LETTER

Changes in carbon storage under alternative land uses in biodiverse Andean grasslands: implications for payment for ecosystem services

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Abstract

Ecuadorian páramo grasslands have become the focus of Payment for Ecosystem Services programs that have promoted land-use changes such as afforestation and reduction of burning, grazing of cattle or sheep, and agricultural expansion. However, limited information exists on the relationships between land use in páramos and the production of ecosystem services, including the direction and magnitude of changes in carbon storage. In an evaluation of eight sites representing incentivized land uses, we found significant differences in both soil carbon and aboveground carbon. The results support previous findings on the effects of pine plantations and suggest that limiting burning may be effective at enhancing carbon storage. They also indicate that, in highly productive grasslands such as the páramo, both belowground and aboveground C storage can be high, even when compared to some types of afforestation, providing support for greater attention to the role of grassland conservation in climate mitigation strategies.

Introduction

The alteration of land use or management has potential to contribute to climate change mitigation, with carbon credits generated through these activities playing an important role in voluntary carbon markets as well as existing and future compliance markets. These land use and management approaches include afforestation, reforestation, and avoided deforestation as well as improved management of forests, grasslands, and rangelands (Gibbon *et al*. 2010). In many cases, projects that aim to improve terrestrial carbon storage through land management are seen as having the potential to provide biodiversity and livelihood cobenefits (Bekessy & Wintle 2008; Venter *et al*. 2009; Charnley *et al*. 2010; Strassburg *et al*. 2010). However, carbon-oriented land management faces a number of challenges (Agrawal *et al*. 2011). Among them is the fact that the relationships between the land uses incentivized by Payment for Ecosystem Services (PES) programs and the production of ecosystem services are often not well understood, and some have expressed concern about possible trade-offs between carbon-oriented land management and biodiversity conservation or local livelihoods (Edwards *et al*. 2010; Muradian *et al*. 2010; Agrawal *et al*. 2011). In many cases, PES projects have moved forward ahead of scientific understanding, in particular regarding potential synergies or trade-offs among ecosystem services under incentivized land uses, creating the need for research that clarifies these relationships (Bennett *et al*. 2009).

In the case of páramo grasslands, the need for improved scientific knowledge on synergies and trade-offs has grown along with PES programs, which promote changes in land use such as afforestation and changes in burning regimes (Farley *et al.* 2011). Páramos are neotropical alpine grasslands that provide important ecosystem services, including carbon storage, biodiversity, and water supply, with more than 10 million people relying on water from them in the northern Andes (Buytaert *et al.* 2006). Páramo soils receive abundant inputs of belowground organic matter from grass roots, while cool, humid conditions lead to slow rates of decomposition, promoting high soil carbon stores (Luteyn 1992; Hofstede 1997; Podwojewski & Poulenard 2000). As such, páramos play a unique role in the context of global climate change (Buytaert *et al.* 2011). The large stocks of soil carbon in páramos constitute a valuable terrestrial carbon store; at the same time, they are the site of tree plantations established to sequester aboveground carbon for climate change mitigation, an activity which has the potential to either increase or decrease the large soil carbon stocks (Paul *et al.* 2002; Hofstede *et al.* 2002; Farley *et al.* 2004). Despite the importance of ecosystem services provided by páramos, research on their response to land-use change remains limited, and the value of maintaining intact páramo grasslands under various management regimes versus establishing plantations has not been fully evaluated with respect to carbon storage (Buytaert *et al.* 2007; Farley 2007).

