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Concepts and a methodology for evaluating environmental services from trees of small farms in Chiapas, México

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ABSTRACT

We propose a methodology to estimate the environmental service (ES) value of small farms in Chiapas, Mexico, involving trained farmers-promoters in field sampling. We considered the ways in which the landscape's principal organisms, the trees, contribute to ES. We proposed a species functional value (FV) index based on their functional traits and key ecological characteristics, and estimated each site's ES value using FV weighted by the dimensions and abundance of individuals in different land uses (LU). Tree contribution to carbon storage (C) was defined using species wood density and biodiversity conservation value (BD) using food and habitat provision for wildlife and species existence (non-use) value (EX). Many species and individuals had high C, as wind-dispersed species with dense wood were common, but low BD prevailed, with high BD species common only in riparian forests. Few species and fewer individuals had high EX conferred by dense wood, large size, harvesting pressure and animal dispersal, among others. High variance in value within LU types, suggested that LU is a poor estimator of ES value, and that the measurement of species FV and tree dimensions is essential. This tool accurately reflects the ecological values of farm tree cover, allowing negotiation of compensation for environmental services. This methodology could be implemented combining open-access regional traits databases and field sampling by local people, and can easily be adapted for the measurement of other ES, and to other ecological and cultural contexts.

1. Introduction

In wide areas of the tropics, increasing tree cover to promote sustainability of the farmscape is a response to the drastic degradation and loss of forest cover and environmental resources (MEA, 2003). Tree components in agricultural production systems, besides contributing directly to production objectives, have the potential to offer a variety of environmental services (ES) (Harvey et al., 2006; Pagiola et al., 2007). A way to increase on-farm tree cover is through payment for the environmental services (PES) produced – thereby internalizing ecological externalities (Campos et al., 2005; Naeem et al., 2009; Redford and Adams, 2009). Considerable work is nevertheless required to develop methodologies for the estimation of environmental service value of tree cover on farms in the tropics that are both accurate and practical.

In this study, we develop a methodological tool for estimating tree species and farm tree cover ecological value as ES providers, sampling small cattle farms in the Lacandon Forest of Chiapas, México, to validate it. The Lacandon rainforest is one of the most important biodiversity regions of Mesoamerica (CONABIO, 2004); the landscape is however, fragmented by numerous small farms, and there is an urgent need for action for conservation and restoration by farm owners. In line with the CIFOR C&I Team (1999), we emphasize that we are not proposing an ideal and universally applicable tool, but a basic approach or template that can be modified and customized to comply with local conditions.

We take into account climate regulation through carbon capture and storage and biodiversity conservation, two of the most widely recognized ES in PES schemes applied in countries like Costa Rica, Nicaragua, and in the state of Chiapas, Mexico (Ibrahim et al., 2007; Pagiola et al., 2007; Soto-Pinto et al., 2007). Previous methodologies for estimating site ES values in tropical farmscapes are based on a *priori* expert land use rankings and/or the use of geographic information systems (Murgueitio et al., 2003; Dutton et al., 2010; Smukler et al., 2010; West et al., 2011). Instead, we use simple sampling techniques applied by trained local people to have direct field data, as we believe that their involvement in work like this,

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rather than being simply a means, is one of the purposes of a sustainable process (Leff et al., 2005).

Additionally, to estimate tree cover ES value, we take into account the different ecological value of tree species using functional traits and community functional properties, which has not been done in previous methodological proposals (Quétier et al., 2007; Martin and Blossey, 2009; Díaz et al., 2011). We believe that it is essential to give value to trees that are on farms for purely ecological reasons and not because they represent an economic or other type of benefit; as we found on many small farms in the Lacandon Forest and in other farmscapes.

We adapt the functional ecology approach (Díaz et al., 2007) to develop our methodological approach to estimating tree ecological value. We estimate tree species contribution to the carbon service on the basis of species wood density. We define tree species contribution to biodiversity conservation in terms of three "sub-services": food and habitat provision to wild fauna, and existence or intrinsic value of tree species. Existence value is a difficult concept to measure and is strongly culturally dependent (Groom et al., 2006), but we share the view of Naess (1986) that conservation value should not only be defined in utilitarian terms. Our methodology proposes to scale up from species traits to tree cover type and farm ecological value in three steps (cf. Díaz et al., 2007). First, we propose a species functional value index based on ecological characteristics and functional traits (hereafter, "traits"). Then we estimate the ecological value of each tree combining species functional value and individual tree dimensions. We sum individual tree values to obtain the value of each land use and finally, farm value for the ES taken into account. In spite of the apparent complexity of the mathematical equations used, the investigation's main idea focuses on the practicality of estimating something multifaceted, like tree contributions to the biodiversity conservation service, based on logical sequences and calculations.

2. Methods

2.1. Study area

The Lacandon rainforest in Chiapas, Mexico is one of the most important biodiversity regions of Mesoamerica (CONABIO, 2004). However, the landscape is fragmented by small farms typical of tropical lowland cattle farms in general (Harvey et al., 2006). The land uses (LU) studied on these farms were pasture with trees, fallows, riparian forests, live fences, coffee plantations and plantations with mixed trees.

The climate is tropical moist (Holdridge, 1982) with a mean annual temperature between 24 and 26 °C, unimodal rainfall with a mean annual precipitation between 1000 and 2000 mm, and a four month dry period from January to April. Natural vegetation is predominantly high evergreen forest with areas of medium evergreen forest (García, 1973). The most frequent types of disturbed or secondary vegetation are the fallows.

Fieldwork was carried out on 45 small cattle farms from three communities located in Montes Azules Biosphere Reserve and one in the buffer zone; between 16°14′–17′N and 91°12′–18′W, at 200 to 300 masl on the alluvial plain of the Lacantun River.

2.2. Data collection methods

We developed our methodology combining deskwork and fieldwork with local people's collaboration. We gave special emphasis to farmers' participation, the main stakeholders, involving them actively in parts of the study. Thus, they gained awareness of the relation between quantity and quality of trees and environmental services, were trained and employed for field sampling, and they transmitted generated information to the farmers of the studied plots. We trained ten local promoters in our study area through participatory workshops, one theoretical, and two practical, including the use of a promoter's manual.

