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ANALYSIS

An economic analysis of the potential for carbon sequestration by forests: evidence from southern Mexico

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Abstract

Forestry has been proposed as a means to reduce net greenhouse gas emissions, by either reducing sources or enhancing sinks. This study assesses the potential of an incentive-based program to stimulate small farmers and communities to adopt biomass accumulating measures such as agroforestry or improved forest management. Current vegetation type, land use and stocks of carbon were assessed for an area of around 600 000 ha in southern Mexico, and the carbon (C) sequestration potential of a number of alternative 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. The most cost-effective method for sequestering carbon appears to be the improved management of natural forest on communal lands. We estimated that 38×10^6 MgC could be sequestered for under US \$15 MgC^{-1}, of which 32×10^6 MgC through forest management. The choice of a baseline rate of biomass loss under a 'business-as-usual' scenario remains a critical issue for estimates of the cost-effectiveness of carbon sequestration by forestry. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Carbon sequestration supply; Forest management; Land-use change; Cost-effectiveness analysis

1. 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

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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 60–90 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 (Masera et al., 1995; Dixon et al., 1996; 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.

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?

2. Methods

2.1. Study area

The Central Highlands of Chiapas (Los Altos, 607 500 ha, 1500–2900 m a.s.l., Fig. 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-Xolocotzi, 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 (García, 1982). The soils are predominantly derived from calcareous rocks, and include cambisols, leptosols, luvisols, ferrosols, nitisols, lixisols, acrisols, and feozems (De Jong et al., 1999).

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.

2.2. 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

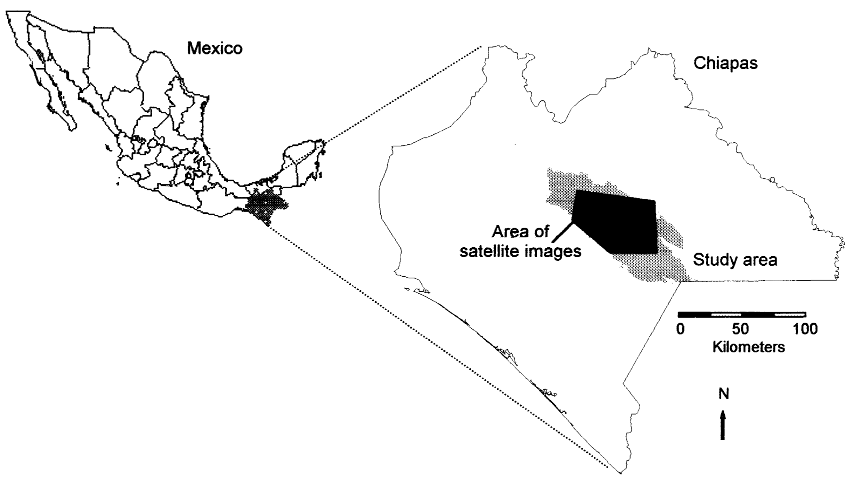

Fig. 1. Location of the study area.

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.

2.3. 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 per US \$ 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 are 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 one head of cattle per ha, and a reproduction rate of 0.6 calves per year (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.

2.4. 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

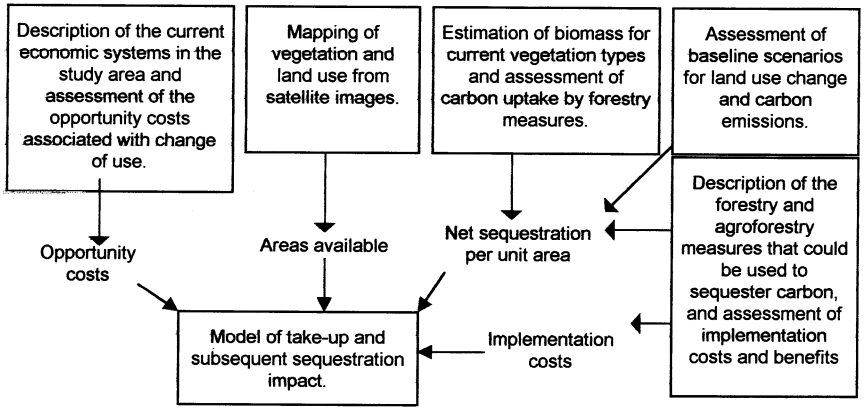

Fig. 2. Outline of the information flow to calculate the sequestration potential of an incentive/service payment-based forestry program.