Although carbon sequestration is considered a primary benefit of afforestation strategies, it has been noted that, in general, soil C storage is not expected to increase with afforestation in humid grasslands (Paruelo *et al.* 2010). Both increases and decreases in soil carbon have been found following afforestation, though the tree species planted has an important effect on the outcome and the greatest decreases have been found with pine (Guo & Gifford 2002; Laganière *et al.* 2010; Paul *et al.* 2002; Berthrong *et al.* 2009). In the case of páramo grasslands, the establishment of pine plantations was found to significantly lower soil carbon content at one site, with changes occurring within the first 5–10 years following plantation establishment (Farley *et al.* 2004), while an analysis of soil properties under plantations at a range of sites also indicated a tendency towards lower levels of soil organic matter (Hofstede *et al.* 2002) and a third study found similar organic matter levels between the two (Chacón *et al.* 2009). Although pine plantations continue to be established in páramos for carbon sequestration, some efforts have now shifted towards planting *Polylepis racemosa*, a species native to the Peruvian Andes (but frequently referred to as native in the Ecuadorian Andes), based on the assumption that these plantations will provide the service of carbon sequestration without trade-offs to other ecosystem services. This reflects more widely held views about the use of native species to promote biodiversity and prevent "carbon farms" (Edwards *et al.* 2010); however, no data currently exist on effects of plantations with this species on carbon stocks in páramos. Understanding the effects of afforestation on soil carbon is particularly important in tropical alpine grasslands, where the soil carbon stock can be as large as or larger than the aboveground stock of planted forests (Gibbon *et al.* 2010).

While information is limited on the effects of afforestation on ecosystem services, even less is known about other incentivized land uses in páramo grasslands. Globally, improving grassland management has potential to sequester carbon at the same level as forestry sequestration (Conant 2010). In Ecuador, grassland management strategies such as the removal of burning and limits on grazing have been incorporated into a program recently established by the Ecuadorian Ministry of Environment known as SocioPáramo, which seeks to enhance carbon, water, and biodiversity through conservation incentives to participating landowners (MAE 2009). Exclusion of burning can clearly be expected to increase aboveground carbon stores; however the effects on soil carbon are uncertain (González-Pérez *et al.* 2004). Piñeiro *et al.* (2010) have noted that, while it is clear that fire and grazing can alter soil carbon, generalizing the effects is complicated by varied responses across contexts. In the páramo, Hofstede (1995) found that the main effects of burning and grazing on soils were in physical characteristics, while burning itself was not found to have an effect, either positive or negative, on soils. Similarly, Suárez & Medina (2001) did not find significant differences in organic matter in soils with and without burning. However, both note the need for further research on the topic. While grasslands have strong potential to contribute to climate change mitigation, understanding their response to land use is essential to evaluating their potential as carbon sinks (Gibbon *et al.* 2010; Paruelo *et al.* 2010). A better understanding of these effects has implications for programs such as SocioPáramo that seek to enhance production of ecosystem services, including carbon sequestration.

In this research, we evaluated the effects of land use on carbon storage in the páramo. We focused on incentivized land uses intended to enhance ecosystem services production, with the aim of contributing to the discussion on the effectiveness of these conservation measures.

Methods

We conducted our study in a páramo grassland in northern Ecuador owned by the Comuna Zuleta, in the northern Ecuadorian Andes (Figure 1). It includes areas where burning and grazing have been excluded since establishment of a locally managed protected area; areas where either pine or *P. racemosa* have been planted; areas

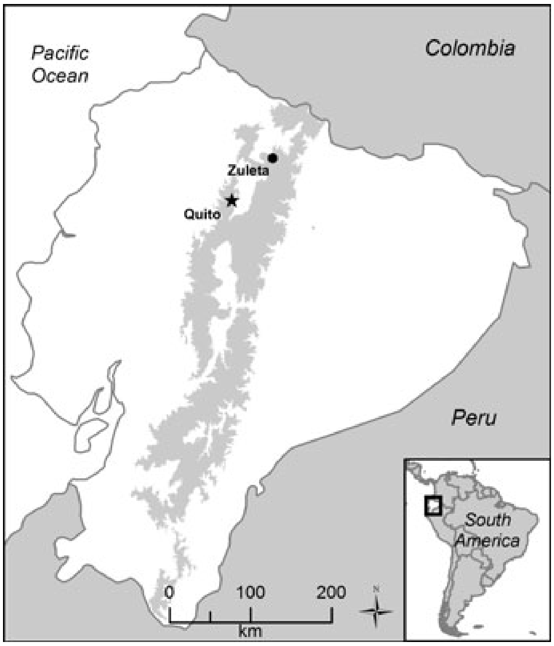

Figure 1 Map of Ecuador indicating the extent of páramo grasslands (in gray) and the location of the study area.

