Title: Shifting cultivation in the context of REDD+: a case study of Mexican tropical dry forest

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Abstract:

The article considers the relation of shifting cultivation to deforestation and degradation, and hence its impacts in terms of carbon emissions. There is a need to understand this relationship better in the context of designing strategies for international policy on Reduced Emissions from Deforestation and forest Degradation (REDD+). The article reviews the way in which shifting cultivation has been incorporated in global and national estimations of carbon emissions, and assembles the available information on shifting cultivation in Tropical Dry Forests (TDF) in Mexico, where it is widely practiced. It then takes the case of two specific villages, Tonaya and Temazcal, which lie within the basin of the River Ayuquila in Jalisco, Mexico, using field data to demonstrate the typical carbon stocks and fluxes that are associated with shifting cultivation in this dry tropical forest area, and comparing these with stocks and fluxes associated with more intensive agricultural production and the shortening and potential lengthening of the fallow cycles in the same villages. We find that per ton of maize produced, the overall emissions from shifting cultivation are higher than those from permanent cultivation, although we did not take into account the additional emissions from inputs such as fertilizers and pesticides associated with permanent agriculture. However we find that shortening of the fallow cycle, which is occurring in the study
area as a result of various government subsidies, results in higher remaining stocks of carbon and lower emissions at the landscape level.

**Keywords:** Shifting cultivation; slash-and-burn; swidden cultivation; tropical dry forest; local livelihoods; carbon stocks; rates of carbon accumulation; agricultural intervention; REDD+; land sparing; Borlaug hypothesis
Under REDD+, shifting cultivation should be considered degradation rather than deforestation.

Overall emissions from shifting cultivation are higher than those from permanent cultivation, but this does not take into consideration inputs in the form of energy and fertilizer, which are much higher in permanent cultivation.

Lengthening of fallow cycles does not in most cases result in increased carbon stocks at the landscape level, and it increases the emissions per ton of maize produced.

When fallow cycles are shortened as a result of responses to policy rather than to population pressure, they result in higher carbon stocks at the landscape level.
1. Introduction

Shifting cultivation (SC), also known as swidden cultivation or slash and burn, is a traditional agricultural system which is still widely used, and studied, in tropical countries (Grigg, 1974; Manshard, 1974; Ruthenberg et al., 1980; Collinson, 1983; Peters and Neuenschwander, 1988; Silva et al., 2011; Aweto, 2013; Nigh and Diemont, 2013). SC is a continuous, cyclic agricultural system, characterized by a rotation between cultivated plots and regenerating fallow/secondary forest; the way in which it is carried out varies from place to place (Kass et al., 1993; Kleinman et al., 1995; Aweto, 2013a). In general, forest or secondary vegetation is cut, the debris is burned, and a cultivation phase of two or three years starts, with very limited use of pesticides or fertilizers. After a few years of cultivation the yield falls and the farmer moves on, clearing new land. The old cultivation plot is fallowed, leaving the vegetation to recover and allowing secondary forest re-growth. Shifting cultivators often return to re-cultivate the fallowed plot some time later; the rest period may range from 5 to 50 years).

During the colonial period, shifting cultivation was generally considered ´wasteful´ and ´primitive´ (Geertz, 1963; Spencer, 1966; other such attitudes are mentioned in Dove, 1983; Jarosz, 1993; Kuchelmeister, 1993). The term ´slash and burn´ originates from this period and the choice of words has a clearly negative connotation. Despite the fact that during the last 25 years a large number of scientific articles have vigorously contested the negative effect of shifting cultivation (Rambo, 1990; Dhakal, 2000; Fox et al., 2000; Ickowitz, 2006; but see also earlier re–assessments such as Schultz, 1964), this negative point of view is still very common.

In the context of climate change, shifting cultivation has been blamed as a leading cause of tropical forest cover loss (Myers, 1992; 1993; Angelsen, 1995; Geist and Lambin, 2001; 2002) and for the associated carbon emissions (Houghton et al., 2003; Houghton and Goodale, 2004; Nigh and Diemont, 2013), and this message is reflected in many national policy documents relating to mitigation (see for example the R-PPs and ER-PINs of Congo, Ghana, and Dominican Republic
submitted to the Forest Carbon Partnership Facility of the World Bank (www.forestcarbonpartnership.org).

The aim of this paper is to consider the impact of shifting cultivation on carbon stocks and fluxes in the context of the policy on Reduced Emissions from Deforestation and forest Degradation (REDD+), and to compare this with alternative forms of food production (maize), under the assumption that this production must continue in one form or another. We focus on one particular ecosystem, tropical deciduous forest, typified for the case of Mexico by *selva baja caducifolia* and commonly known as tropical dry forest (TDF). In other parts of the world, similar formations are called by other names such as *cerrado* (Brazil) and *miombo* (East Africa). We examine the impact of SC for maize in TDF at the community scale in Mexico, based on a detailed study in two villages, El Temazcal and Tonaya, in Jalisco state, Mexico.

We set out to answer two questions: (1) per ton of maize produced, how much carbon is emitted in shifting cultivation systems as compared to a permanent cultivation system and what are the remaining carbon stock levels in each case, and (2) what would be the effects on carbon emission rates and stocks, of shortening or lengthening the fallow period. While the answers to these questions may be of immediate relevance to policy makers designing interventions for national REDD+ programmes, the issue also has bearing on more general considerations. The first question relates to the Borlaug hypothesis (Angelsen and Kaimowitz, 2001; Rudel et al., 2009; Lobell et al., 2013), which posits that switching to sedentary agriculture (e.g. more intensive cultivation with higher yield per unit area) will ‘spare’ more forest for conservation purposes. The second relates to the commonly held view that increasing fallow lengths will enhance biodiversity as well as carbon stocks (Read and Lawrence, 2003; Lebrija–Trejos et al., 2008; Raharimalala et al., 2010).
Following this introduction, we first review some of the literature on carbon stocks under SC. In section 3 we explain how deforestation and degradation are understood in UNFCCC policy on REDD+, and what is meant by Tropical Dry Forest (TDF). In section 4 we deal with how emissions from shifting cultivation have been estimated at global level and in Greenhouse Gas Inventories for Mexico. Section 5 summarizes a number of detailed studies of shifting cultivation in Tropical Dry Forest (TDF) in Mexico, where literature has focused particularly on the Yucatán peninsula, and Chamela, in Jalisco. In Section 6 we describe conditions in Tonaya and Temazcal, two villages within the basin of the River Ayuquila in Jalisco, which we take as the study case and in 7 we explain the methodology we use for estimating carbon stocks and emissions related to shifting cultivation. Section 8 presents the results and conclusions are drawn in section 8.

2. Carbon stocks in SC systems

While a number of studies have discussed possible schemes, challenges and opportunities for SC in the context of reducing CO₂ emissions and increasing carbon sequestration (Gibbs et al., 2007; Mertz, 2009; Hett et al., 2012; Antunes et al., 2013; Orihuela-Belmonte et al., 2013, Aryal et al., 2014), there have been very few studies quantifying the emissions from of SC in the context of REDD+ (Mertz, 2009). It is obvious that during clearance, there will be emissions, not only as a result of loss of the woody vegetation but also because carbon concentrations in soil are lowered as a result of soil oxidation and because the litter supply to the soil is temporarily disrupted. There may perhaps be an increase of carbon stocks in the cultivation phase and a concentrated recovery of carbon stocks would be expected in the fallow phase due to accumulation of organic matter (Antunes et al., 2013), although the time needed to reach the original stock levels may vary greatly (Lebrija–Trejos et al., 2008; Vargas et al., 2008). For example Detwiler and Hall (1988) found a loss of 18% in soil carbon (including both mineral and soil organic carbon, SOC) during the clearance phase, and Don et al., (2011) found increases of between 38 and 50% in SOC when the land is in the fallow or secondary
succession stage, between 7 and 37 years after cultivation. There is little doubt about the general nature of these dynamics, though of course the specifics vary from location to location.

