Grasslands to Tree Plantations: Forest Transition in the Andes of Ecuador

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Over the past four decades the establishment of pine plantations in high altitude páramo grasslands has been a growing land use change in Ecuador. As a result, plantation forestry has transformed some highland landscapes from grasslands to ones dominated by exotic trees. This transformation is analyzed in the context of forest transition theory, which provides a framework for explaining scenarios of increasing forest cover. Forest transition theory predicts that reforestation and afforestation, encompassing the establishment of secondary forests and plantations, respectively, occur when economic development leads to the abandonment of agricultural land or when forest scarcity prompts increases in plantation establishment. This research demonstrates that projected forest scarcity has played an important role in páramo to pine transitions. However, it also indicates that, in Ecuador, afforestation has been seen as a potential means to economic development rather than a consequence of it. Furthermore, this case brings into question some of the assumptions of forest transition theory with respect to the environmental benefits of transition. The evidence presented indicates that, in the case of páramo to pine transitions, the biophysical response includes a loss of soil carbon, nitrogen, and water retention capacity, implying important trade-offs between the ecosystem services provided by páramos and those provided by pine plantations. These results suggest that both the existing land cover prior to forest transition and the type of forest cover established during transition merit more attention in forest transition theory. *Key Words: ecosystem services, forest transition, land use change, páramo grassland, pine plantation.*

Researchers focused on the human dimensions of land use change have called for an improved understanding of the causes of change, particularly the "triggering mechanisms" that promote land use change under different human-environment conditions (Lambin et al. 2001, 263). Patterns of land use can vary greatly even under similar natural conditions due to different social and economic driving forces (Haberl, Batterbury, and Moran 2001). Lambin et al. (2001) suggest that the tendency to simplify the causes of land use change has hampered our understanding of how and why change occurs in some contexts but not others, and that moving beyond these misconceptions requires analysis of the opportunities and constraints for new land uses that operate at local and regional scales. Landscape change researchers have increasingly pointed to the need for a better understanding of the underlying processes by addressing why, how, when, and where land use change occurs (Bürgi, Hersperger, and Schneeberger 2004; Rindfuss et al. 2004).

Along with greater attention to the mechanisms behind land use change, more emphasis has begun to be placed on the role of changing land use in altering the provision of ecosystem services (Lambin et al. 2001; DeFries, Foley, and Asner 2004; Mustard et al. 2004; Rindfuss et al. 2004). The link between land use and ecosystem services was made in broad terms over a decade ago, including the estimate that 43 percent of the world's terrestrial land surface had a diminished capacity to produce ecosystem services as a result of land use change (Daily 1995). However, Daily and Ehrlich (1999) pointed to the paucity of research linking the intensity of land use to the production of ecosystem services, despite the fact that these changes can have broad-reaching implications for the ability of societies to meet a number of human needs. Recently, more research has begun to characterize the function or value of particular ecosystem services, such as carbon sequestration, pollination, or biodiversity, with some explicitly recognizing the ways in which those functions change when land is converted to different uses (e.g., Guo and Gan 2002; Ricketts et al. 2004; Postel and Thompson 2005).[1]

To a large degree, these two areas of research—one focused on the causes of land use change and the other focused on its consequences for ecosystem services—have been carried out separately. Analyzing them jointly requires approaches to understanding both socioeconomic and biophysical components in order to shed light on the conditions that lead to land use change as well as the effects of such change (Klepeis and Turner 2001; Rindfuss et al. 2004). A generalized framework for analyzing these questions would account for the socio-

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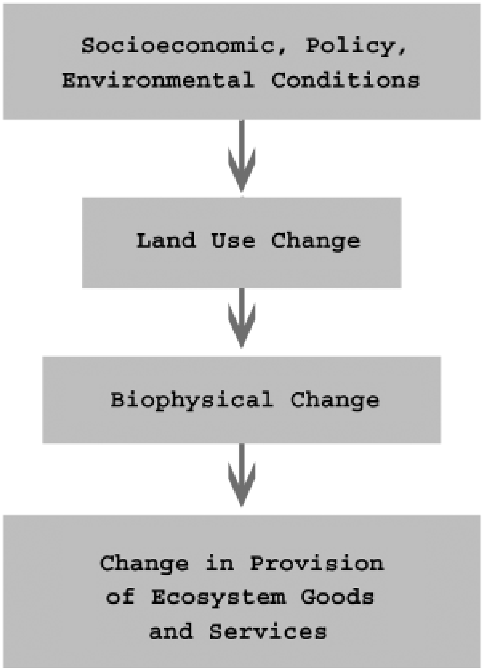

Figure 1. A generalized framework for analyzing the socioeconomic, policy, and environmental conditions that promote land use change, as well as the biophysical response to that change, which influences the type and quantity of ecosystem goods and services produced.

economic, policy, and environmental conditions that act as the mechanisms that promote and trigger land use change. In those locations where land use change does occur it induces a biophysical response, which then influences the suite of ecosystem goods and services that can be produced in that landscape (Figure 1). This framework can be seen as an extension of political and cultural ecological approaches that focus on the evaluation of the past and current roles of human agency in altering nature through land use (e.g., Walker 2003; Butzer and Helgren 2005; Byers 2005; Chowdhury and Turner 2006; Caldas et al. 2007; Dull 2007), while also recognizing that nature, whether in its "natural" or modified form, is an agent that shapes the way humans can use those landscapes (Zimmerer 2007a). This article employs such a framework by jointly analyzing the socioeconomic and policy factors that have influenced the transition from páramo grasslands to tree plantations in the Ecuadorian Andes, as well as the ways in which this transition has altered the provision of several key ecosystem services.

Forest Transition Theory

Much of the focus of land use change research has been on those processes that are occurring over large areas, including deforestation, rangeland degradation, agricultural intensification, and urbanization (Lambin et al. 2001). However, at the same time that these losses of vegetative cover or quality of cover have been documented, research on forest transitions has focused on explaining scenarios of increasing forest cover. These forest transitions occur when periods of forest loss shift to periods of net gain in forest cover, and encompass the reversion of previously forested lands to secondary forests as well as the establishment of plantations on nonforested land (Mather 1992; Rudel 1998; Rudel et al. 2005; Perz 2007).

