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Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico

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ABSTRACT

Tropical montane cloud forests deliver important goods and services to society, such as timber, the supply and purification of fresh water, and carbon sequestration. In spite of their relevance, current deforestation rates are very high, at the expense of affecting the provision of ecosystem services. We explore the impact of land use change in terms of provision of ecosystem services by following two approaches, one very detailed (focused on hydrological services – water quality) and another one with a broader perspective (at a large scale and considering the ecosystem service value (ESV) of several ecosystems and their ecosystem services at the same time). In the highlands of the State of Veracruz, previously forested lands were converted into coffee plantations and cattle ranches. To evaluate the role of species composition and community structure on water quality, we studied nine small watersheds (<15 ha) covered by pristine cloud forest, coffee plantations and cultivated grassland (three each). Species richness of the three land use types was similar, although species composition was as different as 90%. Overall species diversity as well as that of woody species, and growth form diversities decreased in the transformed land uses. Water quality of streams flowing through these watersheds declined: nutrients (nitrate), conductivity, cations, chloride and suspended solids were lowest in the forest streams and highest in streams from coffee watersheds, whereas grasslands were intermediate. We also calculated ecosystem service values (using the transfer value method) and estimated economic market–non-market gains and losses owing to land transformation. Loss of natural ecosystems may imply a significant economic loss to society in terms of ecosystem services, although market gains may still lead land owners to land conversion because revenues are higher. Adequate Payment for Ecosystem Services may be a good option to prevent deforestation, but the compensation should be at least equal to the opportunity cost of the promoted land use. Our estimates are indicative of the urgent need to go beyond water quantity as the most relevant ecosystem service considered in PES schemes.

1. Introduction

Land use has been changing ever since humans first began to manage their environment. However, the changes that have taken place over the last 50 years have been especially important and intense (Metzger et al., 2006), as society is becoming increasingly urbanized, while natural ecosystems become deteriorated. Land use changes may eliminate species locally and decline natural habitats and ecosystem functioning, affecting thus, biodiversity and provision of ecosystem services (Priess et al., 2007; Turner et al., 2007a; Ricketts et al., 2004, 2008; Steffan-Dewenter and Westphal, 2008; Lavorel et al., 2007; Turner et al., 2007b). Evidence of loss of ecosystem services owing to land use changes is gradually accumulating, especially in the case of pollination services (Priess et al., 2007; Ricketts et al., 2008; Steffan-Dewenter and Westphal, 2008); carbon storage (Huston and Marland, 2003; Kirby and Potvin, 2007); hydrology (Strange et al., 1999); and climate change (Schroter et al., 2005), among others. These changes are likely to have implications on human well-being (Balmford and Bond, 2005; MEA, 2005; Butler and Oluoch-Kosura, 2006; Pattanayak and Wendland, 2007).

Tropical montane cloud forests ("cloud forests" from now on) offer important ecosystem services, such as water supply and quality (Bruijnzeel, 2004; Bonnell and Bruijnzeel, 2005). The high-elevation of these forests usually increases local water supply by

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raking moisture out of the fog-saturated atmosphere that would otherwise remain in vapor form. In addition, evapotranspiration in cloud forests is also reduced, because their foliage is constantly wet, which means they pump less moisture from the soil back to the atmosphere. For instance, Bruijnzeel (2004) has estimated that moisture interception of cloud forests is as high as 1000 mm rainfall per year and consequently, soils are saturated more quickly and for a longer period. As a result, for a given level of rainfall, stream flows originating from cloud forests tend to be more constant than those from grasslands or other types of land cover (Postel and Thompson, 2005).

Besides capturing water, it has also been demonstrated that the vegetation that surrounds rivers and streams (such as cloud forests) acts as natural filters against pollutants dumped on the slopes of the micro-watersheds, and thus, plays a key role in water quality (Peterjohn and Correl, 1984; Malanson, 1993; McDowell and Asbury, 1994). In this sense, not only vegetation cover, but its diversity and structure, both, above and below-ground, as well as the interaction between the canopy, litter, and soil, affect ecosystem functioning and provision of ecosystem services, such as water quality (Diaz et al., 2007; Quetier et al., 2007; Turner et al., 2007a). For instance, cations (Na^+, K^+, Ca^{++} and Mg^+), Cl^- and nitrate concentrations are expected to increase with logging followed by agriculture (Johnson et al., 1997; Herlihy et al., 1998; Williams et al., 2005).