What is more in doubt are the effects of different fallow lengths on carbon stocks. It is commonly suggested that longer fallow lengths are associated with higher levels of above ground biomass and higher concentrations of soil carbon (Read and Lawrence, 2003; Lebrija–Trejos et al., 2008; Raharimalala et al., 2010). We argue that both a space and a time frame are needed to analyze this properly.

Under a shifting cultivation regime, the landscape is a mosaic of forests of different ages in a continuum of forest succession (Aweto, 2013). Shortening the SC cycles will usually result in lower average above ground biomass (Lawrence et al., 2010; Schmook, 2010) in the areas which are included in the cycle, but we argue that if at the same time this means that other areas, that were earlier part of the cultivation cycle, are abandoned completely, then it is quite possible that over the entire area, average carbon stocks will rise. What matters therefore is whether cycles are shortened as a result of population pressure and the need to bring a higher proportion of the total surface area into cultivation at any one time, or whether they are shortened as a result of policy measures, such as subsidies which encourage this. The case of Programa de Apoyos Directos al Campo (PROCAMPO), a grant paid to small farmers all over Mexico, is an example. Farmers may register land for this scheme, but the money is only paid provided the land is under cultivation or pasture. In the first two or three years of fallow following the cultivation period the land may be counted as pasture because re-growing trees have not yet reached an height of more than a couple of meters, and the area is used for grazing. Once the trees become taller and denser the area will no longer be eligible for the subsidy. Since registration of land was on a one-off basis, it is in farmers´ financial interest to bring the plot back into cultivation after three years of fallow. In addition to PROCAMPO, farmers have in recent years been able to obtain subsidies for inputs such as fertilizers and herbicides, which may reduce their dependence on the natural recuperative processes of long fallows.
Another factor to be considered is that the rate at which carbon is captured by re-growth in young secondary forests, which in the first years may be higher than the capture rates in older secondary and primary forests. Hence continuous ‘harvesting’ of the trees, such that the average tree population remains youthful, may result in higher average annual uptake of carbon. Recent research in humid tropical forests (rainforest in the Maya region, Amazon and in Panama) has suggested that shifting cultivation is much less damaging to carbon stocks than had been earlier thought (Pelletier, 2012; Nigh and Diemont, 2013). Whether the new plot is cut in previously un-touched (primary) forest or in secondary forest in recuperation may however have important consequences for CO$_2$ emissions and carbon sequestration (Detwiler and Hall, 1988; Fukushima et al., 2008; Lawrence et al., 2010).

Clearly, there are different pathways of vegetation change in tropical dry forest succession (Romero-Duque et al., 2007; Lebrija–Trejos et al., 2010), and although cyclical use of forest resources can result in a steady state with sustainable, if lower than ‘intact’ carbon stocks (Lawrence et al., 2010; Antunes et al., 2013), it could also result in a steady loss of stocks as a result of continuous over-exploitation from additional activities, such as timber harvesting, fire wood collection and grazing (Morales-Barquero et al., 2014) or because of increasing pressure of shifting cultivation in the sense outlined above (Eaton and Lawrence, 2009, Lawrence et al., 2010). What needs to be stressed is that the spatial–cyclical character of SC necessitates analysis at the level of the management unit (i.e. the whole area used in the cycle), not the pixel or the individual patch currently under cultivation, and the time horizon should be long (e.g. 20 years).

### 3. Definitions of deforestation and degradation under REDD+

Under UNFCCC policy (the Marrakech Accords) a forest is considered to become non–forest when it falls below a certain threshold for canopy cover (crown density), and remains so on a permanent basis (>20 years). The threshold lies between 10
and 30% and is selected by each country to meet its own requirements. Mexico has selected 30%; whereas Ghana for example has selected 15%. Following definitions used by FAO, an area that is temporarily destocked, but which is expected to revert to forest at the threshold level, is considered forest. Degradation has not been formally defined in UNFCCC documents up to now, although for the purposes of REDD+, it is understood to mean a lowering of biomass (and hence carbon) stocks while the forest retains a canopy cover above the selected threshold\(^1\). One can therefore argue that temporary de–stocking (e.g. in areas which are sustainably logged in a cyclical system with replanting) is a form of degradation. In SC systems, one part of the management unit is under cultivation while the rest is under successional forest re-growth. The canopy cover may be temporarily below the threshold in one patch, but will return within a few years (Nyerges, 1989; Rouw, 1993; Van Breugel et al., 2007; Van Doet al., 2010); on average the tree cover remains above the requisite level to be considered forest. Depending on the length of the cultivation and fallow cycles, and the growth rates of the trees during the fallow periods, SC may not lead to a continued decline in stocks, only to an average stocking level which is lower than that of the ´intact´ forest. What is clear is that in most cases SC does not lead to permanent (> 20 years) removal of tree cover, which is what is implied by ´deforestation´ in the context policy on REDD+, because during the fallow period, secondary forest re-grows.

For the purposes of REDD+ it is therefore more appropriate to consider SC as a cause of forest degradation rather than of deforestation (Houghton, 2012; Pelletier et al., 2012), since it results in forest with lower average above ground carbon

\(^{1}\) This definition is specific to UNFCCC, and has been disputed by ecologists and conservationists, who tend to see degradation in more holistic terms, relating not simply to quantitative loss of biomass but to reduced potential for the provision of goods and eco–services (which is the standard FAO definition). While not denying that degradation means much more than loss of carbon, REDD+ policy, which is based on the idea of marketing reduction in carbon emissions, is likely to continue to define degradation in terms of carbon stocks. However, it is likely to include other indicators – such as biodiversity – as ´safeguards´. In other words, a land owner claiming carbon credits for improved management of forest will have to demonstrate that these improvements have not led to loss of biodiversity, before the credits are valorized.
stocks (Houghton, 2005; Lawrence et al., 2010; Soto-Pinto et al., 2010) and soil organic carbon (SOC) (Lugo et al., 1986; Houghton, 2005).

In terms of policies or interventions to deal with SC as a cause of degradation, it is important to understand that the reason that SC has been a stable and popular cultivation system throughout agricultural history and in many parts of the world is mainly because although yields per hectare are low, it gives relatively high returns to labour (Raintree and Warner, 1986; Seidenberg et al., 2003). It is therefore an efficient form of production, provided population densities are low and potential cultivation area abundant, so that the fallow period matches or exceeds the time necessary for recovery of the sites. However given sufficient inputs, much higher yields per hectare could be achieved (West et al., 2010), such that for a given demand for maize, much less area would have to be under cultivation. A common policy assumption associated with REDD+ is that intensification of agriculture would raise yields in areas already under production, reducing the need for further expansion into the forests, thus reducing forest carbon emissions, and possibly enabling some cultivation areas to recover their forest status, thus increasing carbon sequestration.