In addition to characterizing the nature and extent of this land use change, frameworks have been developed to explain the underlying processes promoting change. Forest transitions have been described as "management" transitions (Mather 1992, 367), which may be brought on by factors such as changing population trends, changing demands for forest products and services, and changes in the ways in which forests are valued and perceived (Mather 1992). More recently, two broad categories have been suggested to characterize the causes of transitions from nonforested to forested land: the "economic development path" and the "forest scarcity path" (Rudel 1998; Rudel et al. 2005, 24). The first occurs as agricultural land is abandoned and reverts to forest, either by regeneration or planting; the second occurs in response to a shortage of forest products, with increasing prices for forest products prompting landowners to plant trees (Mather and Needle 1998; Rudel et al. 2005). However, these paths have primarily been exemplified though the experiences of developed countries. Alternative versions of these paths may exist for developing countries, where socioeconomic conditions are different and where the establishment of more forest cover does not necessarily imply that land has been abandoned (Rudel, Bates, and Machinguiashi 2002; Klooster 2003; Perz and Skole 2003).

Furthermore, it has been noted that not all locations are equally likely to experience land use change, and some "attractors" or "precursors" of change may exist that make change more likely to occur under some conditions than others (Bürgi, Hersperger, and Schneeberger 2004, 857). Land use change may be triggered by a variety of factors including land availability or scarcity, changes in markets, subsidy programs, sponsored projects, and other types of interventions (Lambin et al. 2001), and the institutions that mediate access to resources can strongly influence how land is used and can ultimately influence land quality (Batterbury and Bebbington 1999; Bebbington and Batterbury 2001). Although broad categories can be used to characterize the causes of forest transitions generally, Mather (1992, 372) notes that "details and proximate causes may vary greatly from country to country," pointing to the need for studies that examine the conditions that promote forest transitions in particular locations (Rudel et al. 2005). These national differences in the extent and causes of forest transitions can be strongly affected by institutions that provide technology or capital or by differing access to markets; moreover, government policies that promote forest transitions, in many cases, play key roles in how land is used (Rudel et al. 2005).

Forest Transitions and Ecosystem Services

Bürgi, Hersperger, and Schneeberger (2004, 861) note that one of the objectives of land management is to aim it in "more desirable directions." In the case of forest transitions, the establishment of a greater forested area tends to be treated as necessarily leading to positive environmental outcomes. The negative environmental consequences of past deforestation, such as soil erosion, are contrasted with the potential for reforestation to help control problems of soil degradation and erosion (Mather 1992). As the conception of ecosystem services provided by forests has broadened beyond the focus on soil erosion, forest transitions have been cited for their potential to also improve water quality and sequester carbon, which may help slow climate change (Klooster 2003; Rudel et al. 2005). Furthermore, these transitions are described as "adjustment[s] of agriculture to land capability" through which greater agricultural production on lands with high production capability allows for lower capability land to become available for reforestation (Mather and Needle 1998, 117). Implicit in this transition is that the forest cover will be more environmentally beneficial than the previous land cover/use.

However, critiques of forest transition theory have pointed out that it has inadequately distinguished among different forest types and the benefits derived from them (Perz 2007). Although early descriptions of forest transitions acknowledged that reforested land may not have all the same qualities of the original forest, particularly in terms of plant and animal diversity (Mather 1992; Rudel 1998), little differentiation was made among the types of forests that can be established as part of a forest transition in terms of other biophysical implications. Perz (2007, 108) points out that these "biophysical differences bear social implications, for the structure and composition of the vegetation greatly influence the natural resources available. . . ." This may be particularly true in the distinction between naturally regenerating secondary forests and plantations, both of which increase the area forested, but may induce a very different biophysical response. Transitions to plantations have been questioned in terms of whether they make any contribution to biodiversity, but still have been assumed to be otherwise beneficial and worth promoting. Plantations have been described as doing "little to conserve biodiversity, but they do sequester carbon and conserve soil, so governments should place a high priority on promoting them" (Rudel et al. 2005, 23).

While forest transition theory does not distinguish among the types of forest that make up the new land cover, neither does it clearly distinguish among previous land cover types. Many of the forest transitions documented have occurred as marginal agricultural land was abandoned, but studies also have documented shifts from rangeland to plantation (Mather and Needle 1998; Mather 2004). In the latter case, afforestation as a land use change shares a number of commonalities with agricultural intensification in the sense that it involves increased inputs (e.g., seedlings, labor for planting and harvesting, and machinery for harvest) in order to produce higher outputs than traditional uses of the land (e.g., timber, firewood, and, more recently, carbon, rather than wool and meat). This leaves open the question of how this land use change affects the provision of ecosystem goods and services, and whether the environmental outcomes are necessarily more beneficial than the previous land use. In order to determine whether governments should, in fact, promote these transitions, this question must be more clearly addressed. As noted by Rudel et al. (2005, 23–24), "The significance of forest transitions in creating more sustainable societies depends on the effects of the transitions on the environmental services that forests provide."

Background and Research Problem

Over the past four decades the establishment of pine plantations in high altitude páramo grasslands has been a growing land use change in Ecuador. As a result, plantation forestry has transformed some highland landscapes from grasslands to ones dominated by exotic trees. In order to understand this forest transition more completely, it is necessary to know the characteristics of both the land cover from which the transition begins and the type of forest cover that takes its place.

Páramo Grasslands: Ecological History and Land Use

Páramos are equatorial alpine grasslands that are found generally between 3,200 and 4,700 m above sea level and between 11° north and 8° south latitude, with an estimated extension of 1.3 million ha in Ecuador (Luteyn 1992; Medina et al. 1997). Páramos have very high levels of biodiversity, with an estimated 2,000 species of vascular plants in Ecuadorian páramos and 3,000 to 4,000 species throughout their range, of which as many as 60 percent may be endemic to these ecosystems (Luteyn 1992; Medina et al. 1997).

The Quaternary history of the páramos suggests that "most of the time the vegetation cover was in a state of flux" (van der Hammen and Cleef 1986, 194). This state of flux is relevant to debates on the degree to which today's páramos may be considered of natural or anthropogenic origin. Some early observers focused on the roles of precipitation and temperature in limiting tree growth and allowing for the development of páramo grasslands (Troll 1968). However, others have suggested that páramos are largely an artifact of human activities that have lowered the treeline and created grasslands in areas that were once forested. Ellenberg (1979) viewed forest as the climax vegetation in much of what is currently páramo and argued that under "natural conditions" much of the high tropical Andes would be forested. Patches of forest within páramos have been cited as evidence of past forest cover, and the growth of exotic species at these altitudes has been seen as evidence that tree growth is possible under current climatic conditions (Ellenberg 1979; Gade 1999). Continued research on the influence of humans on tropical Andean vegetation has not fully resolved this question. It is thought that treeline in the northern Andes has been lowered by as much as a few hundred meters due to human activities (Kok, Verweij, and Beukema 1995; Hofstede 1998). However, many argue that this does not imply that the páramo is of anthropogenic origin, but rather that its limits have been expanded through human influence (Ramsay and Oxley 1996).