In the highlands of the central region of the state of Veracruz (located in the central part of the Gulf of Mexico) previously forested areas (covered with cloud forests and oak forests) have been logged and converted mostly into coffee plantations, grasslands for cattle ranching and sugar cane (Williams-Linera, 2007; Muñoz-Villers and López-Blanco, 2007). Natural ecosystems have changed (and even disappeared) drastically, probably at the expense of additional ecosystem services. In this sense, the innovative Program of Payment for Hydrological Environmental Services (Pago de Servicios Ambientales Hidrológicos – PSAH) was conceived to provide incentives to avoid deforestation and to face water scarcity (Muñoz-Piña et al., 2008). Forest cover was used as one of the criteria for eligibility of landowners to be admitted into the PSAH program. In the Mexican scheme, different forests are considered of different relevance in terms of hydrological services. For instance, cloud forests were considered as the most important because of their role in capturing water (Bonnell and Bruijnzeel, 2005). The Mexican PSAH program is financed by the users, who are charged an additional fee in their water bills (Muñoz-Piña et al., 2008).

When demand for participation in the PSAH exceeded supply funds, the government suggested that additional indicators (such as amount of captured water in the rivers within the watershed, instead of forest cover alone) be incorporated in the operating rules, as components of the grading system used to evaluate the applications to qualify as beneficiaries of the PES program (Muñoz-Piña et al., 2008). Nevertheless, in Mexico, additional ecosystem services, such as water quality, have not been considered so far. In this sense, in our study we wanted to demonstrate that additional ecosystem services (beyond water for downstream irrigation) should also be considered so that the benefits of PES are enhanced.

In this study we aimed at analyzing the impact of land use change in terms of provision of ecosystem services. We followed two approaches, one very detailed (focused on hydrological services – water quality) and another with a broader perspective (at a large scale and considering several ecosystem services at the same time). Firstly we wanted to evaluate whether water quality changed in streams flowing through different land uses, with different biodiversity, and community plant composition and structure. Secondly, we aimed at determining whether these land use changes have lead to a diminished ecosystem service value (ESV) (economic value of ecosystem services) because of potentially diminished provision of ecosystem services in general. With these, our intention was to offer additional information (impact of land use change in terms of ecosystem services other than capturing water) that would be useful in terms of guidance or management implications (Troy and Wilson, 2006).

2. Methods

2.1. Study site

We studied the highlands of La Antigua watershed, which is located in the central region of the Gulf of Mexico (between 19°05′–19°34′N and 96°06′–97°16′W) (Fig. 1). The La Antigua watershed covers 2623 km^2. The highland region was determined based on topography and river flows, and it covers 1294 km^2. Nearly 40% of this area is still relatively well preserved with relatively large extensions of cloud forests. These forests are amongst the highest altitude cloud forests, according to the classification of Mexican vegetation elaborated by Rzedowski (1996).

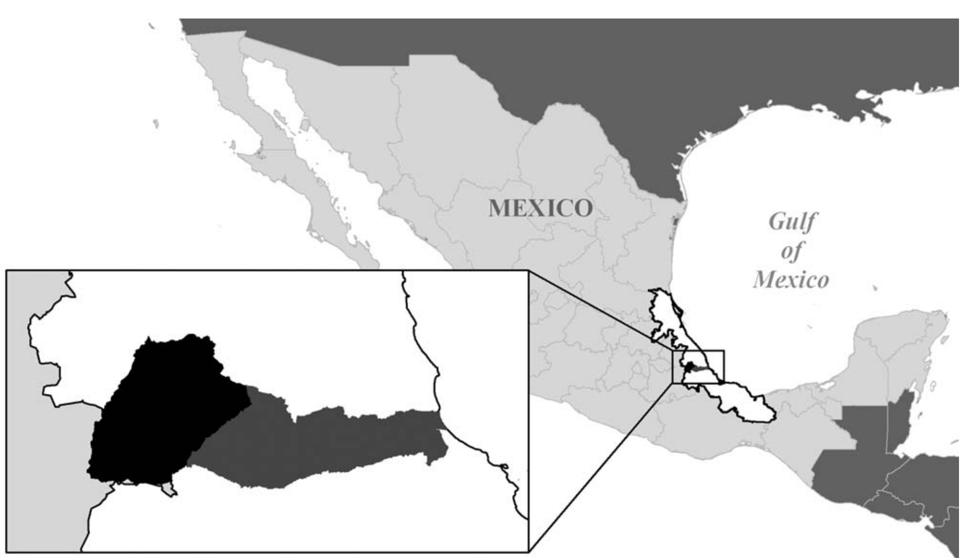

Fig. 1. Location of the La Antigua watershed. The darker area highlights the study site (highlands of the watershed).