Switching from traditional shifting cultivation systems to permanent agriculture is often suggested as part of such a strategy (West et al., 2010), and has been modeled following optimization approaches that would allocate resources in the most efficient way, using a binary view of landscapes (agriculture/forest) to spare land and reduce deforestation (Pirard and Belna, 2012; Byerlee et al., 2014). This kind of proposal is widely propagated in policy documents associated with REDD+ (for example, in the Regional Strategy for REDD+ in the Yucatan Peninsula (ECOSUR, 2012) and Mexico has been identified as one of the six priority countries with the potential for mitigating agriculture-driven deforestation emissions through land sparing (Carter et al., 2015). On the other hand, there is evidence of shortening fallow lengths in some Mexican tropical dry forest (Chávez, 1983; Lambert, 1996; Abizaid and Coomes, 2004; Cuanalo and Uicab-Covoh, 2005; 2006; Dalle and de Blois, 2006), and reversing this trend (increasing fallow lengths)
has also been proposed as a route to increasing carbon stocks and lowering emissions. 

Thus we see that both intensification of agriculture (sedentarization) and de-intensification (extending fallow periods) are being proposed in the context of improved environmental management in general and for the mitigation of carbon emissions in particular. In order to make a rational decision on these kinds of options, it would be important to have a better understanding of the conditions which have given rise to shortened cycles and of the carbon outcomes of all these the different options. In this paper we attempt to explore the options, albeit for the small case study area.

4. Carbon characteristics of Tropical Dry Forest

Tropical dry forest (TDF) is a vegetation type widely distributed in the Africa, Asia and Central and South America (Murphy and Lugo, 1986; Bullock et al., 1995). It occupies about 40% of all tropical forest area (Miles et al., 2006); in Latin America it covers around 700,000 km² of which 81% in South America and the rest in Mesoamerica and the Caribbean (Miles et al., 2006). This forest occurs in tropical regions under a wide range of rainfall conditions (250–2000 mm per annum; Murphy and Lugo, 1986), but where there is a pronounced seasonality in the distribution of this rainfall, and where the mean annual temperature is higher than 17°C (Bullock et al., 1995). This type of forest is widely distributed in Mexico (Miranda and Hernández–Xolocotzi, 1963; Rzedowski, 1978), along the Pacific Ocean coast, the Gulf of Mexico and in some regions of the eastern coast, currently covering 60% of the total area of tropical vegetation (Trejo and Dirzo, 2000), although earlier it was much more widespread. The trees of this type of forest tended to be multiple–stemmed, such trees accounting for 58.0% of total basal area (Dunphy, 2000; Gallardo-Cruz et al., 2005; 2009; Durán et al., 2006; Álvarez-Yépez et al., 2008; Méndez-Toribio et al., 2014). A characteristic of the carbon pools in many tropical ecosystems is that aboveground biomass (AGB) in
living trees represents only a small part of the total carbon, with soil carbon often carrying the larger part. Shrubs, dead material in the litter, woody debris house a very small proportion (Gibbs et al., 2007), and this has been confirmed for the case of Mexico (Hughes et al., 1999; Jaramillo et al., 2003; Marín–Spiotta et al., 2008). However, there is limited information available on current carbon levels in TDF in Mexico, and much less about rates of change of carbon levels.

The most recent estimations of primary dry tropical forest in Mexico show wide variations in total forest carbon stocks, the difference between the minimum and the maximum values of total carbon may be considerable (46.7 to 571 Mg/ha) (Delaney et al., 1997; Jaramillo et al., 2003; Vargas et al., 2008; Kauffman et al., 2009; Jaramillo et al., 2011). Comparing these carbon stocks with those of tropical moist forests, TDF has lower total carbon content (Fig. 1), even taking into account the high levels of carbon in the soil. Carbon stocks in many tracts of TDF in Mexico are however well below those in the intact forest levels, because so much of this forest is degraded. However, from the few studies that have been made on the distribution of carbon, it is evident that in secondary TDF (i.e. TDF that has recovered from clearing and has regenerated) soil carbon levels are almost as high as in primary forest, which may not be the case for humid forests (INECC, 2010).

Fig. 1.

5. Shifting cultivation in Mexico

Shifting cultivation is common in Mexico, often practiced by farmers as a secondary, extensive, production activity to complement the more intensive cultivation of rainfed or irrigated permanent plots (Hernández–Xolocotzi, 1988; Hernández–Xolocotzi et al., 1995; Moreno-Calles et al., 2014). The better quality, low–lying agricultural lands suited to permanent, and particularly to irrigated agriculture, are limited in availability, and therefore the more marginal areas on the hillsides are usually used in a cyclic system with a long fallow period. There are many variations in the management of shifting cultivation which are known by a
variety of names such as *milpa, coamil, ecuaro, tlacolol* and *tamagua*. However, the geographic extent and distribution of shifting cultivation in Mexico is not totally clear. Government statistics, in Mexico as elsewhere, record areas under agriculture but do not differentiate between permanent (irrigated or rainfed) and shifting cultivation (Mertz, 2009).

**The coamil system**

The *coamil* system in the state of Jalisco takes place on slopes or stony areas which are naturally covered by TDF. A piece of land is cleared, the majority of standing trees are removed from the plot, the debris is dried for one to five months and then burned. Then the farmer makes planting holes using a wooden stick with an iron blade (*coa*), into which the maize seeds are placed, often in combination with beans and squash. *Coamiles* are usually used to produce maize for two years and then left fallow for periods ranging from 5 to 10 years, (Chávez, 1983; Gerritsen, 2002, Borrego and Skutsch, 2014), although cattle are frequently allowed to graze on the area during the fallow, and occasionally the cleared areas are turned into permanent pastures and never return to cultivation. *Coamiles* are usually found on hillsides with slopes > 12% and not on the plains, which as noted above are used for permanent cultivation. The nature of the terrain forces farmers to use hand tools rather than ploughs pulled by horses or tractors. The use of chemical inputs (fertilizers, herbicides) is much less than on permanent plots, not least because of the initial fertilizing effect of the burning, but in general yields per hectare are lower than on permanent plots, although the returns to labour are relatively high.

A large part of Mexico’s TDF has been transformed by shifting cultivation (Gutiérrez–Alcalá, 1993; Challenger, 1998; Burgos and Maass, 2004; Maass, 1995). In addition, in the past three decades in particular, there has been clearance of TDF for pasture for extensive livestock rearing, after one or two years of shifting cultivation (González–Flores, 1992; Gutiérrez–Alcalá, 1993; Maass, 1995; Challenger, 1998; Trejo and Dirzo, 2000; Jaramillo *et al.*, 2003; Jaramillo *et al.*, 2011). This represents a complete change from shifting cultivation as a cyclical
system to permanent clearance for cattle rearing (i.e. in terms of REDD+ it implies deforestation, rather than degradation.)

The traditional SC coamil system is characterized by three main stages, slightly different from those described by Kleinman et al. (1995) and consists of: 1) the slash and burn process, 2) the cultivation stage and 3) introduction of cattle grazing, after which the plot may be cultivated again (e.g. as a result of policy incentives as described above), but in the traditional system is allowed to rest (Chávez, 1983).

