontaneous seeding) methods.

Regeneration is natural or human-induced reproduction of trees directly after harvest or removal of pre-existing vegetation.

Relative Reduction in Variance (of a parameter) is the maximum variance reduction that can be achieved by knowing the exact value of the particular parameter in an uncertainty analysis with various parameters.

Relative Specific Variance of a parameter is the minimum residual variance if the value of only the particular parameter remains unknown in an uncertainty analysis with various parameters.

Reliability is the possible size of error in an estimator. NOTE: This term is often used interchangeably with consistency. If the approaches and data sources used in a project are considered reliable, then users will have an acceptable degree of confidence in the data developed by the project.

Remote sensing is the collection of data by a device that is not in physical contact with the object, area, or phenomenon under investigation. EXAMPLES are aerial photography or satellite imagery.

Risk is the probability of an event that negatively affects the expected benefits of a project. NOTE: Land-use projects are exposed to a series of risks, particularly natural and anthropogenic risks, such as excess or deficits in rainfall or sunlight, pests and diseases, reductions in growth rates, encroachment or fires.

Scenario is a plausible description of a plan for future development in a particular way, based on a coherent and internally consistent set of assumptions about key relationships and driving forces.

Scolel Té project is the abbreviated name for the international pilot project for carbon sequestration and community forestry in Chiapas, Mexico.

Secondary succession is the vegetation regrowth starting after the re-setting of the initial conditions by incomplete elimination of an earlier plant community. NOTE: ***primary succession*** is the vegetation regrowth after complete elimination of an earlier vegetation, e.g. by volcanic eruptions, complete erosion of biotic soil horizons or inundation during centuries. Secondary succession starts with a local organic inheritance (e.g. seeds, micro-organisms, spores, eggs), primary succession starts from zero.

Secondary vegetation is the vegetation due to secondary succession. NOTE: regrowth follows incomplete removal of pre-existing vegetation often due to clearance by man leaving behind many small biotic components. In this study the secondary vegetation dominated by shrubs is also called secondary shrub vegetation, distinguished from secondary tree vegetation that is dominated by trees.

Selective cutting is a silvicultural system directed towards the harvesting of certain individual trees of an existing forest which also activates natural forest regeneration and generally leads to a forest mosaic with several age and size classes (See also **uneven-aged stands**).

Sensitivity analysis is the systematic investigation of the reaction of the simulation in response to the extreme values of the model's quantitative factors (parameter and input variables) or to drastic changes in the model's quantitative factors (modules). NOTE: Sensitivity analysis is primarily concerned with the question to know how model outputs are affected by large variations in the value of the model components and provides information in situations where these components are incompletely known or subject to changes or misinterpretation.

Silviculture is the applied ecological science of manipulating a forest in order to fulfill stated management objectives, by controlling the complete cycle of forest establishment, composition, growth and regeneration.

Site is the direct abiotic and biotic environment of a forest, a stand, a tree, or a population.

Site quality is a loose term denoting the relative productivity potential of a site for a particular tree species in plantation, usually expressed as the mean height of a certain set of dominant trees at a certain age.

Slash-and-burn agriculture or ***shifting cultivation*** is the itinerant form of agriculture, whereby the farmer clears a parcel of vegetation, burns the residual plant material and cultivates the soil until it needs to rest in a state of fallow, then moves onto another area where the process starts again.

Species aggregates are artificial groups of tree species, classified according to their similarity in terms of growth and tree form, modeled in this study as compartments of the carbon flow.

Stand is a specific area within a forest with a more or less homogeneous structure in terms of tree species composition, density, and condition (silvicultural definition).

Stand dynamics is the temporal and spatial variation in stand architecture, biomass, species composition, tree height, stand density, and soil condition in a particular site.

Stand structure is in this book defined as the vertical and horizontal arrangement of plants, generally limited to trees.

Standard Deviation (of a sample of size n) is the square root of the sum of the squared differences of the sample values and the arithmetic mean, divided by n -1.

Succession is the general trend in plant community composition and structure against time, in particular regarding the processes of compartments or components replacing each other.