Two of the promoters carried out fieldwork in each community. Every couple sampled all LU in ten or more cattle farms, chosen randomly among those owned by producers willing to cooperate. In each LU they measured trees $\geq$10 cm *dbh* (diameter at breast height, 1.3 m) an almost universally used minimum diameter limit that most likely captures the major contributions to the ES studied. A number of representative square plots of 20 m × 20 m (400 m^2) were established, summing approximately 10% of the size of the LU patch, except in areas with $\leq$30 trees ha^{-1} where a census was made. For LF and RF strips that did not reach 20 m width, they measured two 10 m wide plots. For each tree, the promoters provided a local species name, *dbh* in cm measured with a fiberglass tape, total height (H) in classes of 5 m estimated with the empirical similar triangles method using a fiberglass tape and a ruler, and crown diameter (Cd) in m measured on the ground below the crown using a fiberglass tape (for more details, see Marinidou, 2009).

2.3. Species functional value for each environmental service

We created the functional value (FV) indexes with scales of absolute values 0/1–100, to characterize tree species regarding their potential to generate the climate regulation (C) and biodiversity conservation (BD) services. We estimated FV using equations described below created by combining the average values of different traits classes, previously weighted by a value depending on the ES (Table 1). We determined the number of classes and values for each trait using the Núcleo DiverSus (http://www.nucleo.diversus.org) and Missouri Botanical Garden (http://www.tropicos.org) databases, along with scientific literature (Croat, 1978; Tamarit, 1996; Martínez and Martínez-Pinillos, 1996; Barajas et al., 1997; Bárcenas and Dávalos, 1999; Zavala, 2000; DOF, 2002; Cordero and Boshier, 2003; Pennington and Sarukhán, 2005; Chave et al., 2006; IUCN, 2011), and in a few cases, the knowledge of key local informants.

We proposed tree species valuation based on functional ecological assumptions that could require more validation effort. We do not seek to assign absolute values to species on a universal scale, but to rank objectively the species sampled in a particular area, so that species order is meaningful but not the exact differences between values, that is relative.

2.3.1. Species functional value for carbon storage and climate regulation

As proposed here, the methodology evaluates climate regulation with respect to aboveground tree biomass. Standing carbon stocks depend mainly on the abundance and size of the individuals present, and their wood density as most important functional feature (Brown, 1997). For that we estimated species functional value with respect to carbon storage (FVC) by using only their wood density ranked in five classes (Wd = 1 to 5): very light (<0.30 g cm^{-3}), light (0.30–0.44 g cm^{-3}), medium (0.45–0.59 g cm^{-3}), dense (0.60–0.74 g cm^{-3}) and very dense ($\geq$0.75 g cm^{-3}). We estimated species FVC weighting the average value of each class (Wd + 0.5) by a factor *i*, equal to the maximum value of the index (100) divided by the average value of maximum class (5.5), to bring it within the proposed scale (Eq. (1)).

$$FVC = (Wd + 0.5)*i \qquad (1)$$

Thus we obtained five values (FVC = 27, 46, 64, 82 and 100) corresponding to the categories already described.

Table 1
Species functional valuation.

Environmental service equations	Traits and attributes	Ranks	Weights	Criteria	Source of information
Carbon storage to aboveground tree biomass (Wd + 0.5) $*i$	Wd	1–5 => 1.5, 2.5, 3.5, 4.5, 5.5	$i = 100/5.5$	The average value of each class (Wd + 0.5) weighted by i to bring it within the proposed scale	Brown 1997
Food provision to wild fauna FVFD = $\{[\text{Fr} + \text{Sd} + \text{Fg} + \text{Fl} + (\text{SqFr} + \text{SqSd} + \text{SqFg} + \text{SqFl})*0.5]*i\}^{j} + 1$			$i = 4$ $j = 1.45$ +1	The sum of food types weighted by a factor i and raised to a power j, to show that increasing diversity of food types has an increasing value but diminishing distance To avoid zero values as no tree species is completely without food value to wild animals	Thiollay 1995; Foster 1990
	Fruit (Fr), seed (Sd), forage/foliage (Fg), floral (Fl)	0/1	1	Four types of same value resources	
	Dry season (SqFr, SqSd, SqFg, SqFl)	0/1	0.5	Key factor in the survival of wild fauna	Foster 1990
Habitat provision to wild fauna FVHB = $3^{\text{Ld}-1} * \text{Lph} *i$			$i = 100/27$	The two traits product is weighted by i to bring values to the proposed scale	Thiollay 1995; Faller-Menendez et al., 2005; Fink et al., 2009
	Crown leaf density (Ld)	1–3 => 1, 3, 9	$3^{\text{Ld}-1}$	Each class has three times the value of the one preceding	Pennington and Sarukhan 2005
	Leaf phenology (Lph)	1–3		Semi-deciduous and evergreen species have two and three times the value of deciduous species, respectively	
Existence value FVEX = $\{[a*(\text{Wd}+0.5) + b*(H_{max} - 0.5) + c*\text{Pr} + d*(\text{Ds}^{a}+1) + e*\text{Rp}]/i\}^{j}$	Native/introduced species[a]	0/1	Filter	Equation only for common native species, FVEX = 0 to introduced species	Martini et al., 1994; cf. Poorter et al., 2008
	Threat category[a]	0/1	Filter	FVEX = 100 to species in any threat category	
			$i = 2.27$ $j = 2.84$	The rest of trait values divided by maximum class value and weighted; the total sum divided by i and raised to j to obtain the desired scale	
	Wd	1–5 => 1.5, 2.5, 3.5, 4.5, 5.5	$a = 6.5/5.5$	Highest weight, estimator of species vital rates	Cornelissen et al., 2003
	H_{max}	1–3 => 0.5, 1.5, 2.5	$b = 2.5/2.5$	Considering the average values of classes ($H_{max} - 0.5$); high weight, estimator of species vital rate	Cornelissen et al., 2003
	Destructive pressure from local use (Pr)[a]	0–3	$c = 1.5/3$	Over-harvesting impact populations directly	Finegan and Delgado 2000;
	Dispersal type (Ds)	0–2 => 0, 2–3	$d = 0.66/3$	We applied (Ds + 1) so that the last category had only half value more than the previous one; low weight, quantitatively uncertain effect on species vulnerability	Reyna-Hurtado and Tanner 2005
	Reproductive system (Rp)	0/1	$e = 0.33$	Lowest weight, probable but quantitatively uncertain effect on species vulnerability	

[a] These are not functional traits but cultural dependent local attributes.

