

EFFECTIVENESS AND DISTRIBUTIONAL IMPACTS OF PAYMENTS FOR REDUCED CARBON EMISSIONS FROM DEFORESTATION

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With 7 figures and 4 tables

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Summary: This paper analyses the effectiveness and distributional effects of payments to avoid tropical deforestation. As a first aspect, we investigated whether or not expected payments for avoided deforestation would be acceptable for tropical farmers in Southern Ecuador, with the study area located directly adjacent to the Podocarpus National Park. Second, we explored possible distributional effects resulting from voluntary or mandatory remuneration schemes to avoid deforestation. Finally, a productive sustainable land use was conceptualised to be combined with payments for avoided deforestation to avoid leakage (i.e. deforestation processes elsewhere when avoided at a given farm). Farm level land use scenarios with (“business as usual”) and without deforestation (“conservation strategy”) were compared. Compensation per Mg Carbon (C) that is not emitted into the atmosphere under the “conservation strategy” was derived to achieve a monetary land net present value (NPV, sum of discounted future net revenues) equal to the NPV obtained under “business as usual”. Avoided carbon emissions were computed from above ground C in tropical forests of the project area and supplemented by information on soil carbon from another study. Economic data for cattle pasturing were obtained from a farm survey (130 households) to investigate distributional effects. To derive sustainable land use concepts, a risk sensitive bioeconomic farm model was used that considered effects of risk compensation when combining pasture with reforestation of abandoned farm lands and selective logging of natural forests. The results showed that only a few farmers (20 out of 130) would possibly accept a compensation price of US\$ 10 per Mg avoided C emission, a C-compensation that is believed by other authors to reduce deforestation by 65%. Rather a compensation of around US\$ 25 per Mg C was necessary to address compensation requirements of farmers who hold 50% of the tropical forest area in our study. The implementation of a voluntary remuneration scheme for avoided deforestation would not introduce systematic distributional effects (such as that only the biggest farmers would benefit from compensation), while a mandatory and enforced ban on deforestation coupled with a “fair” compensation payment equal to mean compensation requirements may lead to undesirable effects for many farmers. Finally, we demonstrate a mixed sustainable land use concept that depended on cheap credits for reforestation of abandoned pasture lands. This concept was able to stop farm level deforestation and to enlarge the economic value of farms through various combined land use options (agricultural and forestry options). The combination of land uses led to risk compensatory effects and a more efficient land use by reintegrating unproductive abandoned areas back into the economical process. In our conclusion a combination of payments for avoided deforestation along with productive land use concepts provided a viable solution for tropical forest conservation.

Zusammenfassung: Wir analysieren Effektivität und Verteilungseffekte von Kompensationszahlungen zur Vermeidung von Entwaldungsprozessen. Als ein erster Aspekt wurde untersucht, ob die zu erwartenden Zahlungen zur Vermeidung der Umwandlung von Tropenwald für Farmer in Südecuador akzeptabel wären; unser Untersuchungsgebiet liegt in direkter Nachbarschaft zum Podocarpus Nationalpark. In einem zweiten Schritt wurde getestet, welche Effekte aus einer auf freiwilliger Basis funktionierenden Kompensation bzw. aus einer Entschädigung in Kombination mit einem Entwaldungsverbot für die Einkommensverteilung entstehen könnten. Schließlich richtet sich der Blick auf ein nachhaltiges Konzept zur Landnutzung, welches mit Kompensationszahlungen kombiniert werden könnte, um so die Attraktivität der Farm für ihre Besitzer zu erhalten und eine Verlagerung der Entwaldung auf andere Gebiete zu vermeiden. Zunächst wurden zwei Landnutzungsszenarien auf der Farmebene, eines mit Entwaldung („Herkömmliche Landnutzung“) und eines ohne Entwaldung („Schutzvariante“), miteinander verglichen. Auf dieser Basis wurde die notwendige Kompensationszahlung pro nicht emittiertem Mg Kohlenstoff (C) abgeleitet, um denselben Barwert an zukünftigen finanziellen Überschüssen wie im Rahmen der Strategie „Herkömmliche Landnutzung“ zu erzielen. Die vermeidbaren C-Emissionen wurden anhand der in den Naturwäldern des Untersuchungsgebietes vorhandenen oberirdischen Kohlenstoffvorräte und von Angaben zum Bodenkohlenstoff aus einer anderen Studie kalkuliert. Die ökonomischen Daten zur Weidewirtschaft stammen aus Erhebungen in 130 Farmhaushalten des Untersuchungsgebietes. Basierend auf einem das Risiko der Landnutzung berücksichtigenden bioökonomischen Modell wurde ein nachhaltiges Landnutzungskonzept abgeleitet, welches die Effekte einer Risikomin-

derung bei Kombination unterschiedlicher Landnutzungsoptionen (Weidewirtschaft, Aufforstung aufgelassener Weiden, nachhaltige Nutzung im Naturwald) berücksichtigt. Die Ergebnisse zeigten, dass eine Kompensationszahlung von 10 US\$ pro Mg vermiedene C-Emission – ein C-Preis durch den man hofft, die Entwaldung um 65% zu reduzieren – nur für sehr wenige Farmer möglicherweise interessant wäre (20 von 130 insgesamt erfassten Haushalten). Dagegen wäre ein Kompensationspreis von 25 US\$ pro Mg C notwendig, um die Einkommensverluste derjenigen Haushalte zumindest auszugleichen, die 50% der betrachteten Tropenwaldfläche besitzen. Die Einführung von Entschädigungsprogrammen, an denen man freiwillig teilnehmen könnte, würde wahrscheinlich zu keinen systematischen Effekten hinsichtlich der Einkommensverteilung führen. Ein verpflichtendes Entschädigungsprogramm bei erzwungener Vermeidung von Entwaldung, verbunden mit einer „fairen“ Entschädigung in der Höhe der durchschnittlichen Opportunitätskosten, würde allerdings für viele Haushalte zu unerwünschten Effekten führen, da deren Opportunitätskosten nicht abgedeckt würden. Schließlich stellen wir ein nachhaltiges Landnutzungssystem dar, das aus drei Landnutzungsformen besteht, aber von preiswerten Krediten abhängt. Dieses Landnutzungssystem ist in der Lage, die Entwaldung auf der Farmebene zu stoppen und gleichzeitig den ökonomischen Wert der Beispielfarm zu heben, indem landwirtschaftliche und forstliche Landnutzungsoptionen kombiniert wurden. Die Landnutzungskombination führte zu Risikominderungen und einer effizienteren Landnutzung, weil unproduktive Ödlandflächen durch Aufforstung wieder in den Produktionsprozess integriert wurden. Wir folgern, dass eine Kombination von Kompensationszahlungen für vermiedene Entwaldung mit nachhaltigen und produktiven Landnutzungskonzepten einen Erfolg versprechenden Lösungsansatz für die Landnutzungsproblematik in den Tropen darstellen.

Keywords: Tropical forest conservation, conservation payments, carbon storage, sustainable land use

1 Introduction

The Kyoto Protocol accepts the sequestration of carbon through the biosphere as an offset mechanism to compensate for emissions from burning fossil fuels (UNFCCC 1997). However, concerns about sovereignty and methodology have led to the exclusion of forest conservation from the Clean Development Mechanism, which is limited to afforestation and reforestation (e.g., GULLISON et al. 2007). Although the saving of forests as a global warming countermeasure was initially controversially discussed (e.g., FEARNSIDE 2001), the awareness has grown that reduction targets for CO₂ emissions will hardly be achievable without solving the deforestation problem, which accounts for around 20% of the current CO₂ emissions (STERN 2009).

Since STERN (2006) identified that curbing deforestation is a highly cost-effective way of reducing GHG emissions, climate policy makers have confirmed the need for action to reduce emissions from deforestation and forest degradation, named REDD, at the UNFCCC (United Nations Framework Convention on Climate Change) meeting in Bali 2007 (UNFCCC 2008). After the first Kyoto commitment period (2008–2012), Rainforest Nations might be paid for reduced emissions from deforestation and forest degradation through international carbon markets or funds fed by voluntary international payments (MALHI et al. 2008; ELIASCH 2008). While several models for the implementation of REDD have already been proposed (e.g., BELASSEN and GITZ 2008), the details on the so called “Global Deal”, including the above mentioned issues of REDD, will

be negotiated at the 15th Conference of the UNFCCC parties in Copenhagen (December 2009).

Various opportunities are available to achieve reduced deforestation. REDD mechanisms can be implemented as voluntary transactions between sellers and buyers of emission credits generated by avoided deforestation and would then have the character of payments for ecosystem services (PES, see WUNDER 2007 for details). Alternatives are mandatory and enforced deforestation bans (e.g., protected areas), environmental taxes or integrated conservation and development projects (ENGEL et al. 2008), improvement of property rights, better market access, change of land use intensive consumption systems or reduced impact logging. However, although much has been said about opportunities and limitations of REDD mechanisms (e.g., BELASSEN and GITZ 2008; HOLMGREN 2008), little is known about the important aspects of effectiveness and the distributional implications of payments for reduced carbon emissions from deforestation (see ENGEL et al. 2008 for a discussion of both aspects in designing PES).