The possibility of tree growth in the páramo provides the historical context for what has become a growing land use there. The most common human activities in páramos continue to be grazing of livestock (frequently combined with burning to improve availability of forage), along with agriculture and the collection of woody species for firewood (Parsons 1982; Luteyn 1992; Ulloa and Jørgensen 1995). In addition to these long-established uses, over the past four decades the use of páramos for pine plantations has become a more common practice (L. Suárez 1989; White 2001). Monterey pine is considered to have an elevational limit of 3,800 m in Ecuador (Jongsma 1998), allowing it to be planted in the páramo vegetation belt and transforming these landscapes from grasslands to trees.

Research on the ecological effects of land use in páramos has been directed primarily toward the practices of burning and grazing. Burning in the páramo has been found to cause the destruction of much of the bunchgrass cover, which later regenerates from the base, while eliminating most of the shrubs (Janzen 1973; Williamson et al. 1986; Horn 1989). As a result, the composition of the vegetation is altered as the more fire-tolerant vegetation gains greater dominance while some fire-sensitive species, including many woody plants, fail to regenerate (Williamson et al. 1986; Hofstede 1998; Keating 1998). When livestock grazing occurs after the páramo is burned, the regeneration of bunchgrasses may be impeded as the regrowing tussocks are consumed and trampled (Hofstede 1998). In cases where burning and grazing are repetitive, they can cause a reduction in vegetative cover, which begins a positive feedback: less plant cover leads to fewer nutrients in the system—particularly in páramo soils where released nutrients tend to be quickly immobilized—resulting in lower productivity and slower regrowth (Hofstede 1995). The intensity of this land use varies from location to location within the Ecuadorian Andes, leading to a range of conditions in páramo grasslands, from lightly burned to highly degraded (E. Suárez and Medina 2001). Research on the effects of burning and grazing in Ecuadorian páramos is still relatively limited, but even less research has been done on the types of biophysical changes that may occur in response to the establishment of tree plantations in páramo ecosystems (an exception is Hofstede et al. 2002).

Pine Plantations: Early History

There is a long history of planting exotic trees in the Andes, dating to the mid-1800s in parts of Ecuador, Peru, and Bolivia (Dickenson 1969; Gade 1999). The earliest plantations were established with objectives focused on meeting fuel and timber needs and, in some cases, with the intention of improving soils that had been degraded by intensive grazing (Gade 1999). Eucalyptus was planted widely at the end of the nineteenth century in Ecuador, then in the twentieth century pine began to be used, either for reforestation or for afforestation of grasslands. In many cases, particularly with eucalyptus, these plantations of exotic species were established with noncommercial objectives and were intended to restore deforested and/or degraded landscapes. Most state-sponsored forestry projects in Ecuador, as well as in other parts of the Central Andes (Young 1998), have used species of eucalyptus and pine rather than native species, about which little silvicultural information tends to be available.

The first trials with Monterey pine (*Pinus radiata*) and seventy other species of conifers with potential for timber production were begun in the 1920s in Cotopaxi province. They were initiated by former Ecuadorian president Luciano Andrade Marín, whose conception of altitudinal zones as being interchangeable with latitudinal zones (cited in Troll 1968) led him to believe that species from temperate zones could successfully be planted in "corresponding" temperature zones in the tropical mountains of Ecuador (C. Aguirre, Director of the Research Division of the *Estación Experimental Conocoto*, Conocoto, Ecuador, in-person interview, 22 June 1999). Monterey pine was one of the species considered to have the best potential for growth in Ecuador, but it only began to be planted extensively in the late 1960s, after the Ministry of Agriculture (MAG), through the Ecuadorian Forestry Service (later to become INEFAN, *Instituto Ecuatoriano Forestal y de Áreas Naturales y Vida Silvestre*), began programs that provided incentives and subsidies for plantation establishment (C. Aguirre, in-person interview, 22 June 1999).

Research Problem

Despite the growth of plantation forestry in the Ecuadorian Andes, little research has been done on the factors that have been influential in bringing about this landscape change, or whether plantation establishment has produced the desired outcomes in terms of improved biophysical conditions. In this article, I address this gap while putting this land use change in the broader context of forest transition theory and the mechanisms and biophysical outcomes that have been associated with these transitions. Specifically, I address the following questions:

1. What are the socioeconomic and policy mechanisms that have been involved in the páramo to pine transition in Ecuador?
2. Has this transition conformed to either the economic development path or the forest scarcity path proposed in forest transition theory (in the sense of Rudel et al. 2005)?
3. To what degree can this transition be seen as directing land management toward more desirable conditions? Specifically, given the objectives of forest transitions with respect to improvement of soil quality, to what degree have they been accomplished under páramo to pine transitions?

Methods

This research draws on "hybrid" social and physical research methods (Batterbury, Forsyth, and Thomson 1997; Chowdhury and Turner 2006; Zimmerer 2007b) that are used to identify and understand the causes and consequences of páramo to pine transitions in Ecuador. It shares similarities to Klepeis and Turner's (2001, 29) "integrated land history" in that it seeks to identify the processes affecting land use as well as to demonstrate the ecological impacts of those uses. At the same time, this work contributes to a growing body of geographical political ecology that is situated at the interface between social and natural sciences (Zimmerer and Bassett 2003; Byers 2005; Benjaminsen et al. 2006). Research situated within these "environmental borderlands" bridges human and biophysical systems in a way that allows for the evaluation of both political and ecological processes (Zimmerer 2007b).

I conducted research on the socioeconomic and policy aspects of páramo to pine transitions using a "multimethod" research design (Zimmerer and Bassett 2003, 9; Zimmerer 2007a, 10). I conducted archival work in Ecuador on government policy with regard to the plantation forestry sector as well as on government contracts for the establishment of plantations in specific areas. Because much of the history of plantation forestry in Ecuador has not been recorded in written documents, I also conducted semistructured interviews with representatives of the former Forestry Service, with representatives of the Ministry of Environment in Quito and Cotopaxi, with individual plantation owners and managers, and with representatives of nongovernmental organizations involved in plantation forestry. These sources allowed me to create a history of forestry policy with respect to plantations since the time they began to be established in Ecuador.