There are 17 municipalities in the highlands of the watershed; 13 belong to the state of Veracruz and 4 to Puebla. Several municipalities (Banderilla, Huatusco, Las Vigas de Ramírez, Perote, and Tlachichuca) are marginally located within the watershed (less than 5% of their surface lies within the watershed) and thus, they were not included in the current analysis. Because the upper watershed covers a wide altitudinal range from 480 to 4200 m.a.s.l., with a mean altitude of 1920, it is affected by many weather regimes, ranging from hot-subhumid in the lower watershed to temperate-cold at higher elevations. Similarly, mean annual temperatures range from 9.5 to 25.5 °C in the higher and lower altitudes, respectively. Many permanent and temporal streams and rivers cover the area. The largest and widest rivers running through the highlands are: Sordo, Pixquiac, Pintores, San Andrés, Calpixcán, Texoco, Caracol, Tecomatla, and Gavilán. All of them drain into La Antigua River after which the watershed is named.

2.2. Land use change

Land use change from 1973 to 1989 was calculated from satellite Landsat images (May 5th 1973 and October 26th 1989) from the Global Land Cover Facility (http://glcf.umiacs.umd.edu/index.shtml). The bands were coupled with ERDAS (v. 8.7). In order to make both images compatible and with the same spatial resolution, the 1989 image was sampled with a resolution of 57 m. The images were classified by means of multi-resolution segmentation, which yielded regions in the image that belonged to the same spectral class (i.e. land use or vegetation type). These regions were simplified by means of an "Unsupervised Classification (isodata)" (ERDAS). The resulting clusters and polygons were labeled according to texture, color, cluster and status in 1989. Information on land use changes from 1990 to 2003 was gathered from the estimations by Muñoz-Villers and López-Blanco (2007). Finally, rates of land use change were calculated as follows:

$$\text{Rate of land use change} = \frac{A_{\text{final}} - A_{\text{initial}}}{A_{\text{initial}}}$$

where A_{final} and A_{initial} represent final and initial area (respectively) of a given land use.

2.3. Vegetation composition and structure

Nine micro-watersheds with first-order streams were selected based on satellite images within which we determined those micro-watersheds that were mostly covered (>70%) by each of our focal land use types (cloud forests, coffee plantations and cultivated grasslands). Later, the final selection of micro-watersheds was made based on accessibility to the potential study sites.

We sampled standing vegetation in order to determine community composition, diversity and structure. We were fully aware of the relevance of underground community structure in terms of water flow and quality. However, assessing the below-ground attributes of a community is certainly complex. Nevertheless, because it is very likely that above-ground diversity mirrors below-ground diversity (Negrete-Yankelevich et al., 2007a,b) we considered it reasonable to expect that the higher above-ground biodiversity of cloud forests would likely affect below-ground functioning and, ultimately, the quality of water flowing through it. We thus focused solely on above-ground vegetation and used it as an approximation of below-ground community attributes and functioning.

In each micro-watershed, we sampled trees (DBH $\geq$ 5 cm) and shrubs (taller than 50 cm) in ten 10 m $\times$ 10 m plots per micro-watershed. Six of these plots were located perpendicular to the riverbed and four parallel to it. Plots were from 5 to 10 m apart (depending on accessibility) and were placed in sites with no evidence of man-made paths. We identified every plant species its estimated percent cover. Plant cover was estimated by means of the modified Braun-Blanquet scale (Westhoff and van der Maarel, 1978), where 1 = 1–2 individuals, covering <5% of the sampled area; 2 = 3–10 individuals covering <5% of the sampled area; 3 $\geq$ 10 individuals, covering <5% of the sampled area; 4 = abundant individuals, covering <5% of the sampled area; 5 = plant cover ranging from 5 to 12%; 6 = plant cover from 12 to 25%; 7 = plant cover from 25 to 50%; 8 = plant cover from 50 to 75% and 9 = plant cover from 75 to 100%. We also studied herbaceous vegetation by following a similar procedure. In this case we worked in three 2 m $\times$ 2 m subplots that were randomly placed within the 10 m $\times$ 10 m ones. We thus sampled ten 10 m $\times$ 10 m plots for trees and shrubs (a total of 1000 m^2 per land use) and 30 2 m $\times$ 2 m subplots for herbs (a total of 120 m^2 per land use). We calculated Simpson's Diversity Index for each land use type (Magurran, 1988) and Jaccard Similarity Index to compare among different land uses, considering all the sampled species as well as separating woody and herbaceous plants. Finally, in each land use type we estimated structural diversity following the same calculations as is done for Shannon-Wieners Index (Magurran, 1988). However, we used the proportional value (percentage) of each growth form as our P_i values, instead of P_i representing the proportion of each species found. Comparison of diversity indices was done by calculating a *t*-value for each pair of calculated indices, according to Magurran (1988).