The first stage, often called cleaning, requires considerable skill on the part of the cultivator. For a complete burning of woody elements, an intense fire is needed, and the optimal timing of the burning will be calculated by the cultivator based on the expected duration of the dry period (the longer the drying period, the more intense the fire; González-Flores, 1992; Jaramillo et al., 2003; Kauffman et al., 2003; Jaramillo et al., 2011). The burning date is established by the cultivator, after considering temperatures, clear sky and speed and direction of winds. Burning releases carbon into the atmosphere in the form of carbon dioxide, but some is also deposited in the soil as a result of incomplete combustion and ashes (Gonzalez-Flores, 1992). The organic carbon in the top layers of soil may be lost through volatilization (García–Oliva et al., 1999), although it appears to be replaced rather rapidly during the fallow period (Fig. 2, Vargas et al., 2008). This depends however on the characteristics of the site (Fig. 2).

Figure 2

The second stage, cultivation, starts at the beginning of the rainy season, around June, and the harvest will be in November or December. The third phase starts after this when cattle are allowed to graze of the stubble and any crop residues are removed, but generally the land will be used for a planting a second time the following year. After two years, the natural vegetation is allowed to regrow, and cattle are often allowed to wander freely over these areas, giving a 2:8 rotation cycle. Initial re–growth is rapid; generally the areas are characterized by shrubby
vegetation for about four years after cultivation ceases, but after that the area can be more properly considered a regenerating woodland, as the trees will have already attained a height of over 5m. If not re–cleared, biomass and carbon increase with forest age after the fire event, as would be expected (Vargas et al., 2008). Secondary TDF may regain its original AGB stocking levels in 30–40 years, although it is already fairly dense after 10–15 years.

6. Description of the case study area

The villages of Tonaya and Temazcal both lie in the centre of the basin of the River Ayuquila in Jalisco state (Figure 3), at an altitude of approximately 990m above sea level, in an area where the natural vegetation is tropical dry forest, with some oak and coniferous forests on the higher slopes. The area is semi–arid with rainfall of approximately 650 mm per annum, falling almost entirely between June and September (Jardel et al., 2012).

The typical soils in both settlements are Regosol and Litosol, accounting for ~80% of the surface area in Tonaya and 93% for El Temazcal: both these soil type have little organic matter. Regosols predominate, these are young soils with depth >20 cm, but with a layer that becomes hard and crusty when vegetation is removed, preventing the penetration of water into the ground. Litosols are shallow (depth <20 cm), and not well suited for agriculture.

Many of the farmers in these settlements have a plot for permanent cultivation in the valley but also a forest plot on the slopes, which is used for shifting cultivation. In the terrain that is used for shifting cultivation there is a mosaic of forest states; areas that have recently been slashed and burned and are now being cropped, combined with areas at various stages of the fallow cycle. Permanent agriculture (rainfed and irrigated) is found at the lower elevations. These fields are cultivated either once or twice a year, year in year out, with considerable use of chemical fertilizers and other inputs (Gerritsen, 2002; Borrego and Skutsch, 2014)

Fig. 3.
Tonaya

Tonaya village is located within the larger municipality of Tonaya, and has a population of 3,497. In this village the farmers make an active selection of tree species to retain in the *coamil*, leaving in particular a small number of individuals of hardwood species with straight trunks and large branches, which are well suited for fence posts and house supports and construction materials. The rootstocks of these species are left intact which allows rapid regeneration by coppicing.

El Temazcal

El Temazcal forms part of the municipality of Tuxcacuexco. This is an agricultural municipality, the main economic activity is horticulture (vegetables), which are grown for sale by 80% of the farmers. It has a total of population of 184 distributed over three distinct settlement areas. In El Temazcal, in addition to cutting trees for firewood and fence posts, there is an active selection of tree species in the coamil patches by farmers by removing the species with small crowns and allowing a limited number of bigger crown species to gain space in the canopy, increasing shade in the fallow period for the benefit of the cattle.

7. Methodology

Sites selection

The landscape in the study area consists of a mosaic of permanent pasture (PL) and permanently cultivated fields (PA), SC fallow plots (FP) which range in age up to 10 years, old fallow (OF) of more than 10 years, which may never come back into the cycle, fields currently used for the cropping stage of the shifting cultivation cycle (SCc), secondary forests and fragments of old-growth forest (OG), which is forest which has not been used for cultivation but is generally conserved by the
local population because it is in steep and rocky areas and protects water sources (Jardel et al., 2012). In order to gather information on carbon stocks during different phases of the shifting cultivation cycle, biomass and soil carbon (including both mineral carbon and SOC) data were obtained in a survey carried out in August 2013 and March 2014. Plots were set up in FP and OG, where AGB was measured and soil samples taken for laboratory analysis of carbon content. Soil carbon data were also obtained from SCc, and for comparison, also from PA and PL.

A total of 23 FP sites were identified where maize had been grown in the previous 10 years; 10 in Tonaya and 13 in El Temazcal, and 18 sites of PA, SCc, OG and PL were sampled. In addition, we have data on above ground biomass 15 plots in these villages in old fallow (OF) areas, that is areas which have been under fallow for 10-30 years. For these we do not have soil carbon data, but we conservatively assume this is not higher than that of fallows of 10 years. The 23 FP sites were of different ages, which we divided into three age categories (years 3-4, 5-7 and 8-10, following two years of cultivation). Information on plot age was obtained from the owners. OF forests plots supply information on the level of carbon stocks that could be achieved within the next 20 years if SC were to be halted; these were used as a reference level.

**Field sampling**

A circular sample plot of 11.28 m of radius was laid out within each site to estimate the AGB carbon stocks. Diameter at breast height (DBH: 1.3 m height) was measured for all the individuals with DBH ≥ 2.5 cm, using diameter tapes or calipers, and the species, genera or morpho–species was identified. Trees of less than 2.5 cm DBH were counted as woody stems. The dry AGB of trees was inferred from the allometric models of Martínez–Yrízar et al., (1992). AGB of multi–stemmed trees was calculated separately for each stem. To obtain estimates of the above ground carbon (AGC), we used a conversion factor of 0.5.
Sites with no shrubs or trees (permanent pasture sites, cultivation phase of shifting cultivation plots and permanent agriculture sites) were considered to have zero AGC. This gives a conservative estimate of above ground carbon stocks for the cultivation stage of the shifting cultivation system since, as mentioned above, it is common for farmers to leave one or two trees standing.

Data on soil carbon were obtained by collecting four 10–cm deep soil samples (ca. 16–44 g each, with a bulk density ranging from 0.59–1.58 gr/cm³) at the northern, southern, eastern and western limits of each sampling plot. The four samples from each site were pooled to form a single sample and were sent for laboratory analysis. This was done following the OHHW protocol with the Perkin Elmer 2400 Series II Elemental Analyzer in CHN mode.

Total carbon was obtained for each plot as the summation of AGC and soil carbon.

**Statistical analysis of field data**

We used analysis of variance to examine differences in carbon storage in the coamial and fallow stages of shifting cultivation, in old fallow, in permanent agricultural plots, permanent pasture, and old–growth forest. AGC and soil carbon were considered dependent variables, with land–use class as the independent variable. The active fallow phase (years 3 to 10) was analyzed in two ways, first as three fallow phases of different ages (FP1, FP2 and FP3) and then as single category (FP). AGC and soil carbon values were log and square root transformed when necessary to meet ANOVA assumptions regarding the homogeneity of error variances and distribution of residuals. The normality of the data distribution for AGC and soil carbon were tested separately for each class using the Shapiro–Wilk test while homogeneity of variances was evaluated using Bartlett Test. The data set for AGC with the fallow phase considered as a single factor, and the other dependent variables, all fulfilled ANOVA assumptions (one–way ANOVA). Soil carbon and AGC analyzed as three separate fallow phases, and AGC with fallow phases as a group did not meet the ANOVA assumptions and so the Kruskal–Wallis One Way on Ranks was run instead. Differences in AGC and soil carbon
among treatments were analyzed using post hoc Dunn's Method. All the statistical tests were conducted using the R software (R Core Team, 2012) and performed at a 0.05 significance level.