with recent burns; and areas where alpaca grazing has been introduced as an alternative to cattle. We chose eight sites representing different land uses, including: two recently burned sites, one with and one without planted *P. racemosa*; four sites that have not been recently burned (9–15 years since the last burn), two with and two without planted *P. racemosa*; one site planted with *Pinus radiata*; and one former agricultural site planted with *P. racemosa*. Sites were chosen to minimize environmental variation, with all located on 10–20° slopes and between 3,518 and 3,655 m elevation (Table 1).

In each site we established three evenly spaced 20-m transects. We collected soil samples for carbon analysis with a soil corer at 2-m intervals from depths of 0–10 cm and 10–20 cm; each sample had a volume of 21.64 cm^3. We focused on shallower depths, as they are expected to respond most strongly to afforestation (Berthrong *et al.* 2009). Samples were air-dried in the field, transported to the Ecology Program Area Analytical Facility at San Diego State University, ground and sieved through 2 mm mesh, and analyzed for total carbon and nitrogen on a Carlo Erba NCS 2500 elemental analyzer. We also took 11–15 samples per site (3–5 samples per transect) for bulk density measurements and used an average value for each transect in calculating total carbon. We used ANOVA to evaluate whether significant differences in total soil carbon existed at each depth among land uses. The data were checked for normality and equality of

Table 1 Land use, elevation, and slope of the sites included in the study. Time since last burn refers to the time of soil sampling; aboveground biomass was sampled 1 year later at all sites. B = site burned within 1–3 years; UB = site not burned for 9–15 years; P = planted with *Polylepis racemosa*; Ag = former agricultural field. All sites have a history of burning and cattle grazing prior to the establishment of a protected area in 1995

Site	Land use	Elevation (m)	Slope (°)
B-P	Páramo with recent burn (1.5 years since last burn), planted *P. racemosa* (4 years old; some burned in recent fire, some replanting post-fire), no current grazing	3626	20
B	Páramo with recent burn (<1 year since last burn), alpaca grazing	3654	15.5
UB-1	Páramo burned 9 years ago, alpaca grazing	3655	13
UB-P-1	Páramo burned 12 years ago, planted *P. racemosa* (4 yrs old), no current grazing	3610	127
UB-P-2	Burned 15 years ago, planted *P. racemosa* (4 years old)	3643	11.5
UB-2	Burned 15 years ago, no current grazing	3645	12.5
Pine	Planted pine (40 years old), no current grazing	3601	16
Ag-P	Former agriculture site (potato cultivation 10 years prior), planted *P. racemosa* (5 years old), alpaca grazing	3518	12

variances and three outliers were removed from each of the 0–10 cm and 10–20 cm sample data sets. Where differences among land uses were significant, we used Tukey post hoc tests to compare means.

To measure above-ground biomass, we established a 10 m × 20 m plot around each transect, within which we calculated the biomass of all trees. We randomly chose three 2 m × 2 m plots, within which we harvested all shrubs, and a nested 1 m × 1 m plot, within which we harvested the litter and herbaceous biomass. We calculated the total wet weight of shrubs (live and dead) in the 2 m × 2 m plot and of litter, herbaceous live biomass, and herbaceous dead biomass in the 1 m × 1 m plot. We collected a 0.5–1.0 kg subsample of each biomass type, dried it at 70 °C for 24–48 hours or to a constant weight, and calculated the wet:dry ratio. The wet:dry ratio was then used to calculate a dry weight for each biomass type. We then calculated the biomass of all trees within the three 10 m × 20 m plots. All *P. racemosa* individuals <150 cm were treated as shrubs and harvested when