In order to summarize changes in community structure and composition with changing land uses, and also in order to assess variability between our sampled plots within each land use, we used principal component analysis (PCA) using PC-ORD for windows (McCune and Mefford, 1999). Our biotic survey was amenable to a PCA, because our data "did not encompass so great a range of environmental variation that species responses are non-linear" (Kenkel, 2006). We carried out the PCA by combining the matrices from the plots located in the three land uses.

2.4. Water quality

We used the same micro-watersheds sampled for vegetation structure and composition. However, when monitoring water quality, we had two rivers located in micro-watersheds covered mostly with cloud forest; three with coffee plantations and three with grasslands. We only worked in two cloud forest rivers because of the rocky and extremely pronounced slopes of the third watershed, which kept us from reaching the river. Field sampling took place at monthly intervals from July 2005 to May 2006 and then values were averaged to have an estimation of mean yearly values of water quality of the sampled streams.

Every month, water samples were collected during three days, one for each land use type. *In situ* conductivity was measured at each stream (multiparameter YSI Mod. 85). Additionally, two 1 l water samples were collected per stream in order to determine in the laboratory (at the Instituto de Ecología, A.C.) the variables listed next: total suspended solids (TSS), alkalinity ($CaCO_3$), nitrates (NO_3^-), chloride (Cl^-), calcium (Ca^{++}), sodium (Na^+), magnesium (Mg^{++}) and potassium (K^+), following the APHA (1998) methodology. Calcium and magnesium were determined with an atomic absorption spectrophotometer (Shimadzu Mod. AA6501) and sodium and potassium were measured through flame photometry (Corning Mod 410). All these analyses were performed within 48 h after water samples were obtained. The samples were kept in the refrigerator (4 °C) until they were analyzed. A non-parametric analysis of variance on ranks was applied to perform comparisons of water quality monitored in the three different land use types.

2.5. Ecosystem service values

We used the ecosystem service values as a proxy measure to estimate the changes in the "non-market" economic value of the natural ecosystems located in the highlands of La Antigua, over the last decades. ESV can be defined as the total value of ecosystem services and products provided by different ecosystem types. It is calculated by multiplying the area of natural ecosystems by the corresponding ecosystem service value.

There are several methods that can be used to estimate the monetary value of the ecosystem services provided by natural ecosystems. One of these valuation techniques is the transfer value method, which transfers the monetary value of environmental goods and services determined in one place and time, to make inferences about the economic value at another place and time (McComb et al., 2006; Rosenberger and Stanley, 2006). Once the value has been transferred to the study site, it is possible to estimate the ecosystem service value. In addition, if land area data are available for multiple years, one can compare past and current changes on ecosystem surface and learn how important that gain or loss of area has been for the economy of the study site, in terms of ecosystem service values (Troy and Wilson, 2006; Rosenberger and Stanley, 2006).

Currently, several international databases are available to perform transfer valuations. These databases allow for the valuation of ecosystem services in areas where local valuations have not been performed yet, as in our case. To perform the transfer value method we used the database compiled by Lithgow (2007) that focused on economic valuations of ecosystem services, calculated in ecosystems and economic and social environments that are similar to the Mexican ones (tropical Latin America). When no information was available, we used the global estimations performed by Costanza et al. (1997). With this we obtained the ecosystem service value per hectare per ecosystem in terms of $US per year. The total ESV calculated for each observation period considering the area covered by different land uses, generated total ESV per observation period.

This approach assumes constant marginal values of ecosystem services within biomes, and it does not account for within-biome, spatial or temporal variation. Marginal value refers to what one more unit of a good is worth to a person in terms of another good (Friedman, 1990). In other words, the marginal value refers to the potential economic difference that having one more good makes to a person in terms of the value of another good. In this sense, we recognize that this and other assumptions are only approximations to the potential economic value of ecosystem services (Costanza et al., 1997; Turner et al., 2007a). Nonetheless, the ESV analysis is the only compilation of valuations of a range of services and habitat types that allows to perform economic estimations at regional (Viglizzo and Frank, 2006) and even global scales.

Table 2
Vegetation composition and structure of three land use types in the highlands of the La Antigua watershed. Numbers in parenthesis show the relative abundance of each growth form (% of species). Letters following Shannon-Wiener Diversity Indices (H') (in terms of composition and structure) indicate significant differences between the three land uses, at $p < 0.05$.