High opportunity costs of forest conservation (i.e. the economic value of the usual, deforestation based land use) may undermine the effectiveness of REDD mechanisms, as BUTLER et al. (2009) have shown. More detailed information on the opportunity costs of tropical forest conservation and their variation, when evaluated from the perspective of individual local actors, would be helpful in making a statement on the effectiveness of payment mechanisms. Moreover, there exist severe problems that may weaken the effectiveness of carbon payments, such as leakage (i.e. the displacement of environ-

mentally damaging deforestation activities to areas outside the geographical zone of payment intervention, as described by ENGEL et al. 2008). When farmers are compensated for tropical conservation they cannot replace their existing and often degrading pastures with new areas. The financial flows will thus be on the decline. In such a situation it is likely that farmers will move to other places to establish new farmlands elsewhere. As a consequence, compensation strategies have to be accompanied by productive but non-destructive land use concepts to avoid leakage effects (KNOKE et al. 2008a).

It is also important to avoid undesirable social effects when designing payment schemes. This is not automatically assured, because PES schemes are primarily designed to improve natural resource use and not for poverty reduction (ENGEL et al. 2008). Using national or even regional averages of per hectare income and farm size may be misleading due to scope and scale effects in smallholder agriculture, as well as in individual differences in productivity. Particularly for tropical low-income countries, an underestimation of compensation payments results in immediate ethical problems and practical implementation issues: Smallholders may be pushed into abject poverty, and/or resist conservation restriction.

To provide case study material that at least partly remedies existing research gaps, our study investigates the following questions for a tropical mountain forest landscape in Southern Ecuador:

- Are expected carbon compensation prices high enough to convince tropical farmers for avoiding deforestation?
- How does required financial compensation vary between differing farm holders?
- How could financial compensation for avoided deforestation be combined with productive land use options to keep the farmers on their established farmlands and thus avoid leakage?

This multi author paper is a joint research of several working groups (WG) who are concerned with economical, socio-economical and ecological questions of tropical land use. We carried out the research as part of the German Research Unit FOR 816, funded by the “Deutsche Forschungsgemeinschaft”. The agricultural land use data were recorded by “WG Barkmann” (Göttingen) while the forest data originated from “WG Weber et al.”, financially valued by “WG Knoke” (both Freising-Weiherstephan). “WG Pohle” (Erlangen) contributed local land use expertise for an appropriate description and discussion of data and results. All data and experiences were combined in a risk sensitive bioeconomic land

use model developed by “WG Knoke” (KNOKE et al. 2009).

2 Study area

Ecuador suffers one of the highest deforestation rates in Latin America (FAO 2007). Since January 2009 the national programme called “Programa Socio Bosque” is being implemented here. This programme provides financial incentives for the voluntary conservation of native forests (up to US\$ 30 per hectare per year). The objective of the programme, financed by the state of Ecuador as a central component of a national REDD proposal, is to preserve about 4 million hectares of tropical forests in the next 20 years. The programme “Socio Bosque” exemplifies the great interest of the Ecuadorian people to save their tropical forests.

We investigated the land use around the Podocarpus National Park, South Ecuador. The Podocarpus National Park is located in the provinces of Loja and Zamora-Chinchipec (Fig. 1), with its northern edge demarcated by the valleys of the San Francisco and Zamora Rivers. The National Park is the core zone of the recently established Podocarpus-El Cóndor UNESCO Biosphere Reserve. We focused on the northern and north-eastern buffer and transition zones (the ‘project area’) where patches of intact natural forests are found close to currently deforested sites, and extensive pastures used for cattle grazing (BECK 2008).

Through the establishment of the Podocarpus National Park in 1982, the scarcity of land available for productive use has increased and several threats to the Podocarpus National Park are reported, including illegal colonisation and pseudo-colonisation (RAHBECK et al. 1995; KEATING 1997). The Podocarpus National Park has an area of ~146,000 hectares. It includes cloud forests, high-altitude grasslands, and a series of small Andean lakes. The area is not only one of the global ‘hotspots’ of biodiversity (BRUMMITT and LUGHADHA 2003) but it is also critical for the provision of fresh water for more than one million people in the surroundings of the park.

The natural ecosystem under investigation is represented by the forests of Reserva Biológica San Francisco (RBSF) located in the Cordillera Real, an eastern range of the Southern Ecuadorian Andes (BECK et al. 2008a). The ecosystem extends from 1800 m to 3160 m a.s.l., while for our investigation the part of the tropical lower montane forests was

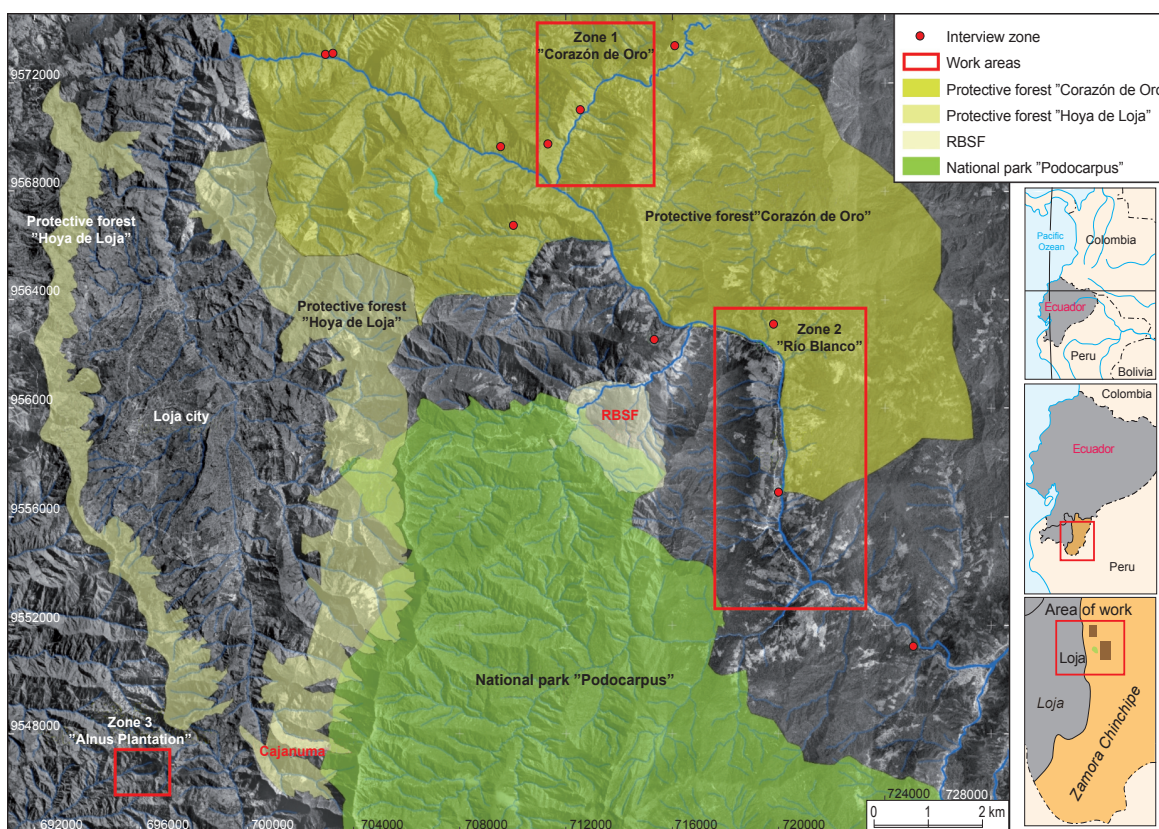


Fig. 1: Map of the project area (mapping: Baltazar Calvas)

most relevant, covering an altitudinal gradient from 1950 m to 2100 m a.s.l.

Outside the still intact natural forests, the landscape north of the Podocarpus National Park is shaped by the activities of small-holder farmers (POHLE 2004). As reported by TORRACCHI et al. (without year), the cover of tropical forests declined by around 30 percentage points over 26 years around the Podocarpus National Park. That implies a loss of 90,927 ha of native forests with an average annual net deforestation rate of 1.16%. These substantial forest losses underline the imbalance between deforestation on the one hand, and reforestation as well as natural succession towards secondary forests on the other hand, leading to high pressure on the remaining forests. Given the presence of the Podocarpus National Park in the considered landscape and the fact that southern Ecuador is known to be a main hotspot of biodiversity, sustainable land use options are urgently needed, especially in a place such as this. The consideration of tropical forest carbon storage as ES, rewarded by carbon payments, might help to achieve this objective.