2.3.2. *Species functional value for biodiversity conservation*

Our methodology evaluates three biodiversity subservices, but here we propose an overall simplified index of species diversity value FVBD as their average value, by assuming that are all of the same importance.

$$\text{FVBD} = (\text{FVFD} + \text{FVHB} + \text{FVEX})/3 \quad (2)$$

Where FVFD is food value for wild fauna, FVHB is habitat value for wild fauna and FVEX is existence value.

For FVFD we used eight traits, provision of fruit (Fr), seed (Sd), forage/foliage (Fg) and floral resources (Fl) (Thiollay, 1995), weighted by adding one-half when provision occurs during the dry season (SqFr, SqSd, SqFg, SqFl), a key factor in the survival of wild fauna (Foster, 1990). All traits were scored on a binary scale (0: no food for vertebrates, or not offered during the dry season and 1: food offered).

To estimate FVFD we used two assumptions: that diversity of types of food has a positive but decreasing non-linear relationship to value and that no tree species is completely without food value to wild animals.

On a scale of absolute values from 1 to 100, we achieved that by weighting the diversity of different types of food with a factor $i = 4$ and raising it to a power $j = 1.446$; and we avoided zero values by adding one point (+1) in the model (Table 1).

$$\mathrm{FVFD} = \{[\mathrm{Fr} + \mathrm{Sd} + \mathrm{Fg} + \mathrm{Fl} + (\mathrm{SqFr} + \mathrm{SqSd} + \mathrm{SqFg} + \mathrm{SqFl})*0.5]*i\}^{j} + 1 \quad (3)$$

There were twelve FVFD classes. In approximate values, species which do not offer any type of food were 8.4 times less valuable than those that offer one, but the last were 2.5 times less valuable than those that offer two types; while, those that offer three types were only 1.4 times less valuable than those that offer four types of food. Species that offer all types of resources including dry season production, like *Inga jinicuil*, had FVFD = 100 and species offering no resources according to these criteria, like *Blepharidium guatemalense*, FVFD = 1.

Species value for habitat and connectivity provision (FVHB) combined the values of two traits: crown leaf density (Ld) and leaf phenology (Lph). These traits reflect the importance of crown foliage volume for bird diversity in agroforestry systems (Thiollay, 1995), its general importance to vertebrates (Faller-Menendez et al., 2005), and the influence of tree species crown and patch characteristics on bird visits (Fink et al., 2009). Crown density had an ordinal scale (1: open crown, to 3: dense crown) and phenology had three classes: deciduous (leafless $\geq$ 4 months) (1), semi-deciduous (leafless $\leq$ 3 months) (2), and evergreen (3) (Pennington and Sarukhan, 2005). Each density class had three times the value of the one preceding and semi-deciduous and evergreen species had two and three times the value of deciduous species, respectively. We again brought FVHB within the proposed scale of 1–100 by weighting the product of the two previous values by a factor *i*, equals the maximum value of the index (100) divided by the product of maximum class values (27) (Table 1).

$$\mathrm{FVHB} = 3^{\mathrm{Ld}-1}*\mathrm{Lph}*i \quad (4)$$

There were seven classes of habitat value with absolute values 4–100, where evergreen species with dense foliage like *Mangifera indica* had the maximum value (FVHB = 100) and deciduous species with open crowns like *Cordia alliodora* had the minimum value (FVHB = 4).

Trees, besides providing food and habitat to other species, also add value to biodiversity of a landsite by purely being there. For that, we refer to existence value, as one of the three components to evaluate trees' value for their contribution to biodiversity conservation. We characterized species existence value (FVEX) using three attributes linked to capacity to maintain populations locally (Martini et al., 1994; cf. Poorter et al., 2008): the native/introduced dichotomy, threat category and destructive pressure from local use (Pr); and four functional traits: Wd, maximum potential height reached (H_{max}), dispersal type (Ds) and reproductive system (Rp). FVEX = 0 was assigned to introduced species like *Mangifera indica* and the maximum FVEX = 100 to those in any threat category like *Astronium graveolens*. Native and common species were valued using the rest of the criteria. Destructive pressure was scored as the sum of preferences for wood products: without use (0), firewood (1), wood or other "destructive" uses (2), wood and firewood (3). For Wd we used the same categories as for the carbon service. For Hmax we categorized species in three classes: up to one-third of the maximum canopy height reached by the original forest (Pennington and Sarukhan, 2005; authors' observation) or 15 m (1), up to two-thirds or 30 m (2) and above 30 m (3); considering the distance of the average values between classes (H_{max} − 0.5). We categorized dispersal type in three classes, autochorous or anemochorous species (0), species dispersed by birds and mammals (1) and species only dispersed by medium and large-sized birds and mammals, due to the probably low densities and vulnerability of these dispersal agents in this farmscape (cf. Reyna-Hurtado and Tanner, 2005) (2). We applied (Ds + 1) to all except the first category, so that the last category had only half the value of the penultimate one. For reproductive system we used a binary classification (0: not dioecious/1: dioecious).

We calculated existence value dividing the value of each trait by the maximum class value and weighting it by a factor proportional to its importance. We gave the highest combined weights to Wd (a = 6.5/5.5) and H_{max} (b = 2.5/2.5) as estimators of species vital rates (Cornelissen et al., 2003), and to destructive pressure (c = 1.5/3) because over-harvesting can impact populations directly. Dispersal system and reproductive system were given lower weights (d = 0.66/3 and e = 0.33) because they have a probable but quantitatively uncertain effect on species vulnerability. We divided the sum of resulting values by a factor i = 2.27, which equals to the sum of minimum values, and weighted it by a factor j = 2.84, to again obtain the desired scale of absolute values 1–100 (Table 1).

$$\mathrm{FVEX} = \left\{\left[a*(\mathrm{Wd}+0.5) + b*(H_{\max}-0.5) + c*\mathrm{Pr} + d*\left(\mathrm{Ds}^{*}+1\right) + e*\mathrm{Rp}\right]/i\right\}^{j} \quad (5)$$

*If Ds = 0, then do not add 1.