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:

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

where: C_{acc}, long-term average accumulation of C of the alternative system; C_i, C, density of the alternative system in year i ($i = 1–100$); 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.

2.5. 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 (Fig. 2). Cost-benefit flows were discounted to present value to provide an estimate of the net present cost per ha (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.

3. Results

3.1. 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, Fig. 1), we found that during the late 1970s and early 1980s the total C stocks were depleted at a rate of approx. 1.7% per year. This rate decreased to

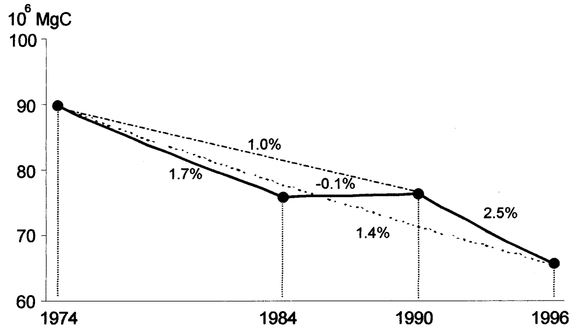

Fig. 3. Historical Carbon depletion (in 10^6 MgC and % annual change) in a sub-area of 308,000 ha, based upon data from Landsat MSS (1974, 1984, 1990) and Landsat TM (1996) images.

around −0.1% per year during the late 1980s, and then increased again to about 2.5% per year in the 1990s (Fig. 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% per year, 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 (Fig. 3).

3.2. 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 US \$186 ha^{-1} for oak and montane forests to US \$217.5 ha^{-1} for open pine forests (Table 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 US \$0 per year for the lowest quartile of open and

Table 1
Costs and benefits of the management options

Land use /land cover Types	Establishment including labor (US \$)	Operational and maintenance costs including project monitoring			Timber harvest 100 year cycle (in m^3)
		Costs (US \$ per year)	Labor input (days ha^{-1} per year)	Total (US \$ ha^{-1} per year)	
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

Table 2
Annual opportunity costs (in US $ per year) 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 $ per year)			
	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

closed pine forests to US $130 per year for the highest quartile of oak and pine-oak forests (Table 2).

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 US $52 ha^{-1}, but in the third quartile this cost was estimated at US $325 ha^{-1}. The fourth quartile represents the most difficult communities, where establishment of communal management systems was considered unfeasible (Table 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 US $212 on agricultural land to US $286 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 US $36 to 49 $year^{-1}$. Maintenance of the management systems for the remaining individual land holdings varied between US $39 per year to convert pasture to US $103 per year to transfer thickets to forest plantations (Table 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 US $0 to 358.50 per year (Table 2).

To estimate opportunity costs for pasture conversion, we calculated that typical ranching systems have around one head of cattle per ha, and a reproduction rate of about 0.6 calves per year. The annual weight gain per calf is around 100 kg, at a value of approx. 8 pesos kg^{-1} on the hoof. The rent foregone per hectare was therefore estimated to vary from US $39 to 152 per year (Table

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 and 305 m^3 ha^{-1} for systems replacing thickets (Table 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 15–25 years after tree planting.

The overall average NPC of viable land management options for C sequestration was less than US \$15 MgC^{-1} (Fig. 4). Forest management options represent the lowest costs, excluding the 4th quartile, where socio-political constraints are insurmountable (Table 2). They can be implemented at less than US \$15 MgC^{-1} for the first three quartiles (Fig. 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 (Fig. 4). The early product revenues from closed forest management offset much of the opportunity and implementation costs, giving a total cost of between US \$2 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 1).

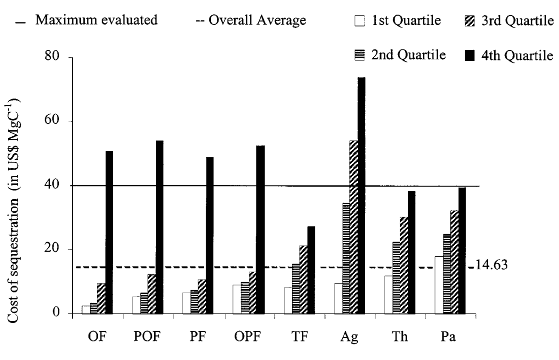

Fig. 4. Costs of carbon sequestration in US \$ MgC^{-1} for the four quartiles of the land use/land cover classes. OF, oak and montane forest; POF, pine–oak forest; PF, pine forest; OPF, open pine forest; TF, tree fallow; Ag, agriculture; Th, thicket; Pa, pasture.