3 Methodology and data

3.1 Expected effectiveness of carbon payments

3.1.1 The baseline: a scenario for “business as usual”

Land use

To obtain information about the carbon emissions to be expected on the farm level, we simulated the land development and deforestation process for three farm types (denoted A, B, C, Tab.1) chosen as characteristic members of a surveyed population of totally 130 farms (Tab. 2, Fig. 2). These farms were selected as the 5-, 50- and 95-percentiles from the data set sorted according to cattle ranching benefits they contribute to the livelihood of farmers (denoted as agricultural revenues in table 1 and classified as low, moderate, high). We started with the given endowment of these farms (Tab. 1) and projected scenarios for their land development by means of a model approach adopted from KNOKE et al. (2009) shown in figure 3a-f. The following

Table 1: Endowment of the example farms that represent 5- (Farm type A), 50- (Farm type B) and 95-percentiles (Farm type C) of the data set when sorted by agricultural revenues (example farms selected from 130 surveyed farms)

| Farm type | Agricultural revenues (US\$ per year) | Area (ha) | | 5%-Net present value (40 years US\$) | Net revenues pasture (US\$ ha ⁻¹ year ⁻¹) |
|------------------------------|---------------------------------------|-----------|----------------|--------------------------------------|------------------------------------------------------------------|
| | | Pasture | Natural forest | | |
| A (5-percentile) | 329 | 1.8 | 14.0 | 2,902 | 100 |
| B (50-percentile, median) | 2,203 | 18.5 | 161.5 | 29,550 | 99 |
| C (95-percentile) | 16,065 | 37.5 | 64.2 | 220,433 | 386 |

Table 2: Statistical information on the total data set containing 130 farm surveys

| | Area (hectare) | | Agricultural revenues (US\$ per year) | Gross-income (US\$ ha ⁻¹ year ⁻¹) | Pasture net revenues (US\$ ha ⁻¹ year ⁻¹) |
|---------------------------|----------------|----------------|---------------------------------------|----------------------------------------------------------|------------------------------------------------------------------|
| | Pasture | Natural forest | | | |
| Median | 12.5 | 23.2 | 2,203 | 190.6 | 137.2 |
| Mean | 18.3 | 40.2 | 4,008 | 227.9 | 170.5 |
| Standard deviation | 17.1 | 44.6 | 4,891 | 158.5 | 140.8 |

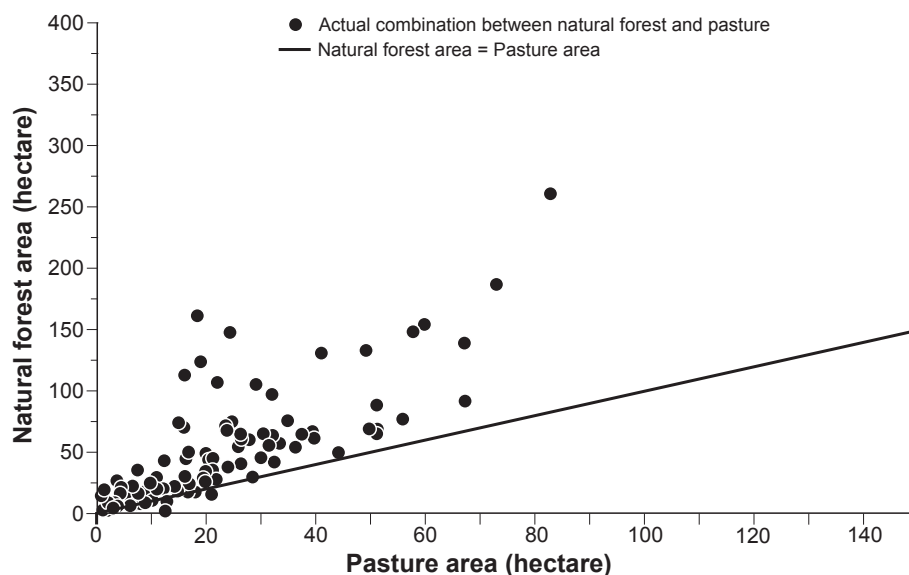


Fig. 2: Combination between natural forest and pasture area

assumptions were made to obtain the possible scenarios regarding future land use, deforestation and the related financial flows: the major source of agricultural income for the farmers in the study area is a cattle ranching on pastures. The land use is thus dominated by pasture management, with pastures being mainly cultivated with mequerón (*Setaria sphacellata*). Reasons for the ongoing deforestation

include degrading pastures that must be replaced by new pasture areas and farmers who wish to improve their livelihoods (POHLE et al. 2009). In our study, we worked with continuous pasture degradation and the assumption that degraded farmland has to be replaced after the accumulation of degraded land during a 10-year time step (details see below).

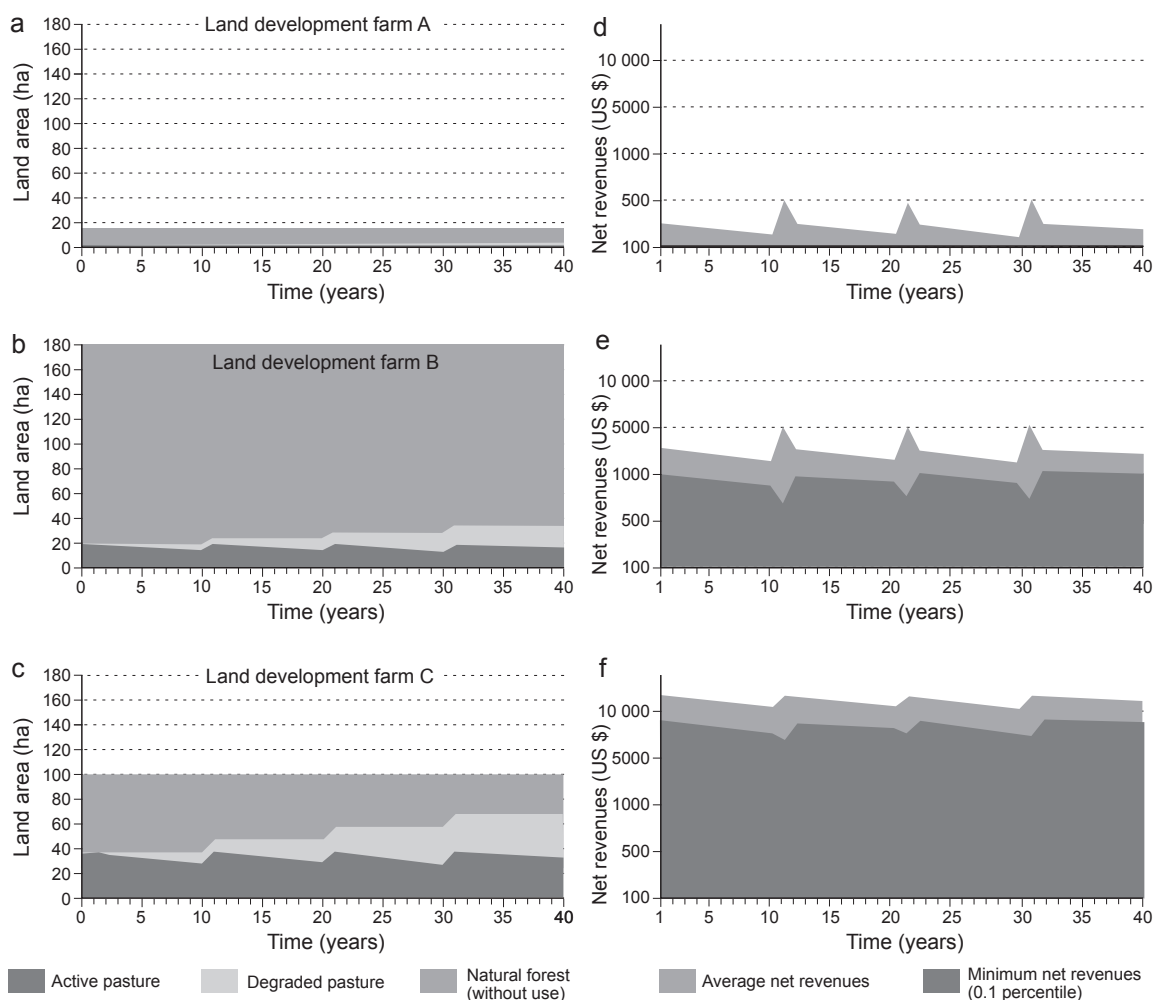


Fig. 3: Land use development (left) and net revenues (right) for the example farm types A, B, C, selected as 5-, 50-, and 95-percentiles according to the agricultural revenues from farm management

Various types of pasture exist in the area under investigation, differing, for example, in the presence of trees, deadwood or bracken fern (*Pteridium arachnoideum* (Kaulf.) Maxon) (PAULSCH et al. 2001). Working at the deforestation frontier, we considered only pastures without trees for our modelling and their expected financial flows after timber extraction and slash and burn (BECK et al. 2008a). We assumed that the productivity of a newly established pasture starts to decline, for example by bracken fern that can overtop pasture grasses starting after 10 years of pasture usage (BECK et al. 2008b). For already existing pastures, we assumed that degradation is already in progress.