The differences in carbon stocks for AGC were tested at two levels of plot aggregation: (1) all plots in the fallow stage versus old growth sites, (2) the forest fallow plots grouped by age class categories. Soil carbon was analyzed in a similar way. To show the relation between soil carbon density and AGC density across the different wooded land uses, the mean ratios for both were calculated for FP1, FP2, FP3 and OF.

**Methodology for estimating carbon balance of production of one ton of maize**

We compared: (a) the carbon stocks and emissions that would pertain in a shifting cultivation system with a 10 year cycle (2 years cropping and 8 fallow) with those in a permanent cultivation system over the same time period; (b) the average carbon emissions and stocks in different SC production regimes (6 year, 10 year and 24 year cycles, with 2 years cropping in each), under the assumption that the shortened cycles are not due to pressure on the land and (c) a six year SC system under land pressure, *i.e.* in which the shortened cycles are the result of an increase in demand for land. To create a fair basis for comparison made all calculations for production of one ton of maize and we held the land area constant within each one of these scenario comparisons.

Average maize yield for SC and PA plots in both communities were obtained from a survey administered to 39 farmers in the area of study. These values gave us a yield of 1.684 tons/ha for SC and 3.768 tons/ha for permanent cultivation. The carbon stock in parcels of SC and PA was estimated taking into account the area needed to produce one ton of maize in each system. In the SC system this is 0.59 ha for cultivation and 2.38 ha fallow, totaling 2.97 ha. For PA the cultivation area is 0.27 ha and to allow a fair comparison we assume that the farmer keeps the
remaining part of the 2.97 ha (2.70 ha) in its existing state, which is taken to be OF. In PA, the cropping field is cultivated year after year without a fallow stage.

(a) Shifting cultivation versus permanent agriculture

Using the carbon levels measured in the field we obtained the average annual carbon sequestered in both agricultural systems associated with the production of one ton of maize per year, taking into account an equal area of land and we compare these levels with carbon stock levels in OF forest (i.e. to test the effect of converting such forest to each production system). We then calculate the annualized emissions from PA and SC. The AGC for the coamil stage of SC and for PA was considered to be zero, although in practice, in SC there is a little woody vegetation is left in the cultivation area. The only carbon pool considered for permanent agriculture was the soil carbon, and the total carbon concentration for FP and coamil stage of SC (see Table 1).

(b) Changing lengths of fallow

We then compared scenarios to test the effect of changing length of SC fallows on average carbon stocks, on the assumption that the farmer is essentially free to select his cycle length (i.e. his choice is not constrained or forced by pressure on the land). To grow one ton of maize requires 0.59 ha in cultivation. In a 24 year rotation, this would require a total area of 7.12 ha. We therefore made calculations over this area also for the 10 and 6 year cycles, assuming that the unused areas would retain their OF vegetation.

(c) Land under pressure

Finally we created a scenario in which the shortened cycle length (6 years) is due to demand for more land to be brought into production (i.e. population growth is forcing more farmers onto the same area of land). In this case, there are 4 sets of a 6 year SC system distributed over the same 7.12 ha, and no land is left under OF.
8. Results

a. Soil carbon stocks

Overall, soil carbon makes a much larger contribution to total carbon ($50.44 \pm 3.10$ Mg/ha) than AGC ($11.31 \pm 1.53$ Mg/ha) in all the land cover types examined (Table 1). The highest average soil carbon levels were found in the coamil (cultivation) stage and in FP3 (fallow in years 8-10 of the cycle). The lowest soil carbon levels were in the sites under permanent agriculture. Within the shifting cultivation system, soil carbon accounts for around 85-87% of total carbon in fallow stages. The high soil carbon levels in the coamil phase appear to fall by about a third in the first fallow period and then increase gradually, but there is a very high variation in levels between sites (Fig 5a). In old-growth (OG) sites, soil carbon forms 64% of the total carbon stock.

The differences in log values of soil carbon in the different land uses were found to be statistically significant ($F = 2.82$, df = 6, $p < 0.05$). Also we found that FP2, FP3, Coamil and PL all had significantly greater soil carbon density than PA (Tukey HSD test, $p$-value < 0.05). On average, there was about 10% more carbon in soils under the shifting cultivation regime than in old growth forests. This is broadly in line with the estimates made by Houghton (2005).

b. Above ground carbon stocks

AGC is highest in OG forest, and as would be expected is much lower in land which is under fallow, again with a high degree of variation between sites (Fig 5b).

FP3 was distinctive in having small trees with DBH $\geq 2.5$ cm, but at the same time it harbors the highest number of stems and has the highest average DBH of the three fallow phases. OF shows the highest number of stems per hectare, but fewer trees than FP3, due to the coppicing characteristics of the species involved.

There were statistically significant differences in overall AGC ($X^2 = 17$, df = 4, $p$-value < 0.05). OG and OF had a significantly higher AGC carbon density than fallow of 3–4 and fallows of 8–10 years (Wilcoxon pairwise test, $p$-value < 0.05). In
general, the stocks of above ground carbon in areas within the active shifting
cultivation system (10 year cycle) are about 30% of those in old growth and about
40% of those in the extended SC system (OF), which is in line with the findings of

It is interesting to note that a high proportion of trees (~70%) were accounted for by
only 7 species. The two most common species in FP were of the multi–stem type,
which are used mostly for fence posts; the two most common species in OG are
less suited for fence posts, indicating a clear impact of human intervention on the
species range in shifting cultivation areas.

c. Total carbon stocks

In aggregate, PA shows the lowest total carbon density (Table 1), and includes the
site with lowest total carbon density in the whole sample (12.38 Mg/ha). The site
with the highest total carbon concentration (128.04 Mh/ha) was in the FP3 set.

Considering all the means of the log of total carbon stocks in the different land use
types, we found statistically significant differences (F = 7.3, gl = 6, p < 0.05)
between them; in particular, the stocks in the second and third age classes of
fallow phases, coamil and PL were statistically higher than those observed in PA
(Tukey HSD test, \textit{p-value} < 0.05) (Fig 5c). There were statistical differences (F=
9.26, gl = 4, \(P < 0.05\)) in the log of total carbon density as well when the three
fallow age classes are grouped into one class (Fig 5d), although permanent
agriculture (PA) is significantly different from the rest of land use classes (FP1, FP2
and FP2 (taken individually), FP as a whole, Coamil, OG, OF and PL) (Tukey HSD
test, \textit{p-value} < 0.05) (Fig 5d).