I conducted research on the biophysical response to this landscape change at the Aglomerados Cotopaxi, S.A. (ACOSA) plantation (0°40′ S, 78°30′ W) in Cotopaxi Province, Ecuador (Figure 2). ACOSA is the largest pine plantation in the country, covering approximately 7,700 ha to the northwest of Cotopaxi volcano and bordering Cotopaxi National Park. Because there were multiple owners of the areas that now make up the ACOSA plantation, land uses prior to plantation varied. At lower elevations, below 3,400 masl (meters above sea level), parts of the plantation that previously belonged to private estates were used for cattle, sheep, or llama grazing and/or agriculture prior to plantation, and some areas were previously mined for pumice. However, at higher altitudes, where I examined biophysical response, agriculture was not practiced and the páramo vegetation was intact at the time of plantation (J. P. Fontecilla, Director of Forestry, Aglomerados Cotopaxi, S.A., Cotopaxi, Ecuador, in-person interview, 28 July 1999

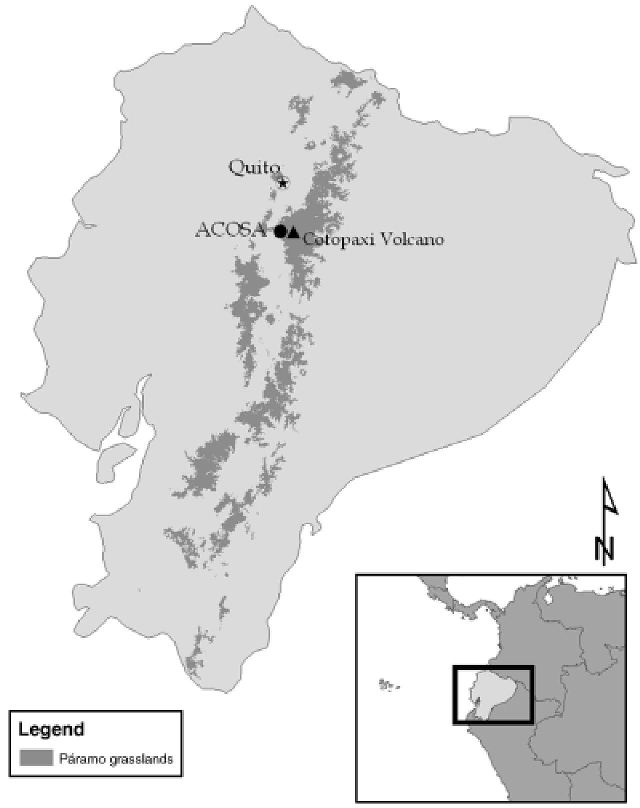

Figure 2. Map of Ecuador, including the extension of páramo grasslands and the location of the Aglomerados, S.A. (ACOSA) plantation.

and 3 April 2001). I examined Monterey pine, which covers 60 percent of the plantation and is planted in the grasslands without any clearing or burning of the existing vegetation before the first planting (rather, pine is planted between bunches of grass, such that direct disturbance from planting is localized). All stands examined were unmanaged (no thinning or pruning). The plantation included stands ranging from 0 to 25 years of age, which were divided into four age classes to establish a chronosequence: páramo grassland (representing age 0), young pine (5–10 years old), intermediate-aged pine (15–20 years old), and old pine (20–25 years old).[2] I located three stands in each vegetation age class (except the 5–10-year-old pine, for which only one stand could be located); the grassland plots were located adjacent to the plantation in sites that had not previously been planted (but were scheduled for planting). In each stand, I collected ten randomly located soil samples from the 0–10 cm layer, and analyzed them for total carbon and nitrogen. I also collected samples by horizon in two soil pits per stand (in all cases, profiles consisted of A, AC, and C horizons, with buried horizons below the C layer) and analyzed them for carbon, nitrogen, and water retention (water retention samples were from the A horizon only; for details of laboratory and statistical methods and full biophysical results, see Farley and Kelly 2004; Farley, Kelly, and Hofstede 2004).

Mechanisms of Forest Transition

Land use change, generally, and forest transition specifically, occurs in response to a variety of mechanisms that provide opportunities and constraints for adopting new land uses (Mather 1992; Lambin et al. 2001; Mather 2004; Rudel et al. 2005). In particular, government programs and policies that promote forest transitions, as well as economic and trade policies, can play important roles in determining to what degree these transitions are carried out (Mather 1992; Rudel et al. 2005). In this section, I review the key policies and programs that have been influential in the conversion of páramo to pine in Ecuador and evaluate the degree to which these can be seen as contributing to the economic development path or the forest scarcity path.

Forestry Policies

The Ecuadorian government was involved in the initiation of many of the country's early plantations and began to actively promote them among private landholders beginning in the 1960s. At that time, the government began to finance programs in which exotic trees were planted throughout the Ecuadorian Andes, mostly with domestic funding but in some cases with loans from international lending institutions. As in other parts of the Andes, the objectives of the earliest plantations focused on meeting fuel needs and attempting to restore degraded landscapes, whereas later plantations began to be seen as potentially important timber producers and contributors to the national economy (Luzuriaga 1995; Gade 1999; COMAFORS-IPS 2001).

A number of incentives to promote forestry projects in Ecuador have been included in national forestry laws and have heavily influenced the plantation of exotic species in highland Ecuador. Pine plantations have been promoted through government programs, in particular through agreements between the Ministry of Agriculture and other public or private rural landholders. The programs initiated in the 1970s involved a small number of landowners and marked the beginning of pine plantations on a relatively wide scale in Ecuador. In these cases, the landholders provided land for the plantation in exchange for 30 percent of the timber, while MAG planted the trees, with an agreement to receive the other 70 percent of the timber at the time of harvest (J. Fontecilla, in-person interview, 28 July 1999).

Although MAG also began working with campesino organizations in the early 1970s, community-owned plantations did not expand until the next decade. In 1982, MAG began *Convenios de Participación*, a program in which the ministry paid the entire cost of new plantations with rights to 50 percent of the timber produced. The communities provided land and labor and agreed to maintain and manage the plantations, in exchange for the other 50 percent of the production. Because the total cost of the plantation included preparation of the land and planting, among other costs, the communities, in effect, were paid for planting pine in addition to the promise of half of the income at the time of harvesting (R. Yaguache, Coordinator of *Desarrollo Forestal Campesino*, Northern Zone, Ibarra, Imbabura Province, Ecuador, in-person interview, 31 May 2001). These programs were given institutional support from the Forestry Law of 1981, which set the goal of converting 127,000 ha to forestry by 1984, of which 100,000 ha were to be planted by the government and the rest by the private sector. The law declared that it was the obligation of the government and in the public interest to afforest and reforest both publicly and privately owned land for which forestry was designated as a potential land use. Other stated objectives of these programs were to increase a renewable resource in rural areas and augment the production of "goods and services" that were expected to be provided by the plantations (MAG-Cooperativa El Abra 1982).