Species with high destructive use pressure, of slow growth as indicated by high wood density, that reach the upper-middle canopy, and that need medium and large mammals and birds for their dispersal, like *Hymenaea courbaril*, are assigned high existence values using this approach (FVEX = 71). Autochorous, medium-sized species with low use pressure and low wood density, like *Cochlospermum* sp., are assigned the lowest values (FVEX = 2).

2.4. Tree ecological value estimation

Tree ecological value is species functional value (FV) weighted by a tree size index (Dim) which combines the categorical value of the dimensions (*dbh*, total height and crown diameter) for each ES (Eqs. (6b)–(9)). We used 20 cm classes for *dbh*; for height (H) categories the relation one-third, two-thirds and three-thirds of the 45 m, giving three classes; and for crown diameter (Cd) we formed four classes: 0–1.5 m (1), up to 5 m (2), up to 10 m (3) and above 10 m (4). For carbon, due to the importance of *dbh*, we used 10 cm classes.

2.4.1. Tree value related to carbon storage

We estimated C in tons from aboveground biomass (AGB) based on the tropical moist forest allometric equation provided by Chave et al. (2005) assuming a constant coefficient of trunk taper cf = 0.06, and the fraction C/AGB = 0.5 (Eq. (6a)).

$$C_{\mathrm{base}} = cf*(\pi/4*\mathrm{dbh}^2*\mathrm{H}*\mathrm{Wd})^{\beta}*(C/AGB)/1000 = i*dbh^2*H*\mathrm{Wd} \quad (6a)$$

Where dbh in cm, H in m, Wd in g cm^{-3}, β is a constant which we assumed = 1 and i is the product of all constants = $cf*\pi/4*(C/AGB)/1000$.

Therefore, in our model we replaced the three variables with species functional value and categorical dimensions. We weighted the species functional value for carbon (FVC) by c = 0.825/100 to bring it from the 1–100 scale to the real categorical Wd values. We used the two categorical dimensions of interest, in average values, *dbh* class for carbon (CatdbhC) and height class (CatH-0.5) weighted by a = 10 and b = 15 respectively, to reflect the true range between categories. In this case, the dimensional metric for carbon (DimC) is the product of the two dimensions, replaced in Eq. (6b).

$$C = i^*(a^*\text{CatdbhC})^2 {}^* b^*(\text{CatH} - 0.5)^* c^*\text{FVC} = i^*\text{DimC}^* c^*\text{FVC} \quad (6b)$$

2.4.2. Tree value related to biodiversity conservation services

To estimate tree value for food provision to wild fauna we took into account species functional value (FVFD) and the dimensional index for this service. As most of the studied LU are dedicated to cattle production, *dbh* is not always proportional to crown diameter due to some trees having been cut back and resprouted, or otherwise damaged. Thus, we constructed a tree size index for food provision (DimFD) based on the average values of both categorical dimensions, (Catdbh-1) for *dbh* and (CatCd) for crown diameter, that can affect it. As well, we assumed that tree seed and fruit production increases with both dimensions but may reach maximum values before maximum size is reached (Snook et al., 2005; Wadt et al., 2005); we therefore created the maximum capacity for food provision tree size index (DimFDmaxcap) so that penultimate classes already reach maximum values (Table 2).

$$\text{FD} = \text{DimFDmaxcap}^*\text{FVFD} \quad (7)$$

For habitat and connectivity provision, we used a tree size index (DimHB) considering Cd as a metric of the crown volume for wildlife shelter (Fink et al., 2009) and H because it influences the movements of species that prefer particular canopy height ranges (Thiollay, 1995). We gave a cubic relation to tree measures, quadratic for Cd multiplied to the lineal for H, reasoning that the greater the DimHB, the larger the tree and the greater the habitat value (Table 2). As FVHB starts in a higher value than the other two services (4), we weighted by $i = 1/4$ to bring it to a similar scale.

$$\text{HB} = i^*\text{DimHB}^*\text{FVHB} \quad (8)$$

The same way, for tree existence value we used the related species value (FVEX) and tree size index (DimEX). We constructed DimEx based on the averages values of *dbh* and H classes: (Catdbh − 1) and (CatH − 0.5), the first was squared so that their product reflected tree volume. However, we used the square root of this result to have a more lineal or a less size influence effect than in estimating tree value for carbon storage.

We further created a maximum capacity for existence value index DimEXmaxcap considering that individuals attain the maximum existence value before the final developmental stages of a tree's life (Table 2).

$$\text{EX} = \text{DimEXmaxcap}^*\text{FVEX} \quad (9)$$

We considered tree value related to biodiversity conservation as an average of the three subservices (Eqs. (7)–(9)).

$$\text{BD} = (\text{FD} + \text{HB} + \text{EX})/3 \quad (10)$$

2.5. Tree cover ecological value

Our methodological proposal estimates tree cover ecological value by summing the values of individual trees. The exact form the relationship could take – linear or some other – could be a subject for discussion, but we consider the linear relationship to be most parsimonious. Thus, having plot tree cover values for each ES, we extrapolated them to the total area per LU and then calculated each farm's ecological value by summing the values of the LU on it.

2.6. Analysis and comparison of trees and sites ecological value

To compare with previous studies and for to offer the bases for comparison with later ones, we analyzed the influence of the dimensional and functional components to the individual and tree cover ecological value relative to each ES by means of Spearman correlations using InfoStat software (Di Rienzo et al., 2009) and visually with regression graphs. In order to compare between LU, we extrapolated the values of the different sampling plots sizes to 1 ha. This comparison gave us a general panorama of LU potential to generate ES but note that in the study area, LU like riparian forest or live fences hardly ever reach this size (1 ha or 1000 m length for live fences). With these values, we carried out descriptive analysis of the dimensional and ecological values with Box-Plot graphs.