Most importantly what the statistics show is that total carbon stocks in areas under
active shifting cultivation regimes are only about 14% lower than those of old
growth forest which are in areas unsuited to agriculture and 16% lower than those
in the old fallows. The implications of this in terms of emissions are developed in
the following section.
The total area required to produce one ton of maize in a 10 year SC system is 2.97 ha. Over the long run (steady state), the total carbon stocks over this area would be 175 Mg, taking into account the varying amounts of carbon in the different fallow stages (Table 2). The carbon stocks associated with production of one ton of maize through PA in this same area would be 197 Mg, taking into account that most of the area would remain under OF forest (the ‘land sparing’ scenario). Our results thus indicate that in the process of producing a ton of maize annually, the total carbon stocks in the shifting cultivation system are lower than those in permanent agriculture (Table 2).

The stock in an equivalent area of OF would be 209 Mg, thus the loss in stock if this land were converted to SC and PA would be 34 Mg C and 13 Mg C respectively. To obtain an emission rate (annualized losses) an arbitrary (and conservative) time horizon of 20 years (two complete SC cycles) has been selected. This results in emissions of 1.72 Mg C/Mg maize/year in the case of SC and 0.65 Mg C/Mg maize/year in the case of PA (Table 2). Expressed in terms of area, the emissions would be 0.57 Mg C/ha/year for the case of SC and 0.21 Mg C/ha/year for PA, taking the whole are into account.
(b) The effect of changing fallow lengths under conditions of no land pressure

In this scenario the unit of land is 7.2 ha (which is what is needed for the operation of a 24 year SC to produce one ton of maize). The scenario with the longest cycle (24 years) has the lowest carbon stocks (449 Mg), because apart from the small patch which is under cultivation in any one year, the rest of the area is all under fallow. The shortest cycles show the highest carbon stocks (476 Mg C), since only a small part of the remaining area is under fallow, while most of the land remains as OF forest (Table 3). If the area brought under shifting cultivation were OF forest, then the losses would be 53.11, 33.72 and 26.39 Mg C respectively for the 24 year, the 10 year and the 6 year cycle and the associated emissions would be 2.65, 1.68 and 1.31 Mg C/Mg maize/year respectively, annualized over an arbitrary period of 20 years. The areal emission rates would be 0.37, 0.24 and 0.18 Mg C/ha/year respectively.

Table 3

(c) Effect of increasing the density of shifting cultivation

The final scenario considers what would happen if SC were to be shortened as a result the need to produce more food from the same area, i.e. the case of increasing pressure on the same area of land. We assume that instead of one ton of maize, four are required from the same area, such that four shifting cultivators, each operating a 6 year SC cycle, are working at the same time in the 7.2 ha area described in the example above. As Table 4 shows, this would result in long run carbon stocks of 394.46, for the production of four tons of maize. Annual emissions, calculated as before, would be 1.35 Mg C per ton of maize, which is only marginally higher than that produced by a 6 year cycle in the low intensity scenario. Emissions would be the equivalent of 0.19 Mg C/ha/year

Table 4

9. Discussion
Shifting cultivation is an agricultural system which has often been blamed for negative environmental effects, most recently in the context of emissions of carbon dioxide (Houghton et al., 2003; Houghton and Goodale, 2004; Nigh and Diemont, 2013), where it has been linked to deforestation (permanent change of land use from forest to non–forest). This however does not take into account the forest succession characteristics which shifting cultivation involve. Shifting cultivation does not, in general, result in deforestation (forest cover loss), but in temporary removal of trees, modifying the vegetative cover to a mosaic landscape of secondary growth forest (Fox et al., 2000, Jardel et al., unpublished data). On average, of course, the carbon stocks will be lower in these forests than in primary forests (Detwiler and Hall, 1988; Lawrence et al., 2010), but this should be considered forest degradation, not deforestation. Both the Marrakech Accords and FAO consider that forest land temporarily unstocked as result of human intervention, but which will eventually surpass the threshold parameters, is forest (FAO, 2002); hence we should consider land under shifting cultivation to be forest land. There are two paths to obtain land for shifting cultivation: 1) clearing in mature forest ‘primary forest to shifting cultivation’ and 2) clearing in secondary forest ‘secondary forest to shifting cultivation’ (Fukushima et al., 2008). In the tropical dry forests of the Ayuquila basin, the first route is seldom found because this area has been managed for agriculture and cattle ranching at least since Mexico’s colonial era, meaning that there are very few primary forests in the region. The few isolated areas of relatively intact forest in this area are old growth forests which are forests on rocky and steep areas which are not suited to agriculture and which are kept by local people to conserve the water supply areas. In the study area we can find a mosaic of forest at different levels of degradation, though with tree crown cover of over 30% on average (Jardel et al., unpublished data).

Although longer fallow periods increase soil fertility (Lu et al., 2002), shifting cultivators in the study area informed us that they prefer not to leave fallows for more than 8 years after cultivation, because if left longer, trees grow to large diameters, and require much more labour when they are eventually cut. Older fallows are therefore left by the local people and their preservation is enhanced by
the fact that the authorities encourage farmers not to clear these areas, for environmental reasons. In addition to this, a government subsidy which is tied to a cycle of maximum of five years is pushing the average cycle length steadily downwards, while subsidies for inputs such as fertilisers mean the natural re-fertilisation of longer fallows is not longer so highly valued.

Time affects various forest attributes, but the time needed for carbon level to recover to the levels found in old–growth forest is in reality highly variable (Lawrence and Foster, 2002; Vargas et al., 2008). The original conditions of forest when it was felled, remnant patches of vegetation (Velázquez and Gómez–Sal, 2007; 2008; Maza–Villalobos et al., 2011), intensity and frequency of management practices (number of slash–and–burn cycles, and length of fallow period (Dalle and de Blois, 2006; Eaton and Lawrence, 2009), type of soil, changes in environmental conditions and resource availability (Lebrija–Trejos et al., 2011) all play an important role in how fast the succession on fallowed shifting cultivation plots rolls out over time. We found that the biomass and carbon stocks in the OG forests were not any higher than it was in fields which had been in fallow for 8 years, but this may relate to the fact that OG forests are on steep and rocky terrain. We also found that the carbon in the OG soils was much lower than in the any of the fallow fields, which relates to the fact that it has not been burned or cultivated for a very long time.

In our study areas we found that farmers make an active selection of trees when clearing for coamil: some species are retained and others are removed when the plot is cleared for cultivation. The selection of species to be retained reflects the use values of particular species, for example hardwood species with slow growth are a source of poles. These trees species are cut when they reach a DBH of around 15 cm.

The results of this study are generally in line with findings obtained in other studies. In terms of the impact of shifting cultivation on carbon stocks, both soil carbon and above ground carbon exhibit rapid growth in the years following cultivation (Delaney et al., 1997; Chazdon, 2003; Jaramillo et al., 2003; Read and Lawrence,
and where the fallow phase is long, the carbon stocks reach levels similar to those of old growth forests, though as noted above, more of this carbon is in the soil and less in the above ground pools. The finding that soil carbon forms such a large proportion of the total carbon in the TDF ecosystem is not new. Soil carbon is known to be the largest carbon pool in the terrestrial biosphere (Jobbágy and Jackson, 2000); the amount of carbon contained in tropical soils has been estimated to be more than twice that contained in above-ground biomass (Post et al., 1982), and potentially soil may store carbon at great depths (Jobbágy and Jackson, 2000). Values approaching those we found in our case studies are reported for TDF also by INECC, the Instituto Nacional de Investigaciones en Cambio Climatico, for Mexico (INECC, 2010). We note however that most of this carbon is in the upper layers of the soil; TDF soils are in general shallow (Trejo and Dirzo, 2000), and their exposure, for example through the creation of fields for permanent cultivation, can result in rapid loss of much of this carbon, as our data shows.