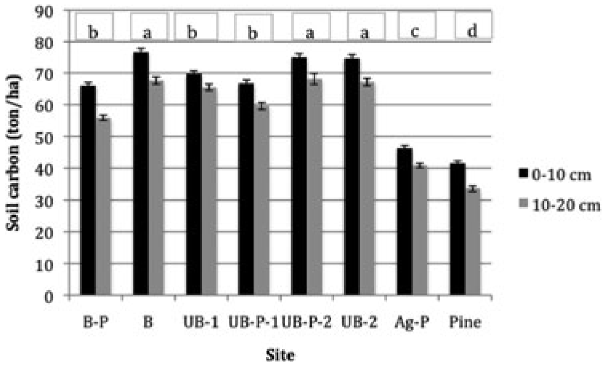

Figure 2 Soil carbon (tons/ha) across land uses. B = site burned within 1–3 years; UB = site not burned for 9–15 years; P = planted with *Polylepis racemosa*; Ag = former agricultural field. Bars with different letters were significantly different ($P < 0.05$) at 0–10 cm depth (groupings were the same for the 10–20 cm depth with the exception of site UB-1 which was in the highest soil C group at that depth).

Table 2 Total carbon (tons/ha ± SE)), including aboveground carbon and soil carbon to 20 cm depth, across land uses (B = site burned within 1–3 years; UB = site not burned for 9–15 years; P = planted with *Polylepis racemosa*; Ag = former agricultural field)

Site	Soil carbon (0–10 cm) (tons/ha)	Soil carbon (10–20 cm) (tons/ha)	Aboveground carbon (tons/ha)	Total carbon (tons/ha)
B-P	66.0 (1.0)	55.9 (0.8)	5.2 (0.2)	127.2 (1.3)
B	76.7 (1.1)	67.6 (1.2)	2.3 (0.4)	146.3 (1.7)
UB-1	69.8 (1.0)	65.5 (1.0)	19.4 (2.7)	154.4 (3.0)
UB-P-1	66.7 (1.1)	59.6 (1.1)	24.1 (4.6)	150.1 (4.9)
UB-P-2	75.0 (1.3)	68.2 (1.7)	14.0 (2.2)	157.0 (3.1)
UB-2	74.7 (1.3)	67.2 (1.2)	22.9 (0.3)	164.9 (1.8)
Pine	41.5 (0.8)	33.5 (0.8)	279.0 (39.9)	354.0 (40.0)
Ag-P	46.2 (0.9)	40.8 (0.7)	17.1 (4.6)	103.1 (4.7)

encountered in 2 m × 2 m plots. For individuals >150 cm, we utilized a less destructive method adapted from Fehse *et al*. (2002). In each 10 m × 20 m plot we counted and measured the height of each *P. racemosa* over 150 cm (those <150 cm were counted as shrubs). We calculated the biomass of 2–3 individuals in the following height categories: 150–200 cm, 200–250 cm, 250–350 cm, and >350 cm. In the 150–200 and 200–250 cm classes we harvested two to three individuals completely. For *P. racemosa* taller than 250 cm, we measured the volume of the bole of each tree, then harvested 25–35% of the crown. We used established wood density values for *P. racemosa* to calculate the biomass of the bole (Fehse *et al*. 2002). To calculate the biomass of the crown we measured the wet weight of the harvested portion, dried a subset to calculate the wet:dry ratio, and calculated the dry weight of the complete crown. We then assigned average biomass values to individuals in each size class, using height for the classes rather than DBH since most individuals were branching at the base. In the pine site, pine biomass was calculated using established allometric equations for tropical/temperate pines based on DBH (Ravindranath & Ostwald 2008). We used this equation because we were not able to find specific allometric equations for high altitude tropical *P. radiata.* Other trees at this site (defined as >1.5 m tall and 2.5 cm DBH) were sampled utilizing the method described above.

Results

We found significant differences in soil carbon among sites, with the lowest levels under pine and former agriculture (75–86 tons C/ha) and the highest levels in the two sites that had not been burned for 15 years (141–143 tons C/ha) as well as one of the recently burned sites (144 tons C/ha) (Figure 2 and Table 2). Soil carbon was significantly higher at all of the grassland sites, regardless of time since last burn, compared with pine or grassland formerly used for agriculture (Table 2). We did not detect a clear effect of planting *P. racemosa* on soil carbon; among the unburned sites, one of those planted with *P. racemosa* had lower soil carbon and one had higher soil carbon than the unburned, unplanted sites (Figure 2).