During this era, two new sources of financing emerged for these forestry programs. The first, FONAFOR (*Fondo Nacional para la Forestación*), was a domestic program in which forestry was funded with a portion of the profits from petroleum exports, in effect using one export sector to augment the growth of another potential export. The largest program of this era, *Plan Bosque*, provided 100 percent financing of plantations by the government, with repayment by the community without interest at the time of harvesting (N. Bedón, Ministry of Environment, Cotopaxi Regional Office, Latacunga, Ecuador, in-person interview, 9 May 2001). The second source of financing was international, with funding coming in the form of a $4 million loan from the Inter-American Development Bank to the Ecuadorian government aimed at planting 18,000 ha to pine and eucalyptus in degraded areas owned by low-income communities and individuals. The participants were paid for their labor in planting the trees on their land, in addition to an agreement to receive 70 percent of the timber at the time of harvesting (IDB 1999). Whereas earlier programs only referred very generally to the goods and services that were expected to be provided by the plantations, this program sought more explicitly to reforest or afforest land with low productivity for agriculture and grazing due to altitude (3,300–3,700 masl) and topography (slopes between 30–100 percent). In addition, it sought to revitalize government-sponsored reforestation programs that had stalled (IDB 1999). By the next decade, government programs began to decline, providing much lower levels of funding to landowners and suffering from inadequate funding (INEFAN 1995). The main program in the 1990s, PLANFOR (*Plan Maestro de Forestación*), was an example of this. It required all but the smallest landowners (those holding less than 10 ha) to provide the initial financing for the plantations, with government repayment coming only after they were established, and it required that the land be provided as collateral. As a result, few landowners participated and plantations that were established under this program were mainly small in extension (N. Bedón, in-person interview, 9 May 2001).

The majority of the area planted to pine in the Ecuadorian highlands, especially that under community ownership, was planted under one of the programs described above. Other incentives, in the form of exoneration from taxes, also have been included in the forestry laws, including the *Ley Forestal y de Conservación de Áreas Naturales y Vida Silvestre* of 1998. However, these incentives have affected only the larger companies involved in the forestry sector, as communities generally were not paying income or property taxes, nor did they import the kinds of materials that were exempt from taxes.

Carbon Sequestration

The factors that promoted páramo to pine transitions in Ecuador began to change in the 1990s. The government-sponsored programs that were so influential in the decades between 1960 and 1990 ceased to exist, and factors at the international level came to play an increasingly important role (Table 1). Government subsidies disappeared and new sources of financing for plantations appeared, as renewed interest in tree plantations was prompted by their potential as carbon sinks and sources of carbon credits (Brown, Lugo, and Chapman 1986; Wright, DiNicola, and Gaitan 2000; Smith and Scherr 2002).

This impetus for tree plantations has developed out of policy and market mechanisms aimed at decreasing the atmospheric concentration of carbon dioxide (CO_2). The Kyoto Protocol to the United Nations Framework Convention on Climate Change (UNFCCC) includes the objectives of reducing emissions and increasing fixation of CO_2 and outlines mechanisms by which a portion of a country's excess emissions can be compensated

Table 1. Initiatives or phases in which the plantation of pine was promoted in the Ecuadorian Andes

Initiative/phase	Time period	Origin	Goals
First plantations	Late 1800s–early 1900s	National	Meet fuel and timber needs, erosion control
Forestry trials	1920s	National	Timber production
First MAG-sponsored programs	1970s	National	n/a
Convenios de Participación	1980s	National	Increase renewable resource, production of "goods and services"
FONAFOR/*Plan Bosque*	1980s	National	Timber for export
IDB	1980s	International	Utilize low productivity land, revitalize government forestry programs
PLANFOR	1990s	National	Promote reforestation
PROFAFOR	1990s–	International	Carbon sequestration

Notes: MAG = Ministry of Agriculture; FONAFOR = *Fondo Nacional para la Forestación;* IDB = Inter-American Development Bank; PLANFOR = *Plan Maestro de Forestación*; PROFAFOR = *Programa FACE (Forests Absorbing Carbon Dioxide) de Forestación*; n/a = information on the goals of this project were not specified in any of the documents reviewed.

for by increasing carbon fixation in other countries. Through the Clean Development Mechanism (CDM), countries with emissions limits can finance programs in developing countries, which have no CO_2 emission limits under the Protocol. These projects must increase fixation of CO_2 and simultaneously contribute to the development of the host country (Vine, Sathaye, and Makundi 2001). Emission reduction units (ERUs) are granted for projects that provide for the reduction of greenhouse gases through the creation of carbon sinks, and these ERUs can then be used to meet part of a country's emission reduction commitments (UNFCCC 2001).

It has been suggested that forest plantations in Latin America could play a large role in carbon sequestration, and that more than 1 million ha in Latin America could be converted to plantations (Wright, DiNicola, Gaitan 2000; Geary 2001). Pilot forestry projects began in Latin America in the late 1990s under a voluntary phase of the CDM, with two in Ecuador as well as several in Costa Rica, Mexico, Bolivia, Panama, and Belize (Heerma van Voss 1999). In Ecuador, the largest effort at compensating CO_2 emissions through forestry has been carried out by FACE (Forests Absorbing Carbon Dioxide Emissions, a consortium of Dutch electricity companies), which began with the objective of planting 75,000 ha in Ecuador (Heerma van Voss 1999). These activities have been carried out through PROFAFOR (*Programa FACE de Forestación*), which contracts with rural landowners to establish and maintain plantations in Ecuador. By 2006, this effort had resulted in 22,000 ha of plantations in highland Ecuador, 94 percent of which are planted with pine (L. F. Jara, General Manager of PROFAFOR del Ecuador, S.A., Quito, Ecuador, personal communication, 2 August 2006). PROFAFOR now has contracts with 146 landowners throughout the country (L. F. Jara, personal communication, 2 August 2006), suggesting that the number of actors involved is continuing to grow. Additionally, a National Office for the Promotion of the CDM in Ecuador (CORDELIM) was established within the Ministry of Environment and currently has several forestry projects submitted for approval (http://www.cordelim.net). Carbon sequestration has provided new incentives for planting in Ecuador, with the sale of carbon credits being seen as a potentially large source of income for the country (COMAFORS-IPS 2001).