What our study also reveals however is that the presence of so much soil carbon mitigates or helps to balance the losses of carbon in above-ground vegetation when an area is cleared, particularly when the clearance is only temporary as in shifting cultivation. The fire event itself helps to increase soil carbon levels because of the charcoal and ash deposits, reflected in the very high soil carbon levels we observed in the coamil phase. These values then drop in the first fallow period but increase rapidly over the next 6 years; faster, indeed, than the recovery of the above-ground carbon stocks. In the case of permanent agriculture on the other hand it is clear that soil carbon levels are rapidly depleted and do not recover.

This has important implications as regards food security and climate policy in areas of the kind we were studying, which are dominated by small scale and subsistence production. There are a number of programmes operating in Mexico (PROCAMPO, Alianza para el Campo, ASERCA) which are promoting a movement from shifting
cultivation to permanent cultivation, essentially based on the idea that intensification of agriculture should result in sparing of forest land, commonly known as the Borlaug hypothesis. This hypothesis is rooted in the Green Revolution of the 1970s (Angelsen and Kaimowitz, 2001; Rudel et al., 2009; Lobell et al., 2013), although the term has also been used more recently in the specific context of Borlaug’s ideas on genetically modified crops (Borlaug, 2000; 2007; Toft, 2012). The idea of ‘land sparing’ through agricultural intensification is controversial, with numerous studies showing that at the local level, intensification may in fact result in increased deforestation as improving agricultural conditions raise the demand for land, illustrating the so-called Jevons’ paradox (Angelsen and Kaimowitz, 2001; Rudel et al., 2009; Perfecto and Vandermeer, 2010; Gockowski and Sonwa, 2011; Pirard and Belna, 2012; Byerlee et al., 2014). However the theory still has supporters (e.g. Grau and Aide, 2008) and there is evidence that at the global level technology–driven (rather than market driven) intensification may be correlated with lower deforestation rates (Andersen et al., 2002; Byerlee et al., 2014). In this context it may be noted that agricultural intensification is the most common intervention proposed in the readiness documents of 43 REDD+ countries to tackle agriculture-driven deforestation and forest degradation (Salvini et al., 2014), and is regularly advocated as a means of reducing deforestation in general (Carter et al., 2015).

Our study adds to the discussion around the Borlaug hypothesis. We show that at least for the case of one important environmental service – the conservation of carbon stocks – agricultural systems based on intensive crop production (permanent agriculture) do result in lower emissions than shifting cultivation, for a given level of production of food. However this does not take into account the much higher energy inputs that intensive crop production requires in the form of fertilisers and pesticides and the long time scales they require for recovery of soil physical status. An analysis of this would be needed to make a fair comparison, but was beyond the scope of this study.

However, we also throw light on an alternative policy option: extending the length
of shifting cultivation cycles, which is another strategy often proposed in the context of improved environmental management (Dalle and de Blois, 2006; Eaton and Lawrence, 2009). Shortened shifting cultivation cycles are frequently blamed for increased levels of deforestation and degradation and for increased carbon emissions (for example, in the analysis of drivers in many of the REDD+ Readiness documents, Salvini et al. (2014)). We show that extending these cycles does not necessarily result in reduced emissions. In general, shortened cycles are only the cause of increased emissions when the reason for the shortened cycle is increased pressure on the land, i.e. when a larger proportion of the land is brought into production, and in our case, the increase in emissions that result from this more intensive use of the landscape is only marginal. In the study area, cycles have been shortening in the last decade for different reasons, and this has resulted not in increased emissions but in a decrease.

In general we urge caution in the promotion of policies to curtailing shifting cultivation and to move towards more intensified ‘spatially segregated’ landscapes (van Noordwijk et al., 2012) as a means of reducing emissions. Under land sparing schemes the goals are strictly separated; agricultural production is maximized in some areas and other areas are set aside for carbon sequestration (Perfecto and Vandermeer, 2010). Bearing in mind the Jevons’ paradox this could result in the reverse of the intended effect. The alternative, land sharing (Perfecto and Vandermeer, 2010), which aims at both objectives on the same area of land, may in the long run be a better solution. Importantly in this context we show that shortened cycles may in fact be an advantage, contrary to the many scholars who have argued the reverse (e.g. Dalle and de Blois, 2006; Eaton and Lawrence, 2009). The wide range of values of carbon stocks we found in the coamil and fallow phases indicates that there may be room to optimize total carbon stocks, and reduce emission rates, through improvements in the way the system is managed, although this will of course depend on whether there is pressure on the land. Decreases of the amount of land under cultivation in SC could also be achieved by improving local management strategies (for a case in the Yucatan, Mexico, see Pascual, (2005)).
The conclusions of our study are however specific to a particular area of TDF in a particular part of Mexico and may not be generalizable to other areas, because the biophysical and socioeconomic conditions under which shifting cultivation is performed vary widely. For example, the ratio of soil carbon to above–ground carbon is completely different in rain forests and even within Mexico there are considerable variations in the dynamic processes of TDF. Read and Lawrence working in Yucatan found that although above ground carbon stocks fall over several cycles, soil carbon levels remained high (Read and Lawrence, 2003), although soil conditions in the Yucatan are very different (much thinner, more alkaline) from those in Jalisco. Moreover, shifting cultivation is carried out in very many different ways throughout Mexico (Hernández–Xolocotzi, 1988), and different kinds of management/fallow cycles could modify the uptake of carbon by the soil. We observed differences even between the two villages within our study, which are within the same ecosystem but which practice slightly different management technique in their shifting cultivation systems.

Nevertheless we feel that the results of our work bring important points to light, although of course more systematic research would be needed to reach some more general policy conclusions.

10. Conclusion

Our study shows that shifting cultivation results in a lowering of above ground carbon stocks in the forests affected, which in terms of REDD+ should be considered to constitute degradation. In very few cases does it result in deforestation, since the forest remains forest in the long run. Our indications are that it creates more emissions per ton of maize than would be produced in permanent agricultural systems, although our calculations do not include the additional emissions that would be associated with the much higher energy inputs used in permanent agriculture, which would reduce the observed difference. However we show that per ton of maize produced, shorter cycles result in higher remaining stocks of carbon and lower emissions, provided the shorter cycles are not associated with increased pressure on land. This is the situation in the study
area, where recent reductions in the cycle appear to be a response to a variety of policy initiatives and financial incentives rather than to land pressure.

These findings are very significant for REDD+ policy, as shifting cultivation is often held to be the cause of deforestation and either its replacement by permanent agriculture (´agricultural intensification´ or ´sedentarization of agriculture´), or lengthening of fallows are often put forward the means to reduce carbon emissions. If our findings are correct, they indicate that policies on shifting cultivation in the context of climate change may need to be nuanced. At very least, the reasons for shortened fallows (the context in which they occur) would need to be thorough investigated before a policy to extend them is promoted. Moreover, we think it doubtful that the observed difference in emissions between shifting cultivation and permanent agriculture justifies a policy of land sparing. Clearly, however, more research will be necessary to understand the impacts of shifting cultivation on carbon stocks in other areas and in other ecosystems in order to reach more global conclusions.