Productivity decline was not modelled deterministically but as a pasture abandonment probability (see below). The duration of pasture management on a given piece of land may vary greatly. Many studies

assume an average utilisation time of 15 years until the pasture land becomes infertile and/or invaded by pasture weeds so that further management becomes unattractive (WUNDER 2000; CARPENTIER et al. 2000; BENITEZ et al. 2006). BECK et al. (2008b) even reported pasture utilisation periods of hardly more than 10 years. However, locally even very old active pastures exist (> 40 years). Thus, we modelled continuous degradation up to utilisation periods of 40 years (from this year onwards no further degradation was assumed). Beginning with pasture utilisation year 11, a linear decline of the productive pasture area was modelled so that at year 40 only 25% of the original pasture area is still productive. In order to compensate for losses of active pasture area the opportunity to establish new pastures at years 11, 21 and 31 was included in our modelling. The decline of productive pasture area thus implied a conditional yearly prob-

ability of abandonment between 0.03 at the beginning and 0.07 at the end of the considered period.

Carbon emissions

The considerations above implied that we have deforestation processes and the connected carbon emissions in the years 11, 21 and 31 (Fig. 4). The carbon stored in the tropical forests of the study area

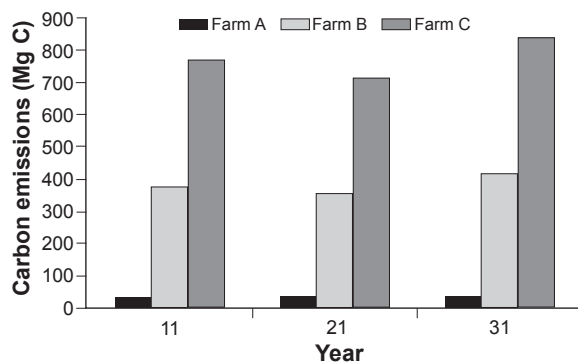


Fig. 4: Derived carbon emissions from deforestation for farms A, B, and C

was calculated by MEDINA (2008) who used regression equations parameterized for various tree species groups by NENNINGER (2006). The tree’s biomass was estimated by means of their dbh (diameter at breast, i.e.1.3 m, height) as an independent variable. The total above ground biomass of standing trees could then be estimated based on stem frequency distributions as measured by GÜNTER et al. (2008). MEDINA (2008) carried out investigations of the carbon that was stored in the necromass (Tab. 3). However, information on soil carbon was not recorded. To bridge this gap we adopted data published by RHOADES et al. (2000) for Ecuadorian tropical forests and pastures. These authors stated that under Ecuadorian pasture we have a reduction in soil carbon of 12 Mg C ha⁻¹ compared to a tropical forest. The carbon stored in the biomass of pasture grasses was ignored in our

calculation. To obtain the potential carbon value of tropical forests we used the forest type with the highest carbon storage for our investigation (table 3, for a justification see below). This means that we assumed carbon emissions of 74 Mg C ha⁻¹ when conversion from tropical forests into pasture lands takes place.

Financial flows

The coefficients to derive the financial flows are summarized in table 4. To evaluate financial consequences of tropical forest conversion into pasture, an estimation was needed of the financial flows from initial timber extraction combined with slash and burn, followed by the establishment of the pasture grasses. To obtain this information we concentrated on comparatively productive tropical forest land as potential conversion area, located in small valleys (“experimental area # 5” in table 3), due to its comparatively greater probability of being used for agricultural land expansion. Estimating the net revenues for the initial year of conversion required the calculation of the financial timber value of the forest to be converted as well as the logging and pasture establishment costs. GÜNTER et al. (2008) reported results from a natural forest experiment within the RBSF area and provided further (though not yet published) data, from which this information could be derived. The distribution of tree diameters (dbh) was typical for uneven-aged, highly structured forests. About 140 different tree species could be distinguished with mean densities of 42.9 trees ha⁻¹ that have a dbh above 40 cms. Only these large trees were considered merchantable. Given the measured length of the branchless bole (on average about 8 meters) we obtained the potentially merchantable timber volume of 42.6 m³ ha⁻¹. Expected losses due to poor timber quality, bark and other felling losses, lead us to reduce the realistically merchantable timber volume by means of multiplying by the factor 0.5 (LEISCHNER 2000), thus obtaining 21.3 m³ ha⁻¹ of timber that could be sold at the timber market. Given an average timber price (Departamento

Table 3: Biomass and carbon contents in tropical forests of the project area

| Experimental area # | 3 | 2 | 5 | Source |
|------------------------------------------------------------------------------------|-----|-----|-----|---------------------|
| Area (ha) | 4.2 | 4.9 | 3.4 | |
| Tree density (dbh>20 cms, frequency per ha) | 166 | 208 | 318 | |
| Tree biomass (Mg ha ⁻¹) | 29 | 72 | 118 | MEDINA 2008 |
| Necromass (Mg ha ⁻¹) | 5 | 7 | 5 | |
| Stored carbon (Mg ha ⁻¹) | 17 | 39 | 62 | |
| Soil carbon: Difference between tropical forest and pasture (Mg ha ⁻¹) | | 12 | | RHOADES et al. 2000 |

Forestral 2005) of US\$ 68.5 m⁻³, we computed gross revenues of US\$ 1,462 ha⁻¹. We calculated 2.62 working days (CARPENTIER et al. 2000) to harvest and provide one cubic meter, with costs of US\$ 10 day⁻¹. Hence, the logging amounted to US\$ 534 ha⁻¹. The establishment of pasture grasses was computed at US\$ 550 ha⁻¹. Overall conversion costs of US\$ 1,084 ha⁻¹ are quite reasonable, given that tropical forest conversion is a very hard and intensive work, requiring 80 to 100 days of labour (PICHÓN 1996). Without costs for pasture plants and other materials, this labour would cost US\$ 800-1,000 ha⁻¹, which largely confirms our results. Thus we expect the average net revenues of US\$ 378 ha⁻¹ from converting a tropical forest into pasture land for the first year.

The yearly net revenues that follow the conversion from tropical forest into pasture lands were taken from the farm survey (see Tab. 1, survey described below).

3.1.2 Financial consideration

We used the net present value (NPV) of all projected future net revenues as a criterion to derive possible financial compensations to avoid deforestation. The NPV is quantified by means of the sum of all discounted future net revenues from land management (Eq. 1):

$$V_0 = \sum_{t=0}^T r_t q^{-t}$$

with V_0 being the NPV, t the considered point in time, T the period of consideration (40 years in our study), r_t the net revenues at a given point in time and q the discount factor ($q=1+i$, with i being the decimal interest rate).

The NPV is seen as an appropriate measure to anticipate the economic value of land (e.g., WUNDER 2000), as it expresses the expected economic future benefits from land use. The choice of an adequate discount rate is a crucial point in obtaining the NPV of land management (PEARCE et al. 2003). When analysing sustainable land use within natural systems, achieving internal interest rates above 5% is hardly possible (PEARCE et al. 2003). We thus decided to use this moderate interest of 5%, which was also applied in other studies (WUNDER 2000; BENITEZ et al. 2006) to discount net revenues from Ecuadorian pastures. We also tested the impact of a 10% interest, as for example applied by BUTLER et al. (2009), on the required compensation and we will also discuss the resulting effect later.

First, we computed the NPV of the deforestation based land use (baseline scenario). Second, the NPV of a land use scenario without deforestation was calculated (conservation scenario), which is necessarily lower compared to the baseline scenario as it does not consider the replacement of the degraded pasture areas by newly established, productive pastures. Finally, we computed carbon prices which would make the conservation scenario equally valuable (in financial units) compared to the baseline scenario. For this we assumed that the avoided deforestation was paid for per Mg of C that is not emitted into the atmosphere.

3.2 Distributional effects and differences in required compensation

To make a statement on the expected effectiveness of given compensation payments, and to simulate their distributional effects, it is necessary to estimate which households compensations may be acceptable, and how acceptance is related to total farming revenues. We used information obtained from the survey of 130 farm households to approach this problem.

Data on the households were obtained by structured interviews in 2008. By randomised population-proportional sampling, 16 of a total of 38 settlements (*barrios*) north and north-east of Podocarpus National Park were selected (settlements indicated as “interview zones” in figure 1). In each settlement, the required number of households was contacted via snowball sampling¹⁾. The questionnaire included questions on farming capital (active and degraded pastures, arable land, forests, operating capital), production techniques (depreciation, cash and labour inputs, animal or pasture varieties), production outcomes (cash revenues, subsistence consumption), and socio-demographic household characteristics. The interviews were carried out by native Ecuadorians and checked for plausibility. All single landholdings were geo-referenced via orthophotos, and their land use subjected to independent field proofing. As a consequence of this process, five of the initially surveyed 135 farms were excluded from the survey due to false information. The farms were sorted according to their economic revenues from cattle ranching (including cash revenue and subsist-

¹⁾ Because of the sensitivity of the household and farming data to be disclosed during the interview, a fully randomized sampling scheme could not be administered.

ence consumption of milk, curd and cattle valued by market price).

To get an overview how much natural area could be conserved with given conservation payments that result from specific carbon compensation prices, we sorted all farm households according to their required carbon compensation, which was strongly dependent upon cattle pasturing net revenues per hectare (Fig. 5). We then summed up their natural forest area and determined the accumulated area up to the last farm in which the compensation requirement would just be covered by the conservation payments.