3. Results

3.1. General characteristics, structure and diversity of the sites and their tree cover

The following overview of our results sets the scene for the subsequent characterization of tree cover ecological value. The 45 study farms had a total area of 212.25 ha with an average size of 4.7 ha, a range of 1.5–20 ha, and with 1–5 different LU each. As we gave preference to working with cattle farmers, predominant LU found were pastures with trees (78.8%), of which 63.3% had more than 30 trees ha^{-1} (PT+) and the rest had ≤ 30 trees (PT−). The only types of forest found, riparian forest remnants (RF) and fallows (FA) made up the 7.5%. Coffee plantations with shade (CF) and plantations with mixed trees (PL) made up the 2.2%, live fences with a variety of tree species (LF) made up the 5.6%, while 5.8% were LU without trees ≥ 10 cm *dbh* (PW). Sample sizes were small for FA, CF ($n = 3$ for both) and PL ($n = 1$).

Comparing LU tree cover, RF showed the highest averages per farm, although also the greater variation in all the structural components (tree density, basal area, crown height and crown area) (Fig. 1). In most of the farms, FA and CF followed as the most extensive cover types and after them PT+ and LF with similar distributions; PT− showed the lowest values.

We measured 2094 trees, the great majority (71%) with *dbh* < 25 cm, represented by 130 species and 41 families. The three most abundant families were Papilionaceae, Anacardiaceae and

Table 2
Individuals' dimensional metrics (Dim) for the estimation of different biodiversity conservation services.

Service	Metric (Dim)	Dimdbh[a]	DimH	DimCd	Product
Food provision	FD	Catdbh − 1	–	CatCd	Dimdbh * DimCd
	FDmaxcap				1.45*$DimFD$ * (1 − DimFD/50)
Habitat provision	HB	–	$CatH - 0.5$	CatCd^2	DimH * DimCd
Existence value	EX	$(\text{Catdbh} - 1)^2$	$CatH - 0.5$	–	$(\text{Dimdbh} * \text{DimH})^{0.5}$
	EXmaxcap				0.02*$DimEX$ * (1 − DimEX/60)

Note 1: Abbreviations Catdbh, CatH and CatCd refer to the categorical value of each dimension, *dbh* class, height class and crown diameter class, respectively.
Note 2: The abbreviation maxcap refers to the created scales, for which trees reach maximum capacity of service already at the penultimate classes of DimFD and DimEX metrics.
[a] To avoid zero values for the first category of Catdbh we used a filter to bring it to 0.5 times of the subsequent class value.

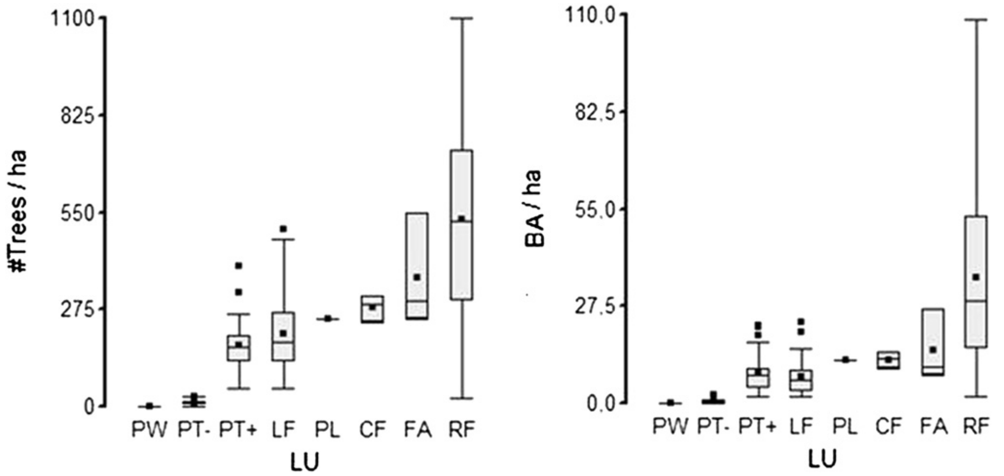

Fig. 1. Density and basal area (BA in m^2) of trees in different land uses (LU). Symbols at the bars show the mean and median (black point and perpendicular line in the box), 0.25 and 0.75 quartiles (lower and superior limit of the box), 0.05, and 0.95 quintiles (vertical lines limits) and extreme values (individual points). PW: LU without any presence of trees with *dbh* ≥ 10 cm, PT−: pastures ≤ 30 trees, PT+: pastures with >30 trees, LF: live fences, PL: plantation with mixed trees, CF: coffee plantations with shade, FA: fallows, RF: riparian forests.

Rubiaceae and the six most abundant genera were *Spondias*, *Lonchocarpus*, *Blepharidium*, *Inga*, *Cordia* and *Bursera*, summing approximately 50% of the total. Due to their abundances, the species *Spondias* sp. and *Lonchocarpus guatemalensis* showed the highest basal area with almost 22% of the total observed.

3.2. Species functional value for environmental services

3.2.1. Carbon

Tree functional value for carbon (FVC) had a clear differentiation into five classes, because it was based on the categorical values of only one trait: wood density (Eq. (1)). The very dense wood class was characterized by species of the Sapindaceae and Sapotaceae families, as well as some Caesalpinaceae, Myrtaceae, Papilionaceae and Rosaceae. At the other extreme, the very light wood category was composed of species of Bombacaceae, Caricaceae and Cochlospermaceae. In spite of the degree of human intervention in the landscape, the highest proportions of species in the sampled plots were of dense wood followed by those with medium wood density; this result is due to the presence of individuals of many species in the families mentioned in the riparian forests but also in pastures with trees (Fig. 2a). The frequency of FVC types by individuals changed slightly, dense wood showing the greater value because wind-dispersed species in this category like *L. guatemalensis* and *Blepharidium* sp. were common in disturbed areas (Fig. 2a).

3.2.2. Biodiversity

Most of the species found showed low food provision value (FVFD) (1–14), although by abundance high value stands out because *Spondias* spp. and *Inga* spp. were common (Fig. 2b). Other species that showed high value were those of the genera *Brosimum* and *Guarea* and the introduced *M. indica*. The minimum (FVFD = 1) was shown by species of Bignoniaceae, Rubiaceae, Papilionaceae and Tiliaceae families and the genera *Astronium*, *Cordia*, *Sapindus*, *Sebastiania* and *Terminalia*, among others, in which wind dispersed seeds unpalatable to frugivores predominate.