Economic Policies

Economic policies have promoted pine plantations indirectly, both by promoting domestic production in the 1970s and by encouraging production for export in the 1980s and beyond. Ecuador followed a series of changes in macroeconomic policy similar to that of other Latin American countries in the second half of the twentieth century. In the 1960s and 1970s, import substitution industrialization (ISI) was adopted, and, as economic policy became geared toward the production of goods for internal markets, government incentives were used to promote the development of national industries (Luzuriaga 1995). In this context, and given that Ecuador was considered to have strong potential in the forestry sector, sawmills were promoted with government subsidies but under private ownership. This then became a factor in the subsequent government promotion of plantations to supply the newly established sawmills with primary material (M. Añazco, Technical Advisor, *Desarrollo Forestal Campesino*, Conocoto, Ecuador, in-person interview, 17 May 2001).

Due in part to increasing external debt, economic policy in the 1970s and 1980s shifted toward an emphasis on exports (Luzuriaga 1995; UNDP 1998). These policies pushed not only an increase in exports, but also a reduction in the country's dependency on oil through the expansion of other export sectors. In this context, national forestry policies promoted the establishment of plantations as a potential export sector, in addition to continuing to supply internal markets (Luzuriaga 1995; COMAFORS-IPS 2001). The perception of plantation forestry as an economic sector with potential to provide large financial returns has been drawn, in part, from the Chilean experience (Ponce 2000). Through incentives and financial support provided by the Chilean government for plantation forestry, the area planted increased from 290,000 ha in 1974 to 1.7 million ha in 1998 and earnings from exports of wood products grew from $39 million in 1973 to $1 billion in 1992 (Clapp 1995; Lara and Veblen 1994; Ponce 2000). Because of this rapid growth, Chile is cited by some as an example of the economic benefits Ecuador could generate by expanding plantation forestry (Ponce 2000).

Although the forestry sector in Ecuador (including native forests and plantations) contributed less than 2 percent of the GNP in 2000, representing $253 million (Ponce 2000; COMAFORS-IPS 2001),[3] as an export sector forestry played an increasingly important role in the 1990s. The value of wood exports rose from less than $30 million in 1990 to more than $103 million in 1997, making it the seventh largest export sector (Cerda 1999; Ponce 2000). Although the total area under plantation was less than 1 percent of the total forested area in the country, 13 percent of the 9.7 million m^3 of timber extracted in 1997 came from plantations, indicating that plantations had become important contributors to total timber production (Cerda 1999). By the mid-1990s, approximately 43 percent of the timber produced from plantations in Ecuador was for export, with the majority of these exports, particularly pine, going to Japan, followed by South Korea and Taiwan (INEFAN 1995; Cerda 1999).

Loss of Native Forests

The focus on producing timber was present in the earliest plantations in Ecuador and in most of the government programs established in the 1970s through the 1990s. This concern over the adequacy of timber supplies continues to be important, as deforestation has continued at a rapid rate in much of lowland Ecuador and timber producers have begun to shift their focus toward plantations as timber sources for the future (COMAFORS-IPS 2001).

The rate of deforestation in Ecuador is not well-established for the country as a whole (R. Sierra, Assistant Professor, Department of Geography, University of Texas, Austin, TX, personal communication, 25 April 2007), however Wunder (2000) cites net deforestation rates of 189,000 ha per year for 1980–1990 and 238,000 ha per year for 1990–1995.[4] Currently, most timber is extracted from the Coast region, with 70 percent of the timber produced from native forests coming from the northwestern province of Esmeraldas (UNDP 1998). In this province, where by 1996 only 18 percent of the original forest area was still intact (Sierra, Tirado, and Palacios 2003), much of the timber extraction has been done without licenses and without complying with management plans or requirements for replanting, leading to rapid depletion of forests there. In the Amazon region, which currently provides a large portion of the primary materials to sawmills in the highlands, the rate of deforestation has been estimated at 54,000–60,000 ha per year (INEFAN 1995; Wunder 2000; R. Sierra, personal communication, 25 April 2007). In addition to the rapid rate of extraction, a large amount of timber is wasted under current extraction processes (Wunder 1996), as 98 percent of sawmills are supplied with timber primarily from *motosierristas*, individuals who work independently and exploit timber using chainsaws, then sell the trees to intermediaries who sell the wood to timber companies (INEFAN 1995; Sierra, Tirado, and Palacios 2003). This type of timber extraction is largely uncontrolled and tends to result in a large amount of wasted timber due to incorrect dimensions and defects, so that extraction is further increased to meet the demand (INEFAN 1995).

Focus has begun to shift toward plantations as a timber source as native forests become depleted and the cost of harvesting less-accessible forests increases (COMAFORS-IPS 2001). Lack of primary material is cited as one of the primary limitations for the forestry sector in Ecuador, so that projections of its growth depend largely on the area afforested or reforested per year (INEFAN 1995; Cerda 1999). COMAFORS-IPS (2001), a timber industry group, estimates that, in order to meet the industry demand, between 160,000 and 240,000 ha would need to be planted to supplement extraction from the remaining nonprotected native forests. The species that are planted will depend largely on the areas in which land is available for this purpose, but because pine is fast-growing and has established markets it will likely continue to be a dominant species in highland plantations. According to Cerda (1999), the principal timber companies in Ecuador already have begun to shift their focus toward plantations rather than native forests as the source of their primary material in order to ensure that the supply will continue into the future.

Economic Development or Forest Scarcity?

Forestry policy in Ecuador has promoted the establishment of pine for a variety of reasons; however, through the course of most of its forestry history the production of timber has been an explicit goal (cf. Table 1). At the same time, economic policy, both in phases when it was directed inward and in those when it was oriented externally, has incorporated plantations as a means of producing timber either for domestic consumption or for export. And, more recently, global climate policy has played a role in creating a new market for carbon credits that can be produced through the establishment of plantations. All of these factors suggest that, in the case of páramo to plantation transitions, the establishment of plantations has functioned as a means of economic development rather than a consequence of it. This transition has not occurred as the process of economic development reached a stage at which the land-based population decreased and land was abandoned, but rather reflects an effort to develop new means of production on lands that have traditionally been considered marginal (INEFAN 1995).

Forest transition theory posits that during the process of economic development the most marginal lands will be abandoned and revert to forest in response to decreasing pressures for the expansion of agriculture (Mather 1992; Mather and Needle 1998). However, the data from highland Ecuador suggest a very different relationship between economic development and afforestation. As in the lowlands of Ecuador (Rudel, Bates, and Machinguiashi 2002), the process of increasing forest cover in the páramo has not followed a process of land abandonment, but rather one of land retention or acquisition. Specifically, the low cost of land in páramo grasslands, coupled with high rural unemployment and underemployment rates that allow for inexpensive labor, have been a draw for some actors to establish plantations there (Ponce 2000). This suggests that not only is land abandonment not necessarily a precursor to afforestation but that plantation establishment can actually be carried out with the objective of increasing the intensity of production on the land rather than relinquishing it to a less-intensive, lower productivity use.