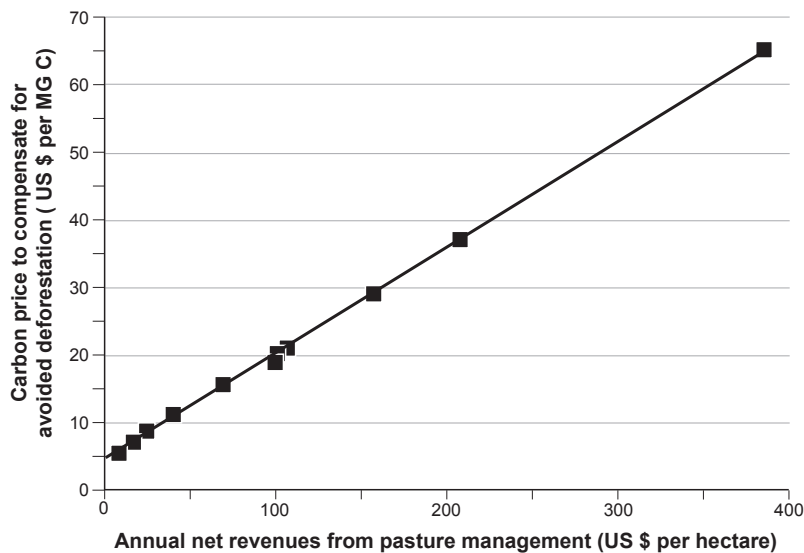


Fig. 5: Linear relationship between required compensation per Mg C and annual net revenues per hectare from pasture management

For the analysis of distributional effects (for example that only large farmers would benefit from compensation), we plotted the individual compensation requirement versus farm size indicated by pasture area. We then compared the farms that were predicted to be financially willing to be included in a US\$ 10 per Mg C (a carbon price expected for example by PEARCE and PEARCE 2001) programme with those farms having a higher compensation requirement (heteroscedastic, two-sided t-test). An US\$ 10 per Mg C (about US\$ 2.7 per Mg CO₂) programme corresponds not only with the expectation of PEARCE and PEARCE (2001) but also roughly with the average carbon compensation of US\$2.8 per Mg CO₂, which is believed to reduce deforestation-based emissions by 65% (ELIASCH 2008).

Three farm types (A, B, C) of the 5-, the 50- and the 95-percentiles of economic farm benefit were selected for detailed analysis. By comparison to the first, the third and the last quintile, we assured that

the selected farms did not represent gravely atypical cases (data not shown).

3.3 Productive but non-destructive land use concepts

To make it attractive for farmers to abstain from deforestation, we need to develop concepts that secure land productivity (KNOKE et al. 2008a). We have already demonstrated the conceptual framework of a bioeconomic modeling to support this objective

(KNOKE et al. 2009). Here, we describe a core concept to establish a more diversified land use to accompany carbon compensation schemes and capable of stabilizing today's unsustainable and thus unstable tropical land uses.

Our productive land use model was based on biological productivity, net revenues, uncertainties, and risk compensation for diversified land uses including: i) "pasture", ii) "reforestation" of 'wastelands' (i.e. degraded pastures), and iii) "selective logging" in natural forests. The goal was to obtain an economical multi use land management system that fulfils financial subsistence demands while minimising or avoiding deforestation, when compared to a single use pasturing system that considered a temporal scope of 40 years (as already mentioned above). The mixed land use system integrated risk compensation via product diversification and the building up of new biological resources through reforestation. The assumptions made were conservative and followed

the concept of ‘precaution’ and sustainable management principles. This meant, for example, that selective logging was designed to maintain the actual structure and standing timber stock of the natural forest. All timber harvests had thus to be compensated by the current timber increment. Due to these assumptions we viewed our low impact selective logging carbon neutral.

Addressing farmers’ objectives and sustainability requirements

The farm modelling was fed by several input coefficients (Tab. 4). A first component defines the farmer’s objectives and sustainability requirements. A main objective of the mixed land use modelling was to secure the achievable NPV of land with great probability, to address the farmers’ risk-avoiding attitude (PICHÓN 1996). We thus considered various biological and financial sources of uncertainty and assumed normal distribution of uncertain net revenues. In contrast to the classical maximization of NPV, we maximised the 10th percentile of the simulated NPV distribution (called land value at risk).

Reforestation of degraded pasture lands served to implement a central requirement of sustainability, where exploited natural resources must be replaced by other natural resources (DALY 1990). As a biological requirement for the mixed land use scenario we thus modelled for every area of degraded pasture that accumulated within 10 years, its reforestation with the native tree species Andean alder (*Alnus acuminata*, production period 20 years). Reforestation assumed a financial credit at a real interest of 5% for the plantation investment (see discussion). Uncertainty of reforestation success (e.g., fire damage may occur) was modelled and the probability with which credit repayment should be possible was set to 0.9. Thinning revenues, which take place before harvesting the final crops and thus deliver earlier net revenues, were limited by the constraint of credit repayment after 20 years. Thinnings could only be carried out if the stumpage value of the final crops were not reduced below the amount required for credit repayment under uncertainty. If the stumpage value of the final crops was greater than the amount required for credit repayment their harvest could also be postponed to compensate declining revenues from pastures.

Uncertainty

Besides the component “objectives and sustainability requirements,” modelling uncertainty was fundamental for our bioeconomic approach. Three

biophysical risks were included in our model: i) pasture degradation, ii) fluctuation of milk productivity and sustainable harvest, and iii) probability of fire damage as a threat to reforestation.

Biophysical risks for all land use options were estimated and combined with market price uncertainty in order to derive the standard deviation (SD) of net revenues. Once the SD for land use options is calculated, the financial risk of land use strategies may be evaluated. SD of net revenues is crucial as it is a well established measure to quantify financial risk (HIRSHLEIFER and RILEY 2002). It is needed to estimate the 10th percentile of land NPVs. Mixing land use options in various fractions may directly affect their combined SD and risk, thus consequently the minimum net revenues. If farmers would mix two or more land use options, which show independent fluctuation of net revenues, they would greatly benefit from risk reduction. This means that one land use may generate unexpectedly great net revenues, while net revenues of the other option are less than expected and vice-versa. For example, if the milk price is down, the timber price may be high or at least moderate and if the timber price declines, the milk price may be acceptable (LÖNNSTEDT and SVENSSON 2000). These financial risk interdependencies were considered in our model to mirror the expected risk-reducing effects of diversification.

Financial coefficients of sustainable land use alternatives

Reforestation

Financial coefficients of land use formed the third component of our modelling. Reforestation of degraded pastures (“wastelands”) with native tree species was a key-activity in our approach. We used available artificial time series data on productivity to model future timber revenues (FEHSE et al. 2002), which was then adjusted according to our own measurements of an Andean alder plantation in the South of Loja. As generally recommended (DUNN et al. 1990), we used a production cycle of 20 years (rotation). Reforestation usually requires an investment of US\$ 800 ha⁻¹, but, to be cautious, we calculated using US\$ 1,000 ha⁻¹ since the survival probability of Andean alder was on average around 80% in a reforestation experiment (WEBER et al. 2008). Moreover, fire can cause damages that have to be compensated for, especially at very young ages (GRAU and VEULEN 2000). After harvesting the forest at year 20, a second rotation is possible without investment, because Andean alder resprouts vigorously after harvesting

Table 4: Coefficients used for the bioeconomic modelling (adopted from Knoke et al. preview online, with alterations, WG: working group)