Taungya is a silvicultural system originating from India, in which the initial stages of woody plantations are mixed with food plants, and cared for by the farmers.

Terrestrial biosphere is a collective term for all living organisms on land of the planet Earth.

Thinning is the selective removal of trees from a stand. NOTE: Thinning may have numerous objectives, e.g. to stimulate the growth of the remaining trees by allowing them access to more light, moisture, and nutrients, or improving stand health by eliminating diseased trees.

Transparency is literally the condition in which light passes through and leaves everything visible, but is also the condition of being clear and free from pretence. NOTE: in a transparent model, the construction of estimates is clearly explained; the documentation is sufficient for another party to reconstruct it; and the documentation sufficiently clarifies the major causes of trends in the data. Transparency is greatly increased if the data collected and reported by different agencies are similar and, therefore, easily understood by other parties and comparable to the data presented by the other parties.

Turnover time is the ratio between the mass of a reservoir and the rate of removal from that reservoir.

Uncertainty is a statistical term for the degree of accuracy and precision of data. NOTE: it often expresses the range of possible values of a parameter or a measurement around a mean or preferred value.

Uncertainty analysis is the investigation of the output distribution, given the model and the pre-defined distribution of the inputs. NOTE: In uncertainty analysis the input variables range between the extreme values investigated in sensitivity analysis. In this study the full variance, that is the variance of the output induced by all sources input variances collectively was investigated in the uncertainty analysis.

Understory species are those plant species that grow close to the forest soil under the canopy formed by other plants.

Uneven-aged stand is a forest stand, in which more than two distinct age classes, usually accompanied by a range of size classes, are present.

Validation is an official confirmation or approval of an act or product. NOTE: in the context of mitigation projects, validation involves checking to ensure that the offset calculation is in line with reporting procedures and guidelines. It checks the internal consistency of the data.

Variability is the set of observed differences attributable to true heterogeneity or diversity in a population or parameter. NOTE: sources of variability are the result of random processes. Variability is usually not reducible by further measurement or study, but can often be described.

Variance of a sample is the square of the standard deviation. ***Coefficient of Variance*** usually refers to the variance, expressed in percentage of the mean of the estimate.

Verification is the process of checking whether the measured greenhouse-gas and non-greenhouse-gas impacts have actually occurred (this book). NOTE: The credibility attached to verification increases with transparency and independence of the verifying person or organization. In the Scolel Té project verification also aims at evaluating the accuracy and reliability of the monitoring scheme.

Vulnerable carbon is the set of those pools of organic carbon in an ecosystem that are easily accessible to natural or human-induced biochemical decomposition.

Units:

Mega	(M)	=	10^6	1 MgC = 10^6 gram C = 1 ton C (1 tC)
Giga	(G)	=	10^9	1 GgC = 10^9 gram C
Tiga	(T)	=	10^{12}	1 TgC = 10^{12} gram C = 1 Megaton C (1 MtC)
Peta	(P)	=	10^{15}	1 PgC = 10^{15} gram C = 1 Gigaton C (1 GtC)

ABOUT THE AUTHOR

Bernardus Hendricus Jozeph de Jong was born in Utrecht on March 21, 1949. He studied forestry at Wageningen University from 1972 to 1979, with majors in Silviculture and Nature Conservation and a minor in Plant Taxonomy and Geography.

From 1979 to early 1990 he worked as a researcher and forestry expert in Suriname and Nicaragua and since 1990 he is stationed in southern Mexico.

Since 1992 he is working as a senior scientist at El Colegio de la Frontera Sur (ECOSUR). From 1994 till present he is involved in research on forestry and carbon mitigation, mainly in the Highlands of Chiapas, particularly within the framework of the "Scolel Té international pilot project for carbon sequestration and community forestry in Chiapas, Mexico". In 1996 he was given the opportunity by ECOSUR to dedicate part of his time to culminate the research in a dissertation.

After finishing his sabbatical leave, which he used to write this thesis, he will continue his research with ECOSUR in the state of Tabasco, where he will start research on forestry and agroforestry in the humid tropics of Mexico. He is looking forward to this new exciting challenge.