Forest transition theory suggests that forest scarcity can provide a second path to increasing forest area (Mather 1992). As noted, most of the forestry policies described have included the establishment of a larger timber supply as at least one of their objectives (Table 1). This history, combined with the more recent emphasis from the forest sector on increasing plantation area as a response to deforestation, suggests that the páramo to pine transition can be considered, at least in part, a response to forest scarcity. Through most of the course of Ecuador's forest history, plantation establishment has been more of a response to a projected scarcity than an actual scarcity, but this is consistent with the forest scarcity path as defined by Grainger (1995): the expectation that current forest supply will not meet future demand. Furthermore, the continued high rate of deforestation suggests that the forest scarcity path is likely to be an increasingly important component of plantation establishment in Ecuador.

In addition to these two primary paths, forest transition theory points to changes in the way forests are valued and perceived as one of the factors that can influence transition (Mather 1992). In the case of Ecuador, the assumed or perceived benefits of forests are part of a societal view of forests as broadly beneficial, as reflected in statements in some of the forestry policies where reforestation itself was the stated goal (e.g., PLANFOR, Table 1). At the same time, páramo grasslands have been seen as marginal and in need of having their value augmented, as suggested in some of the stated policy goals (e.g., IDB, Table 1).[5] Others have noted that dominant notions of "good" management or "good" forests can have important implications for the way land is used and, in turn, for soil quality (Batterbury and Bebbington 1999; Conte 1999), and these notions appear to have played a supporting role in páramo to pine transitions. However, the determination of whether the goal of "good" management is achieved through this transition, and whether the objective of timber production is compatible with other goals related to plantation establishment—particularly the enhancement of soil quality—depends in part on the biophysical response to afforestation in this system.

Effects of Páramo to Pine Transition on Ecosystem Services

Forest transitions are expected to bring about improvements in soil quality along with improvements in water quality and carbon sequestration (Klooster 2003; Rudel et al. 2005), bringing the systems under transition into more desirable ecological conditions. In the case of plantations in highland Ecuador, the goal of enhanced provision of ecosystem goods and services has been stated in some of the policy initiatives along with the goal of producing timber (Table 1), but little evidence has been available to determine whether these goals have been accomplished. In order to address this question, I draw on data collected in an examination of the biophysical effects of converting páramo to pine that was conducted in Cotopaxi province (for full biophysical results, see Farley and Kelly 2004 and Farley, Kelly, and Hofstede 2004). These results demonstrate significant changes in several key ecosystem processes, including soil carbon storage, soil nitrogen storage, and water retention.

Change in Soil Carbon and Nitrogen Storage

The conversion of grasslands to tree plantations allows for the sequestration of carbon in the aboveground biomass of the trees, and, although the assumption is often made that carbon will be sequestered in the soils as well, there is evidence suggesting that the effect on soil carbon varies from one system to another (Guo and Gifford 2002). In many systems, tree planting can increase the soil carbon stock, in particular when trees are established on degraded soils (Brown, Lugo, and Chapman 1986; Bashkin and Binkley 1998). However, páramo grasslands tend to accumulate extremely large soil carbon stocks due to the large belowground organic matter inputs and slow rates of decomposition typical of these ecosystems, combined with the mineralogical properties of soils derived from volcanic ash (Luteyn 1992; Hofstede 1995). Under these conditions, planting trees can cause a loss of soil carbon, offsetting some of the carbon gained in the aboveground biomass of the planted trees and affecting the overall carbon storage capacity of the system (Hofstede et al. 2002). In Cotopaxi, a loss of soil carbon occurred with plantation age, declining from 5.0 kg m^{-2} under grasslands to 3.5 kg m^{-2} under the 20–25-year-old pine stands in the chronosequence ($p < 0.001$) (Farley, Kelly, and Hofstede 2004). In the absence of pine, these soils would continue to accumulate carbon, a process that likely would not begin to level off for several thousand years (Chadwick et al. 1994), and that is effectively being halted by converting these systems to plantations. These results indicate that planting trees in these grasslands constitutes a partial trade-off of belowground for aboveground carbon. This trade-off has implications for long-term carbon storage, as the carbon stock in these systems is transferred from the belowground pool, where it is more stable over a longer time period, to one where carbon is much more susceptible to losses through fire and to uses that release carbon following harvesting.

Similarly, depletion of nitrogen stocks appears to be occurring in association with the conversion of grasslands to pine. In Cotopaxi, nitrogen became increasingly depleted along the chronosequence at intermediate depth in the soil, while it was gained in the litter and surface soil until the plantations reached 20 years, at which point it declined (Farley and Kelly 2004). This pattern indicates that nitrogen is being redistributed under pine from intermediate depths to the surface through litterfall and throughfall. This type of "nutrient pumping" effect transfers nutrients closer to the soil surface, where they may be more susceptible to loss (Jobbágy and Jackson 2003). The fact that soil nitrogen declined in the upper part of the soil profile in the oldest pine stands indicates that these losses from the surface may be occurring. Combined with the nutrient export that occurs with harvesting, these results suggest that nitrogen availability could become problematic, particularly if multiple rotations of pine are planted on these sites.

Change in Water Retention

In addition to storing large amounts of soil carbon, soils derived from volcanic ash tend to have exceptionally large water retention capacities (Nanzyo, Shoji, and Dahlgren 1993; Wada 1989). As a result of this characteristic, páramo grasslands serve as the primary water catchments for much of the northern Andes and play an important role in providing water for cities, towns, and agriculture in highland Ecuador (Luteyn 1992; Podwojewski 1999). However, some kinds of disturbance can lead to irreversible drying of these soils, causing the loss of water retention, and in some cases water repellency, leading to lower water storage (Podwojewski and Poulenard 2000; Poulenard et al. 2001). Results from tests of soil water retention at Cotopaxi indicate that planting trees in these systems may constitute one such disturbance, as water contents were found to be significantly lower under pine plantations (Figure 3; Farley, Kelly, and Hofstede 2004). Water contents were similar under grassland and pine when the soils were saturated, but under drier conditions water contents in the soils under pine were significantly lower and dropped rapidly (Figure 3; Farley, Kelly, and Hofstede 2004). Because of the role páramo grasslands play in local and regional hydrology, these changes in water retention can have important implications for water management in highland Ecuador.