**Acknowledgments**

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## Table 1

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<tr>
<th>Classes</th>
<th>No. of sites</th>
<th>Soil C (Mg/ha)</th>
<th>AGC (Mg/ha)</th>
<th>Total C (Mg/ha)</th>
<th>DBH (cm)</th>
<th>No. of trees with DBH≥2.5cm/ha</th>
<th>No of stems/ha (includes multi–stemmed trees)</th>
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<tbody>
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<td>FP</td>
<td>23</td>
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<td>59.26±4.83</td>
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<td>OF</td>
<td>15</td>
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</tbody>
</table>

FP= mean over entire fallow phase of shifting cultivation; FP1= fallow phase 3–4 years, FP2= fallow phase 5–7 years; FP3= fallow phase 8–10 years; OG= old–growth forest; PES= forest under payment for environmental services; Coamil= cultivation phase of shifting cultivation; PL= permanent pasture, PA= permanent agriculture.

² Without field data on soil carbon in the OF plots, we can assume, conservatively, that the soil carbon in OF will be the same as in FP3
<table>
<thead>
<tr>
<th></th>
<th>PA</th>
<th>SC</th>
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<tr>
<td>Maize yield (tons/ha)</td>
<td>3.76</td>
<td>1.68</td>
</tr>
<tr>
<td>Cropping area needed to produce 1 Mg maize (ha)</td>
<td>0.27</td>
<td>0.59</td>
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<tr>
<td>Area (ha) needed to produce 1 Mg maize/annum in long term</td>
<td>0.27</td>
<td>2.97&lt;sup&gt;3&lt;/sup&gt;</td>
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<tr>
<td>Area that remains OF as uncultivated land</td>
<td>2.70</td>
<td>0.0</td>
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<td>Carbon stock in parcel undergoing cropping (Mg)&lt;sup&gt;4&lt;/sup&gt;</td>
<td>6.46</td>
<td>37.24</td>
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<td>Carbon stock in total area (Mg)</td>
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<td>175.33</td>
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<tr>
<td>Carbon stock in equivalent area of OF</td>
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<td>Loss if converted from OF</td>
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<tr>
<td>Loss annualized over 20 years (annual emissions/Mg maize)</td>
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<td>Annual emissions Mg C/ha</td>
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</tbody>
</table>

<sup>3</sup> Assumes a SC cultivation system of 10 years, 2 cropping and 8 fallow

<sup>4</sup> Includes both above ground and soil carbon
## Changing lengths of fallow

<table>
<thead>
<tr>
<th></th>
<th>SC 6 years cycle</th>
<th>SC 10 years cycle</th>
<th>SC 24 years cycle</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maize yield</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Cropping area per year needed to produce 1 Mg maize (ha)</td>
<td>0.59</td>
<td>0.59</td>
<td>0.59</td>
</tr>
<tr>
<td>Rotation&lt;sup&gt;5&lt;/sup&gt;</td>
<td>3</td>
<td>5</td>
<td>12</td>
</tr>
<tr>
<td>Total area needed to produce 1 ton of maize/annum long term (ha)</td>
<td>1.78</td>
<td>2.97</td>
<td>7.12</td>
</tr>
<tr>
<td>Area which remains as unfarmed OF (ha)</td>
<td>5.35</td>
<td>4.16</td>
<td>0.0</td>
</tr>
<tr>
<td>Total carbon stock in area farmed, including fallow (Mg)</td>
<td>98.61</td>
<td>175.33</td>
<td>449.77</td>
</tr>
<tr>
<td>Total carbon stocks in total area (Mg)</td>
<td>476.49</td>
<td>469.16</td>
<td>449.77</td>
</tr>
<tr>
<td>Carbon stock in equivalent area of OF</td>
<td>502.88</td>
<td>502.88</td>
<td>502.88</td>
</tr>
<tr>
<td>Loss if converted from OF</td>
<td>26.39</td>
<td>33.72</td>
<td>53.11</td>
</tr>
<tr>
<td>Loss annualized over 20 years (annual emissions/Mg maize)</td>
<td>1.31</td>
<td>1.68</td>
<td>2.65</td>
</tr>
<tr>
<td>Annual emissions Mg C/ha</td>
<td>0.18</td>
<td>0.24</td>
<td>0.37</td>
</tr>
</tbody>
</table>

<sup>5</sup>Plot is cultivated for 2 years, so in a 24 year cycle there are 12 cohorts.
Table 4

<table>
<thead>
<tr>
<th>Description</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maize yield</td>
<td>4</td>
</tr>
<tr>
<td>Total area needed to produce the maize yield/annum long term (ha)</td>
<td>7.12</td>
</tr>
<tr>
<td>Area which remains OF (ha)</td>
<td>0</td>
</tr>
<tr>
<td>Total carbon stock in area farmed, including fallow (Mg)</td>
<td>394.46</td>
</tr>
<tr>
<td>Total carbon stocks in total area (Mg)</td>
<td>394.46</td>
</tr>
<tr>
<td>Carbon stock in equivalent area of OF</td>
<td>502.88</td>
</tr>
<tr>
<td>Loss if converted from OF</td>
<td>108.42</td>
</tr>
<tr>
<td>Loss annualized over 20 years (annual emissions, Mg C)</td>
<td>5.42</td>
</tr>
<tr>
<td>Annual emissions (Mg C/Mg maize/year)</td>
<td>1.35</td>
</tr>
<tr>
<td>Annual emission per ha (Mg C/ha/year)</td>
<td>0.19</td>
</tr>
</tbody>
</table>
Tables captions

**Table 1.** — Carbon stocks and other characteristics of sampled sites in TDF (mean ± S.E.).

**Table 2.** — Carbon stock balance for the production of 1 ton of maize, comparing permanent agriculture and shifting cultivation.

**Table 3.** — Average carbon stocks over 6, 10 and 24 years cycles.

**Table 4.** — Average carbon stocks and emissions over a 6 years cycle under a situation of increased pressure for land (maize production rate increased by a factor of 4).
Figure 5

(a) Log of Soil Carbon Mg

(b) AGC Mg Cline

(c) Log Total Carbon Mg

(d) Log Total Carbon Mg
Figures captions

Figure 1.—Total carbon stocks in tropical forest of different types, as reported in the studies indicated (AGC= above ground carbon, BGC= below ground carbon).
Image from Kauffman et al., (2009): TDF= Tropical dry forest, TFF = tropical floodplain forest in TDF, TMF= transitional moist/dry, Cerr= Cerrado, TMO= tropical moist forest, TWF=tropical wet forest and TMW= tropical montane/wet forest. Data:
1, 2 Jaramillo et al. (2003); 3,4,7,8 and 10: Delaney et al. (1997); 5: Castro (1995);

Fig. 2 Below–ground and above–ground carbon stocks for Mexican TDF after burning (from: Vargas et al., 2008).

Fig. 3 Location of study area.

Fig. 4 Fence within community forest, delimiting individualized forest areas.

Fig. 5 Carbon stocks for: a) AGC splitting the fallow by age class categories, b) Soil carbon splitting the fallow by age class categories, c) Total carbon splitting the fallow by age class categories and d) Total carbon considering the three separate fallow phases. FP1= fallow phase of shifting cultivation of 3–4 years, FP2= fallow phase of shifting cultivation of 5–7 years, FP3=fallow phase of shifting cultivation of 8–10 years, FP= fallow phase of shifting, OF= old–fallow forest, Coamil= cultivation phase of shifting cultivation; OG= old growth forest; PA= permanent agriculture, PL= permanent pasture.