Aboveground carbon also was different across sites, with the highest levels in the site planted with pine (279 tons C/ha) and the lowest in the recently burned sites (2.3–5.2 tons C/ha), as expected. Differences in aboveground carbon among the four unburned sites were small regardless of whether or not the sites contained *P. racemosa* (Figure 3 and Table 2). In fact, one of the sites that was protected from burning and planted with *P. racemosa* (UB-P-2) had less aboveground carbon than either of the sites that were unburned and unplanted (Figure 3 and Table 2). This is due to the fact that the trees at the planted site were small (<120 cm), while the unplanted sites had large contributions to aboveground carbon from herbaceous biomass (Figure 3). The other unburned site planted with *P. racemosa* (UB-P-1) had higher levels of total aboveground carbon, but most of it was herbaceous rather than being contributed by the planted trees (Figure 3).

It is notable that, when looking at total carbon in both aboveground biomass and in the soil to 20 cm depth, there is little difference between sites with and without planted *P. racemosa* (Table 2). The only site where *P. racemosa* comprised a substantial component of total aboveground biomass was the former agricultural site, where the trees were one year older than in the other sites but much larger, likely due to the history of fertilizer

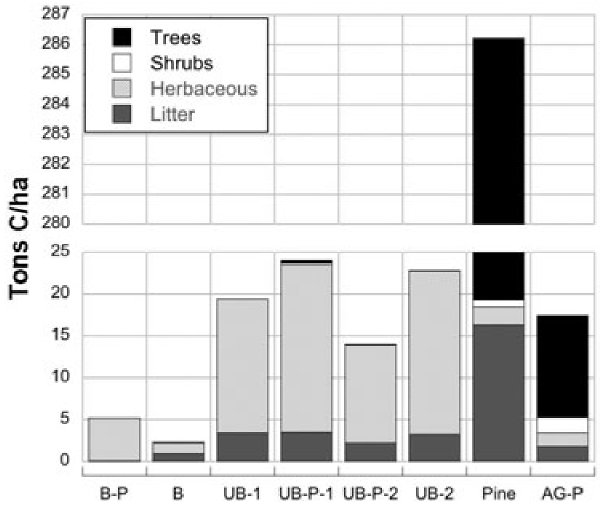

Figure 3 Distribution of aboveground carbon (tons/ha) across land uses. B = site burned within 1–3 years; UB = site not burned for 9–15 years; P = planted with *Polylepis racemosa*; Ag = former agricultural field. Standard errors for total aboveground carbon can be found in Table 2.

use. However, even there, total biomass did not exceed that at the unplanted grassland sites, as the amount contributed by tree biomass was higher but that contributed by the herbaceous component was lower (Figure 3).

Discussion

This research suggests that planting pine is effective for sequestering carbon in tree biomass, particularly if the trees are left unharvested for 40 years or more, as they were in our study site. However, with respect to soil carbon, pine plantations are similar to agricultural sites. This finding supports previous research on pines in the páramo as well as other afforested ecosystems (Farley *et al.* 2004; Berthrong *et al.* 2009) and lends support to the idea that, in some contexts, "…tree plantations are often more akin to an agricultural crop than a forest" (Bekessy & Wintle 2008, pp. 510–511). This finding is relevant in guiding the design of biosequestration schemes and the types of plantations appropriate for them (Lindenmayer *et al.* 2012). In the case of the páramo, planting pine does not help restore one of the primary forms of natural capital in páramo grasslands—that of large soil carbon stores—so climate mitigation programs that seek to promote this type of ecosystem restoration as a cobenefit should use other alternatives.