| Land use option | Item | Coefficient | References | Uncertainty economic data (coefficient of variation) | Reference |
|-------------------|---------------------------------------------------------|-------------------------------------------------------------------------|------------------------------------------------------------------------------------------------------------|-----------------------------------------------------------------------------------------------------------------------------------------------------------------------------------------|-------------------------------------------------------------------------------------------------------------------------------------------------------------------------|
| Pasture | Merchantable timber volume from initial forest clearing | 21.3 m ³ ha ⁻¹ | Computed acc. to data “WG Weber et al.”, GÜNTER et al. 2008, LEISCHNER 2000 | 30% | GÜNTER et al. 2008 |
| | Timber price | US\$ 68.5 m ⁻³ | Departamento Forestal 2005 | 10% | KNOKE and WURM 2006 |
| | Establishment of grasses | US\$ 550 ha ⁻¹ | Calculation “WG Knoke” | 10% | Estimation “WG Knoke” and Pichón 1996 |
| | Logging costs | 2.62 days m ⁻³ x US\$ 10 day ⁻¹ | CARPENTIER et al. 2000 and estimation “WG Knoke” | 10% | Estimation “WG Knoke” and Pichón 1996 |
| | Net revenues from pasture | US\$ 99 ha ⁻¹ yr ⁻¹ | 50-percentile obtained from farm interviews “WG Barkmann” (when data set sorted by total pasture benefits) | 36%; abandonment possible from year 10 onwards, linearly increasing probability of being abandoned (P _a) from year 10 (P _a =0) to year 40 (P _a =0.75) | Probability of abandonment estimated sensu BECK et al. 2008b and observation of GÜNTER (unpublished), milk productivity and price uncertainty sensu BENITEZ et al. 2006 |
| Reforestation | Establishment costs | US\$ 1000 ha ⁻¹ | Calculation “WG Knoke” | Sapling survival probability 0.8 | WEBER et al. 2008 |
| | Stand growth | 10.5 m ³ ha ⁻¹ yr ⁻¹ (up till year 20) | FEHSE et al. 2002, WEBER et al. 2008; stand volume for age 8 adjusted with own measurements | 10%; survival probability 0.82 up till age 20 | FEHSE et al. 2002 and ROMAN CUESTA, fire probability deduced from MODIS-Terra 2000–2006 hotspots |
| | Stumpage price | 7 (year 10) to US\$ 25 m ⁻³ (year 20) | OLSCHEWSKI and BENITEZ 2005 and estimation “WG Knoke” | 10% | KNOKE and WURM 2006 |
| Selective logging | Forest structure, growth | 1.55% basal area and 2.6 mm yr ⁻¹ diameter growth | GÜNTER, unpubl. data, OESKER et al. 2008, GÜNTER et al. 2008. | 26% | OESKER et al. 2008 |
| | Sustainable, merchantable harvest | 0.75 m ³ ha ⁻¹ yr ⁻¹ | Computation based on unpublished data on forest growth & mortality, GÜNTER (unpubl.) | 30% | GÜNTER et al. 2008 |
| | Timber price | US\$ 68.5 m ⁻³ | Departamento Forestal 2005 | 10% | KNOKE and WURM 2006 |
| | Logging costs | 2.62 days m ⁻³ x US\$ 10 day ⁻¹ | CARPENTIER et al. 2000 | 20% | Estimation “WG Knoke” |
| All land uses | Discount rate Correlation coefficient net revenues | 5% | PEARCE et al. 2003 | k _{Pasture,Forestry} =0.0, k _{Reforestation,Selective logging} =0.8 | LÖNNSTEDT and SVENSSON 2000 and estimation “WG Knoke” |

(GRAU and VEBLEN 2000) and can thus be managed by means of a coppice system. First commercial thinnings were modelled from year 10 onwards (DUNN et al. 1990). Reforestation management was modelled as a flexible land use, showing various options for the timing of financial flows (early harvests by means of thinnings, harvests at rotation age and postponed harvests).

Risk of fire damage was modelled with a yearly damage probability of 0.01. This value was calculated from fire events in the selected buffer zone based on MODIS-TERRA 2000–2006 hotspots (Tab. 4). Fire probability was obtained by dividing the annually burned area by the total area. Fire probabilities ranged from 0.005 (inside Podocarpus National Park) to 0.022 for areas between Loja and the Podocarpus National Park. For the eastern and north-eastern buffer zone of the Podocarpus National Park, a fire probability of 0.008 was obtained. However, fire probability is not equal to damage probability. First, our reforestation calculation included fire protection. Second, Andean alder achieves fire resistance from age 5 onwards, great enough to survive low-intensity fires typical of open woodland sites (GRAU and VEBLEN 2000). For these reasons a yearly damage probability of 0.01 is a rather pessimistic assumption used to carry out a careful evaluation.

A given annual damage probability of 0.01 resulted in a survival probability for the Andean alder reforestation of 0.82 up to an age of 20. Survival probabilities for every year were used in correcting the prediction of stumpage values according to losses from fire. Besides the risk of fire hazard, growth fluctuation (10%) and timber price volatility (10%) generated uncertainty involved within our growth scenario. Stumpage values according to minimum stumpage value scenarios were thus once again lower than stumpage values which were only corrected due to fire losses. Despite the careful assumptions, the minimum stumpage value at age 20 amounted to US\$ 3,347 ha⁻¹ and thus exceeded the amount of US\$ 2,653 ha⁻¹ (investment of US\$ 1000 ha⁻¹ plus accumulated interests) to be paid back after 20 years.

Selective logging

An alternative to tropical forest conversion is to manage them on a biologically sustainable basis. To explore the potential for such sustainable, selective use, a natural forest experiment was established in the San Francisco valley (GÜNTER et al. 2008). The results obtained showed no significant effects of

timber harvesting operations on epiphytes under low-impact felling regimes within 12 months after felling. Moreover, neither the number of moth individuals (bioindicators) nor their local species diversity was affected by the silvicultural treatments. The impact of the treatments on nutrient cycling was low as well. It was concluded that sustainable, reduced-impact management that maintains biodiversity, may be possible in the tropical mountain rainforests of Southern Ecuador.

Our calculation referred to a typical tree size structure for a comparatively highly stocked natural forest in the San Francisco valley. This forest type was previously considered to anticipate the financial consequences of its conversion into pasture, while here it is considered for sustainable management under a selective logging regime. Data on forest growth (GÜNTER et al. 2008, OESKER et al. 2008) and mortality (GÜNTER, pers. comm.) were used to predict sustainable harvest. Here sustainable harvest implies that the actual forest structure will not be changed. An average growth of 1.55% for the corresponding forest type within the RBSF area was reported when related to the trees basal areas (OESKER et al. 2008). An average diameter growth of 2.6 mm year⁻¹ corresponded to this growth rate. This average value is certainly an underestimation for dominating commercial trees with free crowns (GÜNTER, pers. comm.). However, to carry out cautious predictions we computed the rather low diameter increment value. Given 5 cm wide diameter classes, the mentioned increment resulted in a yearly transition probability for trees of 0.052, thus indicating the probability that a tree leaves its diameter class to enter the next greater diameter class. Considering the measured mortality (GÜNTER, pers. comm.) the transition probability was used to estimate the yearly surplus of stems in diameter classes above 40 cm in dbh. Only the surplus, but not all trees above 40 cm can be harvested without changing the present tree size structure of the forest. We obtained a yearly surplus of minimum 40 cm dbh trees (net of mortality) of 1.5 trees ha⁻¹, when compared to the given forest structure. This number of trees was seen as the possible harvest for each year without changing the actual forest structure. The harvestable trees represented an average bole volume of 1.482 m³, of which 50% (0.741 m³) were considered merchantable (LEISCHNER 2000). Given a net revenue of US\$ 42.30 m⁻³, yearly sustainable net revenues of US\$ 31.4 ha⁻¹ resulted. The combined uncertainty of timber biophysical yield (30%), timber price volatility (10%) and logging costs (10%) amounted to a total SD of ±21.80.

4 Results

4.1 Effectiveness of possible carbon payments

The data obtained from the farm survey showed a great variation in pasture and natural forest areas, agricultural revenues and net revenues per hectare per year (Tab. 2). The great differences between median and mean in combination with standard deviations on the order of the mean values, indicate a largely non-normal distribution of the recorded parameters. Comparing the relation between existing natural forests and pastures reveals that most farms still have more natural forest than pasture area (Fig. 2). The considerable area of natural forests and the high variability of financial indicators suggest that considerable future deforestation is possible and that compensation requirements will show a great variation.

With a range from 1.8 to 37.5 ha, the area under pasture varied greatly also among the three considered example farms (Tab. 1). Additionally, the yearly net revenues showed a great range from US\$ 99 to 386 per ha. This is an amazing range, but it compares well with yearly net revenues published by WUNDER (2000). At a given site, net revenues in WUNDER (2000) included a range between US\$ 48 and 369 per hectare, only depending on how long the considered pasture has already been utilized. We shall discuss this crucial aspect later.

While the resulting net revenues for farm type A are very small (Fig. 3d), farm management is a very important source of income for farm type C (Fig. 3f). The projected total deforestation area over 40 years showed a considerable range as well: They comprise values from 1.5 ha (farm type A) up to 31.2 ha (farm type C), while farm type B had an intermediate position with 15.4 ha. Farm C, with the highest net revenue coefficient per ha, also showed the highest expected deforestation rate (1.7% annually). In the cases of farm types A and B the deforestation rate was only 0.3% per annum, owing to the fact that the proportion of natural forests was still great (shares of natural forests between 88 and 90).

Given the great variability described above, the land NPVs that result from the expected future net revenues over 40 years were also enormously variable (Tab. 1). They covered a range from US\$ 2,902 (farm type A) over 29,550 (farm type B) to 220,433 (farm type C) per farm. Without the opportunity to convert natural forests into pasture (i.e., under an avoided deforestation strategy) the land NPVs de-

clined by US\$ 883 (farm type A), 8,974 (farm type B) and 57,690 (farm type C) per farm. At minimum, these values had to be offset by carbon payments to compensate opportunity costs from avoided deforestation.

The expected carbon emissions ranged between 37 (farm type A) and 836 Mg C per farm (farm type C) in the years for which deforestation was modeled (years 11, 21, 31, see figure 4). The most profitable farm, type C, was expected to be the greatest carbon source. For a carbon compensation programme of US\$ 10 per Mg C, none of the above reported losses in NPV, caused by avoided deforestation, was covered. With US\$ 19 per Mg C, farm type B showed the lowest breakeven price, where the breakeven price is that carbon price which compensated the NPV losses when foregoing the net revenues from natural forest conversion. While the breakeven price for farm type A was similar (US\$ 20.5 per Mg C), the breakeven price for farm type C (the farm type with the highest net revenues per hectare per year) resulted in US\$ 65.2 per Mg C.