31% of sampled species showed the maximum habitat value (FVHB = 100), being evergreen and of dense foliage (Eq. (4)), many belonged to the Annonaceae, Arecaceae, Magnoliaceae, Moraceae, Nyssaceae and Sapotaceae families (Fig. 2c). However, by individuals, low value trees (FVHB = 4–22) predominated, belonging to open-canopied and often deciduous species of Bignoniaceae, Caricaceae, Cecropiaceae and Cochlospermaceae, and some in Bombacaceae, Boraginaceae, Burseraceae, Euphorbiaceae and Mimosaceae (Fig. 2c).

In order to observe better the distribution of existence values (FVEX) we show the average of value classes grouped into intervals of 10 points, except the first and two last classes. Few species had high existence values and these represented an even smaller percentage of the sampled individuals (Eq. (5), Fig. 2d). Besides seven threatened species that obtained the maximum value, several species of Sapotaceae, as well as *Dialium guianense* and *Hymenaea courbaril* (both Caesalpinaceae) had high values conferred by dense wood, large size, dispersal by medium-sized or large vertebrates, and pressure from destructive use. Species with intermediate or low values prevailed. Aside from the seven introduced species with FVEX = 0, low values were showed by species with light wood, H_{max} <15 m, with no pressure for destructive use, and dispersed by birds and small mammals. Low values were assigned even to dioecious species, like *Carica* sp., or those used only for firewood and autochorous, like the common pioneer *Cochlospermum* sp.

For the integrative value for biodiversity conservation (FVBD) (Eq. (2)), we also grouped values within intervals of 10 points each, showing the average value formed classes. The frequency distributions of FVBD differed markedly from those for FVC. While this cattle farm tree flora has a significant proportion of both species and individuals with intermediate to high potential for the carbon service, most species showed low values for biodiversity and as a proportion of the total number of individuals, the predominance of low values increased. High value biodiversity species like *Brosimum alicastrum*, *Manilkara staminodella* and *H. courbaril* were only common in riparian forests. The lowest FVBD were shown by species traditionally considered pioneers, wind dispersed, with open canopies and low food resources provision, like *Bernoullia flammea*, *Trichospermum mexicanum*, *Cordia alliodora* and *Heliocarpus appendiculatus* (Fig. 2e).

3.3. The ecological values of different land uses and whole farms

The biggest estimated average amount of carbon stored in the tree cover aerial biomass per unit area (1 ha) was found in river forests (RF) with an estimated 85 ton C ha^{-1}. They were followed by fallows (FA) with half of their value, coffee plantation (CF) with

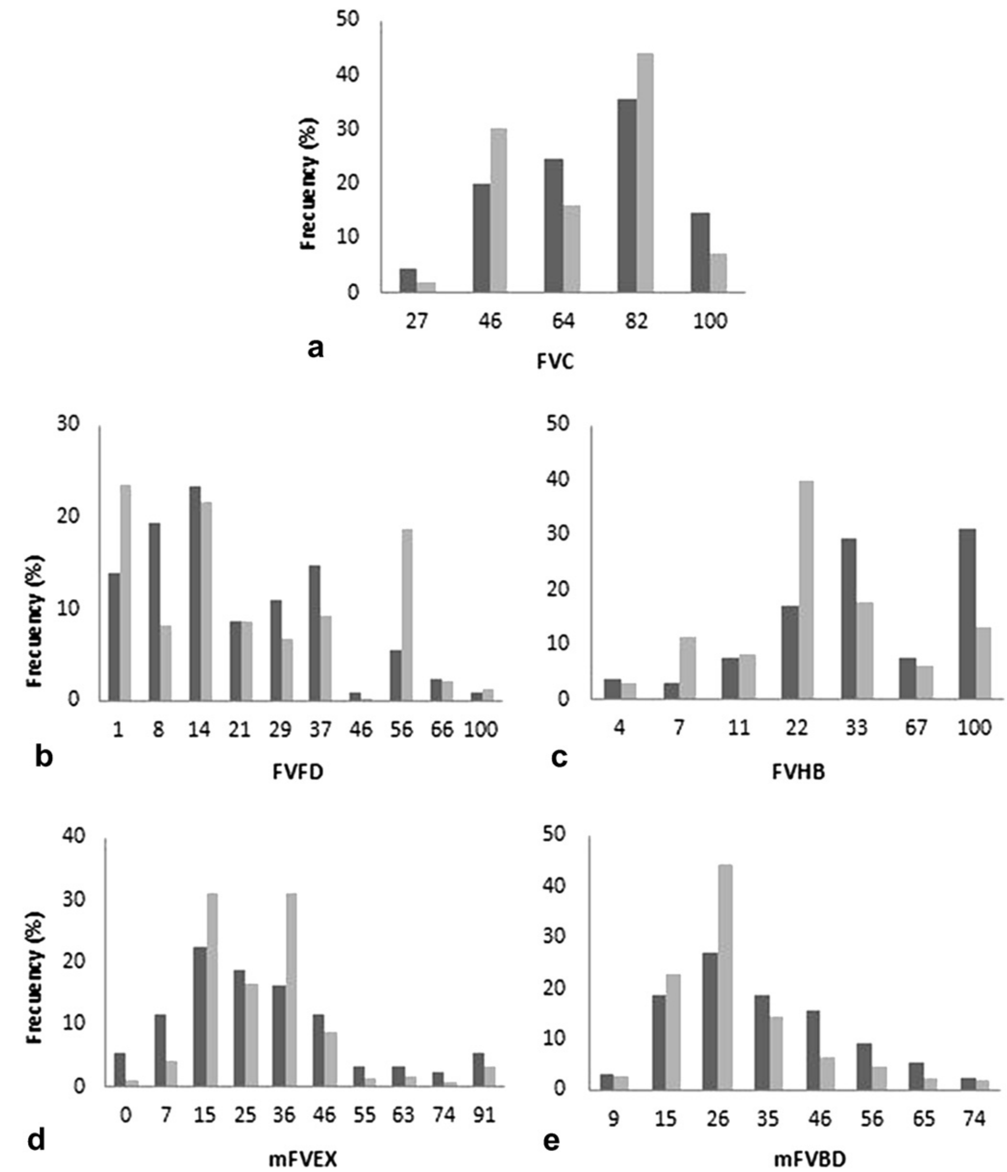

Fig. 2. Percentage of a) FVC types (species functional value related to carbon stocks), b–e) FVFD, FVHB, mFVEX and mFVBD types (species functional value related to food and habitat provision, existence value and biodiversity integrated value respectively, the last two in grouped categories showing the mean value); within the proposed scales 0/1–100, for total of species (dark color bars) and for total of individuals (light color bars) in the study area.