Vegetation type	Cloud forest	Coffee	Grassland
Species richness	260	237	183
Number of families	84	72	62
H' all species	5.576a	5.464a	5.193b
Number of trees	67 (26%)	28 (12%)	13 (7%)
Number of epiphytic trees	1 (0.4%)		1 (0.5%)
Number of shrub species	35 (13%)	17 (7%)	8 (4%)
Number of epiphytic shrubs	2 (0.8%)	2 (0.8%)	4 (2%)
Number of vines	14 (5%)	5 (2%)	1 (0.5%)
Number of herbs	72 (28%)	146 (62%)	118 (64%)
Number of epiphytic herbs	69 (26%)	39 (16%)	38 (21%)
H' woody species	4.654a	3.85b	3.258c
H' herbaceous species	5.069a	5.242b	5.037a
H' growth forms	1.54a	1.15b	1.07b

3. Results

3.1. Land use change

Cloud forests decreased to almost half their original 1973 surface during the last 30 years. Similarly, tropical deciduous forest, croplands and coffee plantations have decreased but at much slower rates (Table 1). In contrast, some natural ecosystems such as pine–oak forests, coniferous forests and alpine grasslands expanded, probably because of natural regeneration of abandoned croplands and also owing to the proliferation of agroforestry (for Christmas trees production). Sugar cane, cultivated grasslands and urban areas more than tripled. Rates of land use change show that sugar cane plantations mostly increased from 1990 to 2003 (Table 1), while urban sprawl and the expansion of cultivated grasslands were most intense from 1973 to 1989. Pine–oak and coniferous forests expanded from 1990 to 2003.

3.2. Vegetation composition and structure

We found a total of 260 plant species in the cloud forest, 237 in coffee plantations and 183 in the grasslands (Table 2), adding a total of 523 species. Species composition was very different in each land use type. The principal components analysis (PCA) explained 32% of the variance and showed that plots located within each land use type were more similar amongst themselves than with the other land use types (Fig. 2). Axis 1 explained 21% of the variance, and represents a conservation gradient, where plots from the forest are on one extreme of the ordination space and plots in grasslands and coffee plantations are located at the other extreme.

Table 1
Area covered (km^2) by different land use types, and rates of land use change (%) in the highlands of La Antigua watershed during three time steps. Positive values indicate increasing surface covered by each land use type and negative values, decreasing surface.

Land use type	1973[a]	1990[b]	2003[b]	Rate of change (1973–1990)[a]	Rate of change (1990–2003)[b]
Cloud forest	520.9	426.9	279.5	−14	−35
Trop deciduous forest	5.3	5.3	5.2	0	−2
Pine–oak forest	92.6	87	112.4	−2	29
Coniferous forest	54.8	41.5	89.8	−21	116
Alpine grassland	9.3	8.8	12.2	−1	39
Croplands	200.2	179.2	143.7	−7	−20
Coffee plantations	305.0	248.6	245	−12	−1
Sugar cane	27.4	13	67.6	−50	420
Cultivated grassland	82.2	239.3	289.4	204	21
Urban area	23.3	65.9	73.7	195	12

[a] Our calculations.
[b] From Muñoz-Villers and López-Blanco (2007).

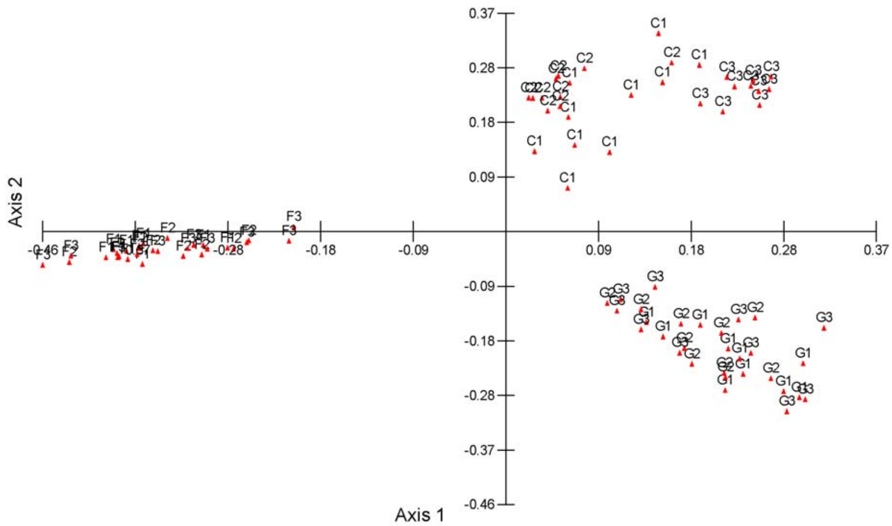

Fig. 2. Principal components analysis showing the variability within and between plots located in the three land uses in the highlands of La Antigua watershed, Veracruz, Mexico. F = forest plots; C = coffee plantations plots and G = grasslands plots.