4.2 Differences in and distributional effects of required compensation payments

The required price per Mg C to finance compensation of avoided deforestation as computed above is determined by the net revenues which can be obtained by pasture management (Fig. 5). These net revenues varied greatly among the surveyed farms. If we sort the farms according to their minimum required compensation price for 1 Mg C and consider the area of natural forests that these farms own, we can determine how much tropical forest area could be protected against conversion at various carbon compensation prices (Fig. 6a). This analysis shows that a carbon compensation of US\$ 10 per Mg C would be interesting for 20 farms representing only about 10% of the total surveyed tropical forest area. To make avoided deforestation and carbon storage interesting for farms that hold 50% of the natural forest area, a carbon price of about US\$ 25 per Mg C were needed. Only for a price of around US\$ 80 per Mg C could almost all farmers consider avoided deforestation as a possibly acceptable strategy.

Surprisingly, pasture size and economic success as indicated by profits is not correlated for the sampled farming households. The 20 farms potentially accepting US\$ 10 per Mg C in compensation have a mean farm size of very close to 16 ha. The

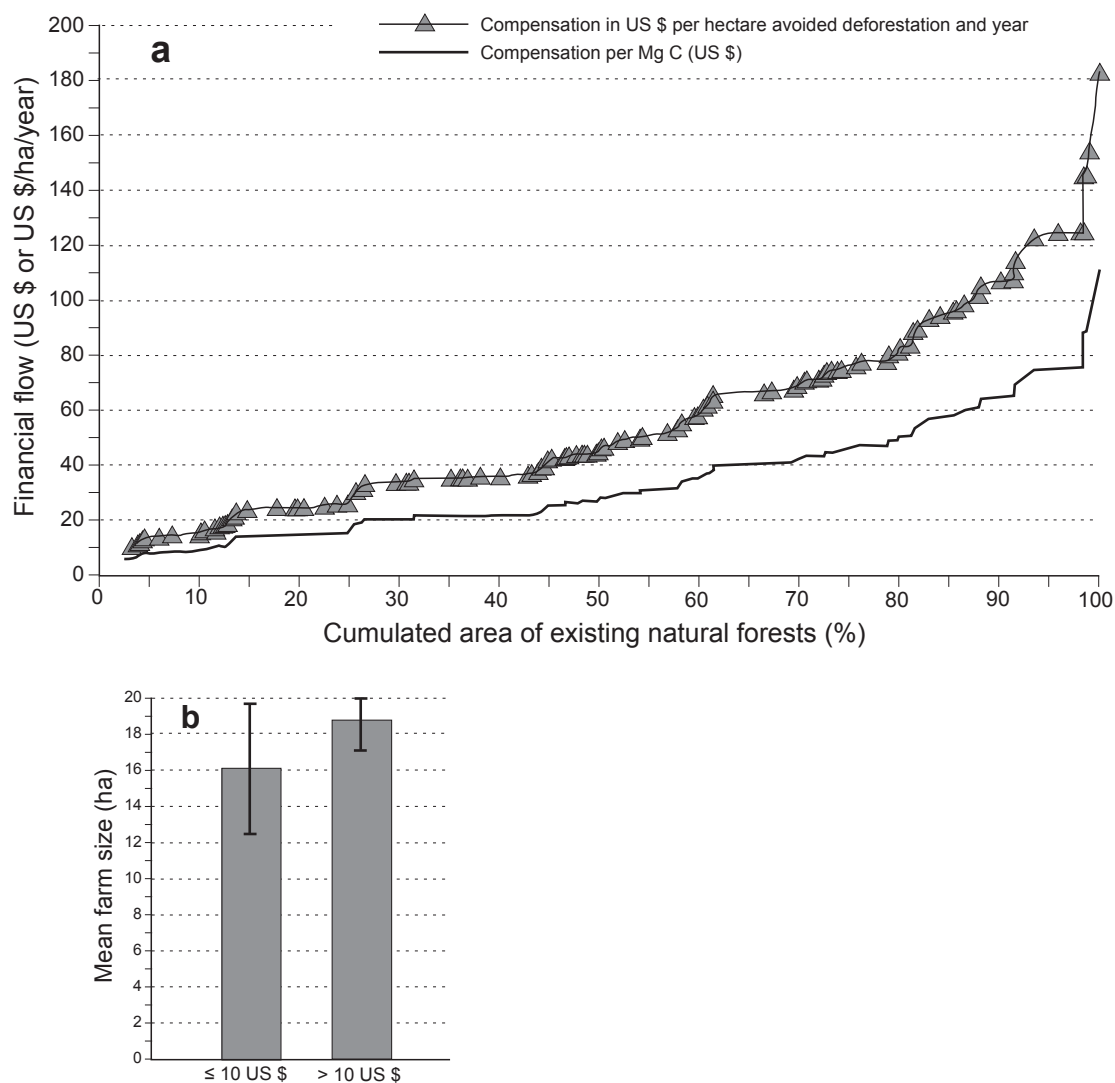


Fig. 6: Compensation requirements of 130 surveyed farms: (a) Ascending sorted compensation requirement over accumulated natural forest area of farms. (b) Comparison of average farm size with a lower or a higher compensation requirement than 10 US\$ per Mg C (error bars are ± 1 SE; the difference is not significant ($p=0.51$ with a heteroscedastic, two-sided t-test))

110 farms with a higher compensation requirement were only little bigger on average (18.8 ha). As the overlapping standard errors depict (Fig. 6b), this difference is not significant from a statistical point of view. If conversely, a mandatory deforestation ban is enforced and farmers are compensated at the average area-weighted compensation requirement (around US\$ 32 per Mg C), 50 farms would not be fully compensated. In extreme cases, such as farms with small intensively used current pastures but rather large forest holdings, the uncompensated opportunity costs may approach or even surpass the current net revenues from compensation.

4.3 A productive land use concept to keep farmers interested in sustainable land management

Given the results described above it could be dangerous to rely only on financial compensation that is financed by the carbon market or other instruments to conserve tropical forests. Even if very high carbon prices could be achieved, it is necessary to keep the farmers interested in established farms to avoid that they establish new farms elsewhere.

The choice of a diversified land use concept might be a solution for the problem of leakage (Fig.

7). Under this type of management, the land use at farm level, demonstrated for farm type B, is immediately diversified into agricultural and forestry uses. Forest uses include reforestation; a small part of the pasture area will be reforested with Andean alder already at the beginning of the considered period with increasing areas when degraded pastures accumulate. Note that a reconversion after 20 years of part of the Alder plantations back to pasture management was considered.

Besides the establishment of Alder plantations, selective but sustainable logging was considered in a portion of the natural forests (white, dotted area in figure 7). The freed working capacity by means of reduced pasture area is allocated to selective logging activities in this portion of the natural forest area to compensate for missing early revenues from reforested pastures. Carbon emissions from selective logging (only 0.75 cubic meters are harvested per hectare per year) are not considered, as we maintain the natural forest in structural and standing timber stock equilibrium. This can be justified because we reduce mortality due to careful timber extraction, which would, if not reduced, also cause some carbon emissions.

Even without considering payments for avoided deforestation, the mixing of land uses stabilises net revenues and accumulates substantial monetary value with the natural growth of the newly established alder plantations until they are harvested at age twenty. In the meantime they deliver net revenues from thinning (from year 11 onwards). After the harvesting of the tree plantations the areas are again used for pasturing. This concept stabilises net revenues, enhances livelihood for the farmers and avoids deforestation.

The financial results of the mixed land use con-

cept were also convincing. The land NPV of farm type B, which we selected as an example, improved from US\$ 29,550 to 36,475 per farm. The total farm thus gets a higher economic value for the farmer, a fact that may keep him/her on the same farm. Sensitivity tests (varying input coefficients) showed rather great stability of these results. However, the mixed land use scenarios were financially successful only when an interest of 5% was applied but not for much higher interest rates. The limiting factor was the internal rate of return of the reforestation with Andean alder that was calculated to be 6.3% (minimum internal rate of return, given the risks described).

5 Discussion and conclusions

Our investigation revealed results to partly answer and discuss the questions that we formulated earlier:

- Are expected carbon compensation prices high enough to convince tropical farmers for avoiding deforestation?
- How does required financial compensation vary between differing farm holders?
- How could financial compensation for avoided deforestation be combined with productive land use options to keep the farmers on their established farmlands and thus avoid leakage?