a quarter of their value, pastures with more than 30 trees per plot (PT+) with a sixth of their value, mixed tree plantations (PL) and live fences (LF) with almost an eighth of their value, and at the end low tree density pastures (PT−) with only one tenth of LF value (Fig. 3b).

For the three biodiversity subservices, RF showed the greatest average value per unit area (1 ha), although with wide ranges and asymmetric distributions of data. By order of value followed, for the food service CF with a small range, and FA for the habitat service. PT+ and LF showed similar values, but lower than those of CF and FA. Biodiversity values for PT−, as expected, were very low (Fig. 3a, b).

The range of tree cover values for stored carbon among the studied farms was 1–259 (mean = 59) ton C, and that of value for biodiversity was 9 − 1148 (mean = 345) BD points. By unit area (1 ha), tree cover values ranged between 0.6 and 69.5 (mean = 13.4) ton C, and 5.7 and 305.6 (mean = 75.6) BD points. Low values for both services were shown by farms of low tree density pastures (55–340 trees ha^{-1}); while the highest values per unit area for both ES were shown by small farms from 2 to 3.5 ha, of high tree density pastures (more than 500 trees and 30 m^2 of basal area, approximately). LF and RF added little to the total value of these farms, mainly for their low extent, 5–10% of the total area.

4. Discussion

4.1. Considerations on the proposed methodology

Our approach to ES evaluation requires more fieldwork than those that use Geographic Information Systems to distinguish between each LU (Murgueitio et al., 2003). However, we consider it essential for the recognition of the true ecological value of trees on small farms. In addition, as in other studies, the inclusion of local people strengthens local capacities (Jiménez-Ferrer et al., 2007) and creates employment contributing to their income, while diminishing costs and dependency on non-local personnel (Kainer

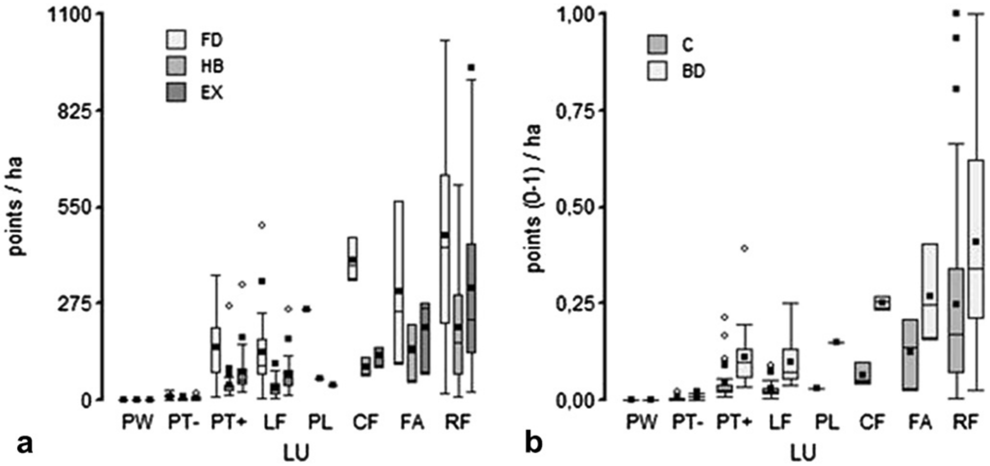

Fig. 3. Value distribution in the different land uses a) for the three services that integrate biodiversity conservation service: food (FD) and habitat (HB) provision, and existence value (EX); b) for biodiversity conservation as a combined value (BD) and for carbon (C), in a scale 0–1 to compare. For explanation of abbreviations, see Fig. 1.

et al., 2009). Moreover, training is an investment in local capacity, supports a more equitable sharing of intellectual power and is probably more feasible for effective implementation of research results (Garnett et al., 2009). The use of categorical values for traits instead of the continuous scale variables (Cornelissen et al., 2003) facilitates the estimations; for example, the wood density categories we used are recognized by promoters as well as in local literature. On the other hand, the involvement of local people brought a tradeoff in loss of statistical rigor, as we did not predefine the number of farms, their size or the distances among them (Kainer et al., 2009) but is expected to promote the emergence and scaling up and out of ecological knowledge (Leff et al., 2005; Garnett et al., 2009).

The wide ranges and asymmetric distributions of our data are a key result; questioning the value of LU as a category for ES value estimation because of the considerable overlap among them, and reinforcing the need for field data. Similarly, a methodology that does not consider species traits would not capture either the potentially high C value of disturbed areas conferred by high wood density, wind dispersed, pioneer tree species, or the higher overall value for C than for BD shown by this tree flora. By implication, the insight gained from assessing the contributions of organisms to ES value using species functional values and plant dimensions is essential and should be sought for a wide range of community types (Luck et al., 2009).

4.2. Overview of tree taxonomic and functional diversity on the small farms

Mesoamerican cattle farms typically have a diversity of LU with tree components, whose composition reflects a transition from natural forest to a human-dominated landscape (Harvey et al., 2006). Thus, the tree component of the sampled farms was made up of remnant species from the original forest (*B. alicastrum*, *Platymiscium* spp.), those introduced by people (*M. indica*), species with desirable characteristics whose natural regeneration is favored (*C. alliodora*), or pioneers (*Cochlospermum vitifolium*, *Guazuma ulmifolia*, *L. guatemalensis* and *Spondias* spp.) which regenerate quickly under the prevailing farm management regimes and are common in farmscapes (Finegan and Nasi, 2004). Certain ranges of values of traits are likely to prevail in each of these groups of species, so that ES value is shaped by the interplay between human activity and species characteristics (cf. Luck et al., 2009; Díaz et al., 2011). For example, trees in the dense Wd category were the most abundant overall, due to the capacity of *L. guatemalensis* and *Blepharidium* sp. to establish and grow in disturbed areas. This combination of trait values contributes to species and LU value for the carbon service being comparatively high, while meaning at the same time that values for biodiversity are low, because the species are poor for food and habitat provision.