As expected, Shannon-Wiener diversity index showed decreasing diversity from cloud forest to coffee plantations and then to grasslands (Table 2), but only grasslands were significantly less diverse. Community structure was also very different between the three land use types. Trees were dominant and significantly more diverse in the cloud forest (26% of the species we found were trees) while herbs were most abundant and diverse in coffee plantations and grasslands, representing 62 and 64% of the species we found in each of these land use types (Table 2). Consequently, our calculations show that the structural diversity of cloud forests was significantly more diverse than that of coffee plantations and grasslands, while it was not significantly different between the last two (Table 2).

3.3. Water quality

In general, the values of the indicators associated with high water quality of the streams flowing through the three land use types, decreased from cloud forest to grasslands and coffee plantations. All the variables showed significantly higher values in streams from coffee plantation rivers and lowest in cloud forests (Table 3). Conductivity was lowest in forest streams, followed by grasslands and coffee plantations. The significantly higher concentrations of total suspended solids, nutrients (nitrate), chloride and cations were observed in those rivers flowing through coffee plantations (Table 3). The second highest values were always registered in grasslands. An exception to this trend was nitrate, that was lowest in grasslands, instead of cloud forests. Chloride did not follow the same trends as the other parameters because we measured similar values in cloud forests and grasslands, although coffee plantations again revealed the highest values (Table 3) among the three land use types.

Table 3
Mean yearly conductivity, nutrient concentrations and cations measured in rivers flowing through different land use types. Numbers in parenthesis indicate standard error; letters show significant differences for comparisons made among sites for each variable.

Variables	Cloud forest	Coffee	Grasslands
Conductivity (μS/cm)	30.9 (2.7) a	98.5 (5.7) c	69.2 (3.4) b
Total suspended solids (mg/L)	3.2 (0.7) a	12.1 (2.7) b	4.2 (0.9) b
Alkalinity (mg/L)	15 (1.1) a	41.3 (3.4) b	35.12 (4.7) b
Nitrate (mg/L)	1.4 (0.2) a	3.7 (0.6) a	0.3 (0.04) b
Chloride (mg/L)	4.3 (0.02) a	5.5 (0.1) b	4.3 (0.02) a
Calcium (mg/L)	4.1 (0.6) a	12 (0.06) c	7.8 (0.6) b
Sodium (mg/L)	3.6 (0.4) a	6.7 (0.5) b	6.1 (0.4) b
Magnesium (mg/L)	1.1 (0.1) a	5.4 (0.4) c	3.6 (0.2) b
Potassium (mg/L)	1.2 (0.07) a	2.5 (0.1) b	2.4 (0.1) b

3.4. Ecosystem service values

Natural ecosystems in the highlands of La Antigua watershed offer a wide array of ecosystem goods and services (ES). We found economic valuations for the following ecosystem services: carbon sequestration, water regulation, water supply, erosion control, pollination, biological control, food production and recreation (Table 4). Following different methodologies, the economic values of these services have been estimated by different authors and compiled by Lithgow (2007) and Costanza et al. (1997). According to these estimations, ecosystem services provided by forests showed the highest values, while croplands had the lowest (Table 4). Because of these relatively high values, and the also relatively large area covered by cloud forest, the estimated ESV was much higher (at least nine times higher) for cloud forests than for any other natural ecosystem located in the highlands of La Antigua (Table 5).

Table 4
Ecosystem goods and services offered by natural ecosystems at La Antigua watershed (CS = carbon sequestration; WR = water regulation; WS = water supply; EC = erosion control; *P* = pollination; BC = biological control; FP = food production; Rec = recreation. Ecosystem service values (2004 $ US per ha per year) are given according to [a]Lithgow (2007) and [b]Costanza et al. (1997). Numbers in parenthesis indicate the number of studies from which the economic value is estimated. Data from Lithgow (2007) were estimated from countries with similar social and economic systems (Tropical Latin America). When these values were not available, we used the global values calculated by Costanza et al. (1997).