Carbon compensation prices

We have seen that only a few farms would possibly consider accepting avoided deforestation for carbon prices on the order of US\$ 10 per Mg C, given the validity of the assumptions made in our valua-

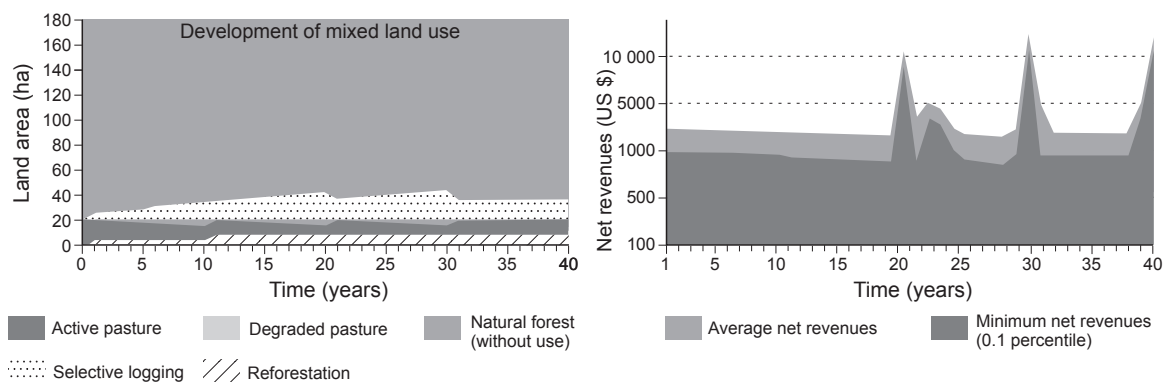


Fig. 7: Mixed land use scenario (left) and yearly net revenues (right) for example farm B when deforestation is stopped immediately and land use is diversified into agriculture and forestry

tion. However, this result depends on the carbon pools that are released into the atmosphere when harvesting tropical forests. Our calculated carbon emission was comparatively low with 74 Mg C ha⁻¹. For example, PEARCE and PEARCE (2001) expect carbon emissions of 122 Mg C ha⁻¹ for the conversion of a closed secondary forest into pasture and 52 Mg C ha⁻¹ if the converted forest is an open forest. We have to consider that the Ecuadorian forests considered here belong to highland ecosystems, which, at least in part, have been exploited before. If we assume US\$ 10 per Mg C as fully available for compensation, an avoided carbon emission of 151 Mg C ha⁻¹ was necessary to make this compensation attractive for farm type B. For a tropical highland ecosystem this avoided carbon emission by tropical forest conservation seems rather high.

Another argument that challenges our assumptions might be seen in the rather low carbon price that we exemplarily assumed as a possible compensation programme (US\$ 10 per Mg C, which is US\$ 2.7 per Mg CO₂). It is important to note that our carbon price is a pure opportunity costs and represents what the farmer completely receives, while prices generated by the international carbon markets or other financing instruments (e.g., voluntary carbon funds) would also have to cover substantial transaction costs (STERN 2006; ELIASCH 2008). Nevertheless, carbon prices achieved within the European trading system are much higher than our assumption on the order of well above US\$ 10 per Mg CO₂ on average; however, the prices were also sometimes zero in the past and showed generally an extremely great volatility. The Copenhagen meeting of the UNFCCC parties in December 2009 will show whether or not reduced deforestation can actually be traded as certified credit under the Kyoto Protocol's Clean Development Mechanism, which would be a precondition for higher carbon prices (BUTLER et al. 2009). Our compensation price of only US\$ 2.7 per Mg CO₂ for standard calculations was, however, not unrealistic. Specifically, it has been suggested that an average carbon compensation of US\$ 2.8 per Mg CO₂ would reduce forest emissions by 65% (ELIASCH 2008); a result which could not be confirmed by our study. This compensation would probably attract only some farmers who hold a bit more than 10% of the natural forest area. Moreover, we should point out that our results are not limited by a compensation assumption of US\$ 10 per Mg C. Rather we computed minimum acceptable compensation prices for all 130 investigated farms, which ranged from almost zero up to more than US\$27

per Mg CO₂. With our maximum estimates we exceed the maximum opportunity costs arising from global model estimates, which end at a maximum of US\$ 18.86 per Mg CO₂ (UNION OF CONCERNED SCIENTISTS 2008).

A further critique to our scenario approach may be that we assumed deforestation on the example farms being carried out not before year 11. A compensation to avoid deforestation on farms which plan immediate conversion of tropical forest is, however, very similar to our results: It would amount to US\$ 18 per Mg C in the example farm B - compared to US\$ 19 per Mg C under our scenario. We can thus assume that the derived compensation prices would also be valid when considering the avoidance of immediate deforestation.

As a crucial component of our valuation we have to discuss the choice of a 5% interest. Changing the interest to 10% (an interest that, for example, BUTLER et al. 2009 decided to use for their valuation) would reduce the required compensation for farm type B from US\$ 20.5 (interest 5%) to 16.1 per Mg C (interest 10%). Given such a high interest would mean that the farmers would not care about income losses due to reduced deforestation which lie in future. The effect is drastic: US\$ 100 received after 20 years have a present value of US\$ 14.86 when discounted by 10%, the same amount received after 40 years would be worth only US\$ 2.20. This effect is not so extreme if a 5% interest is used: the present value is then US\$ 37.69 when US\$ 100 are received after 20 years and US\$ 14.20 when US\$ 100 are received after 40 years. Given a 40-year time horizon it seems hardly justifiable to use an interest rate of 10%. If we did, we would need to predict inflation effects, as a 10% interest can hardly be seen a real interest rate. Instead of using a high interest, we assumed a 5% interest as a real interest rate with all costs and prices being constant over time. This appears rather justifiable even though it is a convention.

Finally, we implied that all compensation payments are certain payments. If compensation was uncertain, higher average compensation was necessary to address the usually risk-avoiding attitude of farmers (PICHÓN 1996). For example, given the financing of compensation payments by means of carbon markets, we have to consider that here the prices are highly volatile (WHITESELL 2009). As a consequence, either the area that can be covered by compensation payments or the payment itself would fluctuate. It is hard to predict which consequences this would have for the effectiveness of compensation payments.

Differences in required financial compensation, distributional effects

The farm survey showed an extremely high variation of yearly net revenues which determines compensation requirements. Although not unrealistic (see WUNDER 2000), one can call into question if these variable net revenues are a sound basis to derive compensation requirements. To come to an answer, we shall first base our consideration on the premise that a voluntary PES scheme seems advisable to compensate for reduced emissions from deforestation, because the land opportunity costs are rather moderate (WUNDER 2007) and because its flexibility, efficiency and social aspects speak for a voluntary compensation scheme. If we assume a voluntary character for compensation, we should bear in mind the results of conservation auctioning, where actors selling conservation offer a defined conservation service for a compensation which they define. These results show that local actors offer conservation for prices higher than their real opportunity costs, while striving for a conservation premium (LATACZ-LOHMANN and VAN DER HAMSVOORT 1997). When looking for acceptable carbon prices we should thus look at the high possible pasture net revenues to be compensated, even if they might be not sustainable. Our analysis of highly variable compensation prices may thus be helpful, although some of the net revenues that formed the basis to compute compensation prices are certainly not sustainable. Farmers will nevertheless use the higher range of possible net revenues, even if only possible for a short time, simply as an orientation. This view is confirmed by the willingness to accept compensation for forest conservation of slash and burn farmers from the Peruvian Amazon, reported by PEARCE and PEARCE (2001). These farmers would accept compensation of US\$ 100-200 per hectare per year to switch from slash and burn farming to full forest conservation. This compares well with our results: The highest compensation requirement was US\$ 183 per hectare per year (Fig. 6a), which would possibly assure that all farmers would be interested in a conservation policy.

Surprisingly, pasture size and farming profit was not correlated in the farm survey data. Thus, the introduction of a *voluntary* remuneration scheme for avoided deforestation would not introduce systematic distributional effects, such as that only the biggest landholders would benefit from a voluntary scheme. However, a *mandatory* and enforced ban on deforestation coupled with a 'fair' compensation payment equal to mean compensation requirements may still have undesirable social consequences for many farmers in

terms of high opportunity costs not compensated for.

In summary we can say that it is rather unlikely that compensating the usually low average land opportunity costs alone will be able to convince many farmers in highland ecosystems, such as those close to the Podocarpus National Park in Ecuador, of the advantages of tropical forest conservation.

Productive land use concept to avoid leakage

The economic results of the mixed productive land use alternative, that was tested to keep farmers financially interested in their existing farms also without deforestation, were promising. In the long term the reforestation of abandoned lands in combination with selective logging of the natural forests and agricultural pasture use enlarged the economic farm benefits by 23% and stabilised the fluctuation of yearly net revenues. At the same time, deforestation could be stopped.

However, one consequence of this concept is a shift from pasture to forestry. The pasture area stabilizes at around 60% of the initial area after 40 years. Although the mixture between pasture and forestry reduces the volatility of net revenues, enhances livelihood for the farmers and avoids deforestation, it thus diminishes the area for agricultural production. Maintaining the initial agricultural area when facing pasture degradation would, however, only be possible under further deforestation or re-pastorisation of abandoned pastures. Other limiting factors of the mixed land use concept include missing availability of cheap credits, lacking silvicultural knowledge and the absence of early net revenues from reforestation areas. If farmers could get compensation payments for carbon storage also under selective logging, this money could be used for reforestation investments, so that the investments that must be financed by credits would shrink considerably. A combination of mixed and productive land use concepts with payments for avoided deforestation, seen as PES, could thus be a viable option for tropical forest conservation.

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