Given that pine species have been found to result in greater losses of soil carbon and that the species planted can be a significant determinant of the effect on soil C (Berthrong *et al.* 2009; Laganière *et al.* 2010), providing higher levels of support for planting permanent indigenous forests than for monoculture plantations is seen as one way to improve the ability of carbon markets to support both carbon and biodiversity (Bekessy & Wintle 2008). In Ecuadorian páramos, *P. racemosa* has been seen as one such promising alternative to pine. Our findings indicate that, at least at this relatively early stage of stand development, planting *P. racemosa* does not appear to have positive effects on soil carbon; however, it does not appear to have the negative impact on soil carbon that pine can have as early as 5–10 years after plantation establishment (Farley *et al.* 2004). Because few sites exist in Ecuador with older *P. racemosa* plantations, continued monitoring of young plantations will be needed to clarify their long-term effects, while sampling of soils to a greater depth could provide additional insight.

With respect to aboveground carbon sequestration, this study raises questions about the effectiveness of *P. racemosa*. Past research in the Ecuadorian Andes found quantities of aboveground carbon in a 45-year-old native *Alnus* stand similar to those we found under 40 year old pines (241 and 279 tons C/ha, respectively) (Fehse *et al.* 2002), suggesting that some native species have potential to replace pine in incentive schemes while still maintaining high levels of aboveground C sequestration. However, carbon accumulation was modest in the *P. racemosa* plantations at our site. Plantations with *Polylepis* species may be able to achieve higher levels of aboveground carbon accumulation under certain conditions, but few studies exist that provide appropriate comparisons. Fehse *et al.* (2002) found much faster aboveground biomass accumulation in *Polylepis incana* than we found in *P. racemosa*, but noted that the rapid early growth of the native regenerating stands they studied resulted from very high sapling densities unlikely to be found in plantations. In addition, this case highlights the problem of defining what constitutes an indigenous forest, since the genus *Polylepis* is native to Ecuador but the species *P. racemosa* is from Peru, and underscores the point made by Sasaki & Putz (2009) that the way forest cover types are defined can have important implications for whether incentivized land management contributes to conservation, degradation, or an intermediate outcome.

Besides afforestation, incentives to eliminate burning are featured in new conservation programs, and our data indicate that this incentivized land use may have potential for sequestering carbon. Although there was not a clear pattern in soil carbon, with the highest levels at two sites with the longest burn exclusion but also at one recently burned site, our results demonstrate that aboveground C in páramo grasslands can play an important role, even in comparison with some tree species. While efforts to increase terrestrial carbon sequestration have largely focused on afforestation, reforestation, and avoided deforestation, native grasslands can be effective carbon sinks (Bekessy & Wintle 2008; Conant 2010). This case supports the idea that grassland C sequestration should be considered in climate mitigation strategies, and should be decoupled from strategies that focus on tree establishment.

Conclusion

This research illustrates that, although the relationship between land use and ecosystem services provision is generally considered to be well established for carbon sequestration projects (Pattanayak *et al*. 2010), these relationships do not always take belowground carbon into account. Monitoring the effects of afforestation on soil carbon is necessary to fully understand changes in total carbon stocks in these systems (Gibbon *et al*. 2010). At the same time, surprising aspects of the relationship between land use and carbon sequestration may exist in the páramo or other highly productive grasslands, where both belowground and aboveground C storage can be high, even when compared to some types of afforestation. Although these results are from a single study area and representation of each land-use type was limited to the sites available there, they provide a starting point for further investigation of the relationship between land use and carbon storage and underscore the need for experimental research that can provide greater replication and monitoring over time.

In addition to carbon sequestration, improved grassland management has potential to increase rangeland productivity and incomes if strategies can be developed to maximize carbon, productivity, and incomes simultaneously (Conant 2010). Results from our study site suggest that there may be potential for burning on less frequent rotations to improve carbon sequestration, and they provide an initial dataset from which policy-makers who are developing land use recommendations can draw. However, additional research in sites where shrubs make up a larger component of the vegetation under burn exclusion would help clarify the effects more broadly, while evaluations of the outcomes for biodiversity and the livelihoods of those participating in the PES programs would clarify potential synergies and tradeoffs of these strategies.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Study area

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