Land use types (INEGI, 2006)	CS	WR	WS	EC	*P*	BC	FP	Rec	ESV
Cloud forest (2)			×					×	728a
Tropical deciduous forest (2)	×								302a
Pine–oak forest (6)	×							×	728a
Coniferous forest (2)	×								302a
Alpine grassland (1)		×		×	×	×	×		232a
Cropland					×	×	×		162b
Cultivated grassland		×		×	×	×	×		232b
Cropland (sugar)					×	×	×		162b
Cropland (coffee)					×	×	×		240b

Table 5
Changing land use (ha) and ecosystem service value (ESV). ESV values are provided in $US per year (2004) (in millions) based on summarized information condensed by Lithgow (2007) and Costanza et al. (1997). Percent values refer to % from total area at the study site.

Ecosystem types	Area (1973)	Percent	Area (1990)	Percent	Area (2003)	Percent	Net area change (%) (1973–2003)	ES (1973)	ES (1990)	ES (2003)	Net ES change (%) (1973–2003)
Cloud forest	52,090	39.3	42,690	32.2	27,950	21.1	−18.22	66.5	54.5	35.6	−30.8
Tropical deciduous forest	530	0.4	530	0.4	520	0.4	−0.01	1.7	1.7	1.6	−0.1
Pine–oak forest	9,260	7.0	8,700	6.6	11,240	8.5	1.49	7.3	6.9	8.9	1.6
Coniferous forest	5,480	4.1	4,150	3.1	8,980	6.8	2.70	2.9	2.2	4.8	1.9
Alpine grassland	930	0.7	880	0.7	1,220	0.9	0.22	0.2	0.2	0.3	0.1
Cropland	20,020	15.1	17,920	13.5	14,370	10.8	−4.27	3.5	3.2	2.5	−1.0
Coffee	30,500	23.0	24,860	18.8	24,500	18.5	−4.53	7.3	6.0	5.9	−1.4
Sugar cane	2,740	2.1	1,300	1.0	6,760	5.1	3.03	0.5	0.2	1.2	0.7
Cultivated grasslands	8,220	6.2	23,930	18.1	28,940	21.8	15.79	1.9	5.5	6.7	4.8
Other land uses	2,970	2.2	7,500	5.7	7,970	6.0	3.77	–	–	–	–
Total	132,460		132,460		132,460			91.9	80.5	67.7	−24.2

When the landscape is transformed owing to human activities, ecosystem services provided by natural ecosystems are altered too. Hence, ESV increased because pine–oak forests, coniferous forests, alpine grasslands, sugar cane plantations and cultivated grasslands increased over the last three decades (Table 5). However, ESV also decreased because the area covered by cloud forest, tropical deciduous forest, croplands and coffee decreased notoriously. Cloud forest alone resulted in very large losses of ESV, which represented almost 93% of all ESV losses in the area, while cultivated grasslands represented 53% of all gains (Table 5). Overall, in terms of ESV, land use change in the highlands of La Antigua watershed has resulted in relatively large economic losses (Table 5).

4. Discussion

In this study we observed that land use changes in the highlands of La Antigua watershed have resulted in loss of species richness and biodiversity. The modified community structure with reduced growth form diversity resulted in altered ecosystem services other than water quantity, such as water quality. In particular, loss of water quality owing to land use change has been observed in previous studies (Peterjohn and Correl, 1984; McDowell and Asbury, 1994; Quetier et al., 2007; Turner et al., 2007a). For instance, conductivity was higher in agricultural lands because $CaCO_3$ is used in agricultural practices (Williams et al., 2005). In turn, total suspended solids are a result of sediments being flushed towards the rivers because of land erosion after removal of vegetation cover (as occurs in coffee plantations). Cations such as K^+ and chlorides are considered as good indicators of disturbance (i.e. agricultural practices) because the former is easily washed from plant tissues and soils (Williams et al., 2005) and the latter are derived from fertilizers and animal manure (Herlihy et al., 1998). In our study site, potassium chloride and calcium phosphate are used as fertilizers in coffee plantations and grasslands, hence, their high values in the rivers in micro-watersheds with these predominant land use types. High concentrations of nitrate are also strongly associated with agriculture, but in cloud forests, they may be explained by the high Nitrogen mineralization and nitrification of tropical soils (Neill et al., 2001; Williams et al., 2005). Finally, calcium and magnesium are also related to natural intemperization of tropical soils (Williams et al., 2005), as well as the addition of agrochemicals owing to agricultural practices.