4.3. Overview of the carbon model

The carbon service model, reached similar results to those from formal research in the region. Aboveground tree cover carbon stocks ranged in averaged values between 84.9 ton C ha^{-1} in river forests to 1.3 in pastures with low tree density. Carbon in river forests was just inferior to the 114 ton C ha^{-1} reported for tropical lowland secondary forests in Chiapas (Castillo-Santiago et al., 2007). Carbon in fallows (42.5 ton C ha^{-1}) was between those reported for traditional and improved fallows in Chiapas (32.2 and 63 ton C ha^{-1} respectively, Soto-Pinto et al., 2007). The coffee plantations showed similar values (22.1 ton C ha^{-1}) to diversified shade coffee plantations in Costa Rica (25 ton C ha^{-1}, Polzot, 2004), though lower than other from Chiapas (36 ton C ha^{-1}, Soto-Pinto et al., 2007). The pastures with more than 30 trees showed greater value (14.4 ton C ha^{-1}) than the reported for silvopastoral systems or monospecific shade coffee plantations (10.1 and 8.2 ton C ha^{-1} respectively, Avila et al., 2001), while live fences showed similar value (9.9 ton C ha^{-1}).

We only considered carbon stored in aboveground tree biomass, with lower values compared to those of studies that include carbon in root biomass. Root biomass could be inferred adding approximately 25% to the original value of carbon stored in the aerial biomass (Cairns et al., 1997). Nevertheless, this would not change the rankings of sites in the carbon scale we proposed.

4.4. Overview of the biodiversity model

The tree cover value in sample plots for the three “sub-services” that compose the biodiversity conservation service (BD) – food (FD), habitat (HB) and existence value (EX) – was most correlated with the sum of species functional value (FV) for each. There are no studies with which we can compare the proposed values for food and habitat that trees provide, or their existence value, but analysis was performed with the information found on tree dimensions and tree cover dimensional metrics, examining each service that composes BD separately.

Our methodology proposes that potential food provision for vertebrates should be estimated on the basis of *dbh* and crown diameter as well as species FV for this service. The decision to estimate a tree's FD relying on these two dimensional metric is supported by, among others, studies in fruit production of mahogany (*Swietenia macrophylla*) and Brazil nut (*Bertholletia excelsa*) in Quintana Roo, Mexico and Brazil, that increased with *dbh* and crown area (Snook et al., 2005; Wadt et al., 2005).

Tree cover value for HB provision had greater correlation – aside from species FV that was the most influential variable – with basal area and crown area. This result, as well as the use of crown diameter and height in the habitat model, parallels those of studies of live fences in pasturelands in the lowland and middle elevation moist tropics of Costa Rica, where the number of individuals and species of birds observed were strongly positively related to tree *dbh*, height and crown area (Lang et al., 2003; Fink et al., 2009).

Tree cover EX was most strongly related to *dbh* among the trees dimensions we measured, having a greater correlation with basal area than with number of individuals. We know no published work on the existence value of tropical tree species. Our approach is ecocentric in assigning a priori high existence value to threatened species and zero value to introduced species. Also, in using traits and dimensions that give high existence value to large trees of species strongly dependent on fauna for their dispersal, probably of slow growth as indicated by high Wd, and possibly with little current natural regeneration (Martínez-Garza and Howe, 2003; Poorter et al., 2008).

In general, trees with low values for BD predominated in the sample plots and if farms in the coming years were managed to increase tree cover, these species will probably become more common in the landscape. Conversely, at some point in the future the survival of high FVBD species in the human landscape may require planting (Martínez-Garza and Howe, 2003).

PES schemes based on this type of estimation of BD value would have to be implemented through a willingness-to-pay or similar approach (Dutton et al., 2010), as it takes into account existence value and this is highly personal and subjective (Groom et al., 2006). We consider that the estimation of ES value based on areal extent of different cover types (Murgueitio et al., 2003) does not capture this critically important biodiversity component, nor can it give incentives to farmers awarded on this basis to conserve these high FVBD species (cf. Dutton et al., 2010).

5. Conclusions and recommendations

Through local promoters' inclusion in the study, they acquired new knowledge and skills and gained direct economic benefits; moreover, they could have possibilities for future work. Sampling methods for estimating the ecological value of farms requires an initial investment in capacity building. However, the direct measurement of trees generates reliable results on each tree cover value, which should contribute to fairer environmental services payments. Based on the above, the wide application of local people's participation would not only facilitate estimations of the environmental services offered by farms' tree cover, but also encourage the increase of these services, a product of both environmental services payment stimulus and environmental awareness.

The use of the areas of different cover types on farms as a criterion for rewarding the supply of environmental services does not take into account the complexity of tree communities revealed by our approach. Tree species ecological-functional value indicates their potential contribution to an environmental service, useful as a criterion for species selection to promote a required service. However, environmental services differ also with respect to tree dimensions and community characteristics that most influence their potential. Farms value for biodiversity conservation had nearly equal correlation with the three components, sum of the species functional value, basal area and number of individuals; nevertheless, the higher the tree density and basal area, the greater the error if tree functional value is not taken into account. This study made an ecological evaluation of 140 tree species and following this method is easy to estimate values for many more species. Moreover, practitioners can easily adopt this general approach for other services, simply requiring the selection and measurement of appropriate traits and tree dimensions. For the adaptation or improvement of estimation models, especially of food and existence value services for biodiversity, a functional diversity index could be used to weight the value of the plot with more diversity of food resources and functional types.

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Integrity of research and reporting

Experiments comply with the current laws of México, the country in which they were performed. In addition, the authors declare that they have no conflict of interest.

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