3.3. 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 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 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.

3.4. Model outputs and sensitivity analysis

Under all baseline scenarios, the supply of sequestration was expected to be negligible with incentives below or equal to US \$5 MgC^{-1}, but rose sharply when increasing the incentives from US \$5 to 15 MgC^{-1}. According to Fankhauser (1997), projects that pass a cost-benefit test within the range of US \$5–20 MgC^{-1} are worth undertaking. Within this cost range, forestry and agroforestry measures in our study area could mitigate from 1 to 42×10^6 MgC, with a maximum eco-

Table 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

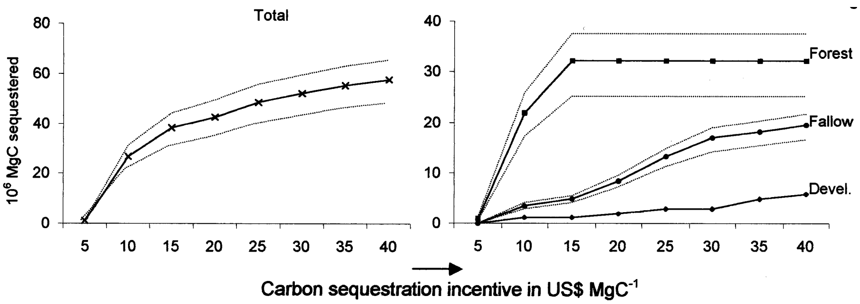

Fig. 5. Predicted carbon sequestration supply curves (in 10^6 MgC) for total, forest, fallow and development (agriculture + pasture) management options, based on low, medium and high baseline assumptions.

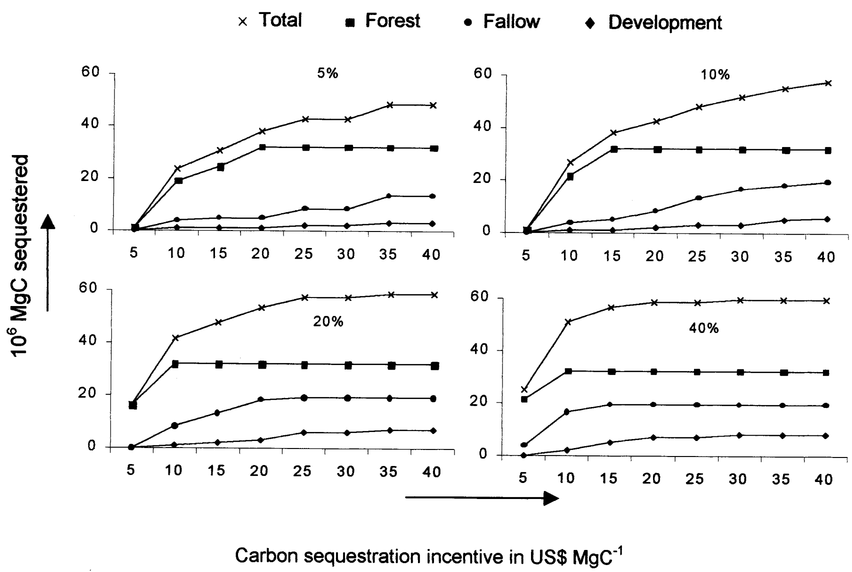

Fig. 6. Predicted carbon sequestration supply curves (in 10^6 MgC) for total, forest, fallow and development (Agriculture + Pasture) management options using 5, 10 (default), 20 and 40% discount rate.

nomic supply of carbon sequestration of around 55×10^6 MgC (Fig. 5) at US \$40 MgC^{-1}. The maximum supply through the management of communal forests was reached at US \$15 MgC^{-1}, the highest supply response from the improvement of fallow vegetation occurred between US \$15 and 30 MgC^{-1}. The response for agriculture and pasture was expected to rise slowly along the whole incentive gradient (Fig. 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 5 years. At low discount rates relatively low levels of sequestration supply were predicted for fallow and development options, where opportunity costs are relatively high (Fig. 6 and Table 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 US \$20 to 25 MgC^{-1} (Fig. 6).

4. 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 1980s. Experiences to date with the Scolel Té Pilot Project also indicate that farmers are generally eager to enter such a 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 farmers' 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.

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 demand 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 3, Fig. 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 et al., 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.

5. 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 US $5 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 cost and design requirements of large-scale schemes.

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