In addition to a degraded water quality, further ecosystem services are lost as the landscape is transformed. The estimated net ESV in the highlands of La Antigua has decreased by 24% over the last three decades, because of loss of natural ecosystems such as cloud forests. Evidently, land is transformed because of basic human needs of food and raw materials, as well as social and economic pressures. Thus, to assess the impact of cloud forest transformation into coffee or sugar cane plantations, we compared the non-market ESV estimated for cloud forests vs. the market gains of coffee and sugar cane production. According to our estimations, the added non-market value of the ecosystem services provided by cloud forests is as high as 728 USD/ha/year (2004 USD), whereas in 2004, coffee plantations yielded 384 USD/ha and sugar cane 2088 USD/ha (SAGARPA, 2004). Considering cloud forests vs. coffee plantations alone, current land use trends show that forest conversion into coffee plantations has lead to economic losses in terms of the ecosystem services, because the ESV of cloud forests is higher than the economic yields of coffee production. However, when comparing ESV of cloud forest vs. sugar cane yearly yield, it is obvious that sugar cane production has almost tripled its gains because of higher per hectare revenues. So, depending on land use choices, land use change alternatives may generate either higher or lower net revenues per ha than the value of ecosystem services estimated for cloud forests. Evidently, from a private owner's point of view the incentive to convert forest into cropland is very strong. Similar trends have been observed in Ecuador and Indonesia, where Olschewski et al. (2006) found that pollination services alone were hardly sufficient to reduce pressure on forest margins.

In this sense, Payments for Ecosystem Services (PES) can be an option to reduce pressure on forests (Olschewski et al., 2006; Kosoy et al., 2007; Muñoz-Piña et al., 2008). In the Mexican PES scheme, forest cover is considered as an adequate proxy to estimate ecosystem service provision (such as water quantity) (Diario Oficial de la and Nación, 2004), with the underlying supposition that more forests means more water. Similar to Mexico, this myth is being perpetuated by PES programs in many tropical countries (Wunder et al., 2008), at the expense of loosing additional ecosystem services and an increased deforestation risk when the expectations (amount of water) are not met by PES schemes.

The fact that Mexican PES schemes (PSAH) have focused on hydrological services provided by forests, especially because of severe water scarcity in many regions (Muñoz-Piña et al., 2008), and that additional ecosystem services are not considered, is an important omission. Certainly, PES alternatives could largely improve if the opportunity costs to convert forests as well as the costs of improving water quality were incorporated in the Mexican PES schemes for watersheds. In this sense, our valuation studies could provide guidance as to how much the opportunity cost of forest conversion vs. conservation is worth. For instance, Mexican PES compensation schemes proposed paying 27.3 USD/ha/year for preserved cloud forests. This value is much lower than the estimated non-market ESV of cloud forest and the opportunity cost of cloud forest transformation into coffee or sugar cane plantations. Therefore, if one of the aims of PES schemes is to preserve natural ecosystems (such as cloud forests) and their ecosystem services, then PES should be more adequately adjusted and include more ecosystem services.

4.1. Caveats of the study

The transfer value technique to assess the economic value of ecosystem services is just one among many methods used for the environmental valuation, and like most methodologies, there are benefits and limitations. Benefits of the transfer value method are that it can reduce both the cost of expensive environmental evaluation studies and the time required to assign values to ecosystem services in a given site (Troy and Wilson, 2006). This method also generates potential values of services and ecosystems at under-studied sites such as the state of Veracruz.

However, in spite of its advantages, the transfer value method has its own limitations. For instance, it assumes that the study site is sufficiently similar to site from where the economic value is being transferred, particularly in terms of consumer preferences and environmental quality. The technique also assumes similar market structures between the two sites (Ready and Navrud, 2006). That is, even though there is no set of standards that should be followed to perform the transfer value method, when possible, it is important to use studies from similar social, economic and environmental features, in order to have a more accurate valuation (Rosenberger and Stanley, 2006). We have considered the above, and as much as possible, in this study we have followed this criterion.

5. Conclusions

Our study showed that land use change affects biodiversity, structure and composition of natural communities, and that these changes affect ecosystem functioning and services, such as water quality. In addition, loss of natural ecosystems may also imply a significant economic loss to society in terms of ecosystem services. However, our estimations indicate that the market gains of land use change in the highlands of La Antigua watershed (for instance sugar cane production) do not foster forest conservation.

Thus, additional ecosystem services (i.e. water quality) need to be considered in PES schemes as a feasible option to forest conservation. Furthermore, the opportunity costs of land use conversion and ESV of natural ecosystems need to be considered if we aim at improving PES as alternatives aiming at forest conservation instead of land use change. This is particularly important in the context of the Mexican PES scheme, where water quality and additional ecosystem services are not considered. Our estimates are indicative of the urgent need to go beyond water quantity as most relevant ecosystem service considered in PSAH schemes.

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