Transition and Trade-Off

Plantation forestry in Ecuador has provided, or has the potential to provide, economic benefits through timber extraction, the sale of nontimber products from plantations, and the sale of carbon credits. The value of Monterey pine exports on the world market is in the range of \$39–600 per m^3, depending on the type of

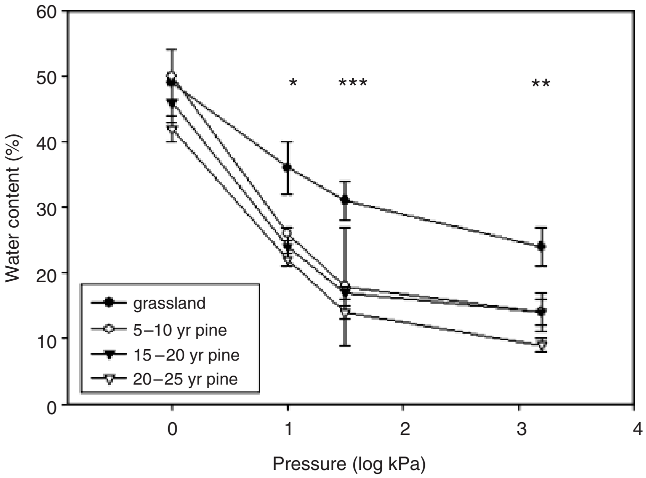

Figure 3. Volumetric water content (mean ± standard error) of A horizon soils at saturation (0 kPa), 10 kPa, field capacity (33 kPa), and wilting point (1,500 kPa). Points farther to the right on the x-axis were subjected to increasing tension to pull water from the soil, simulating increasingly dry conditions. Grassland soils had significantly higher water contents than plantation soils at all pressures measured beyond saturation (***$p<0.001$; **$p<0.01$; *$p<0.05$). *Source:* Farley, Kelly, and Hofstede (2004).

product (from pulp to remanufactured wood). The value of sawnwood, which represents the majority of timber produced from plantations of exotic species in Ecuador, has increased over the past decade (INEFAN 1995; Cerda 1999). In Ecuadorian pine plantations, an additional source of income has been developed from the harvesting and sale of the edible mushroom, *Suillus luteus*, a mycorrhizal fungus that grows in association with Monterey pine (Luzuriaga 1995; Kenny-Jordan et al. 1999). The town of Salinas, which is the largest producer of dried mushrooms in Ecuador, produces 18,000–20,000 kg per year, with approximately 80 percent of the production exported to Europe, and it is anticipated that annual production could reach 50,000 kg if new markets are found (DFC 1998; O. Chamorro, President of *Grupo Juvenil*, Salinas, Bolívar Province, Ecuador, in-person interview, 21 May 2001). Finally, carbon markets represent a relatively new and potentially large source of income that could be derived from plantations. The market for carbon credits generated as part of the CDM has grown rapidly, and afforestation and reforestation are being considered for inclusion in the next phase of the European Union Emissions Trading Scheme (Point Carbon 2006).

In addition to these economic benefits, plantations have become an important fuel source for some communities. The Ecuadorian highlands have been subject to extensive deforestation, leaving fewer than 800,000 ha forested (INEFAN 1995), so that many communities have limited supplies of wood for fuel. As a result, plantations of exotic species (and to a much lesser degree native species) have come to play an important role in the supply of fuelwood. According to Kenny-Jordan et al. (1999), Ecuador consumed more than 2 million m^3 of wood products for fuel in 1996, of which 90 percent came from plantations. In the town of Salinas, for example, prior to the establishment of plantations, bunchgrasses were used as fuel for cooking, but that fuel source has been replaced by pine collected during thinning and pruning of the plantations (E. Punina, Forestry Program, Department of Environment, FUNORSAL/*Fundación de Organizaciones de Salinas*, Salinas, Bolívar Province, Ecuador, in-person interview, 21 May 2001).

Although plantation forestry in páramo grasslands allows for the accrual of these benefits, like other forms of land use change (Mustard et al. 2004) it constitutes a trade-off in which some ecosystem services are diminished. The results from Cotopaxi demonstrate that the change in land use can affect soil properties in páramo grasslands on a decadal time scale, with implications for the provision of ecosystem services such as carbon storage, nutrient storage, maintenance of hydrological cycles, and the generation and maintenance of soils. The effects on other ecosystem services, such as biodiversity, are unclear: species diversity and richness are generally thought to decline with plantation establishment (Armstrong and van Hensbergen 1996) and significant losses of biodiversity have been found following afforestation in some highly diverse ecosystems (e.g., South African fynbos, Richardson and van Wilgen 1986). However, in other locations, plantations have prompted regeneration of native species in the understory (e.g., Geldenhuys 1997; Oberhauser 1997). Few data are available on the effects of plantations on páramo biodiversity, but the existing evidence suggests that they range from an increase in woody species in the understory, to the maintenance of a páramo grassland-type understory, to a loss of all existing páramo plant species (i.e., no understory; Hofstede et al. 2002). The effect is likely to be strongly influenced by the type of management, including whether thinning or pruning are done and whether grazing of the understory occurs (Hofstede et al. 2002). Another important factor is how the sites are treated following harvest, which in páramo plantations often includes burning to eliminate the remaining branches and litter and would eliminate any gains in understory biodiversity that may occur.

Forest transition theory tends to treat the establishment of greater forest cover, whether in the form of secondary forests or plantations, as necessarily leading to improved environmental conditions; however, the results presented here highlight the idea that different landscapes produce different sets of goods and services. Plantations provide greater value in ecosystems goods (such as timber and mushrooms) and the enhancement of some ecosystem services (such as aboveground carbon sequestration), whereas grasslands produce a number of services on which Ecuadorian society has come to depend, including water storage and provision to lower elevations and the storage of soil organic matter (including carbon and other nutrients, which affect site productivity into the future). The new set of biophysical conditions following transition determines the type and degree to which ecosystem services are provided, and includes enhancement of some services and degradation of others.

Conclusion: Forest Transition Theory in the Context of Grassland to Plantation Conversions

Mather (2004) noted that forest transition theory is still in its infancy, and further research will help to clarify the most important mechanisms involved in transitions. This research suggests that the relationship between economic development and forest transition is one that requires further investigation, in particular through additional case studies in developing countries which, as suggested by others (Rudel, Bates, and Machinguiashi 2002; Klooster 2003; Perz and Skole 2003), may follow very different paths than developed countries. Perhaps more critical than the distinction between developing and developed countries, however, is that between reforestation and afforestation. These differences are reflected in the mechanisms that promote increases in forest cover as well as the outcomes of the forest transitions.

In the case of highland Ecuador, forestry policy, economic policy, and global climate policy have all created incentives for plantations to be incorporated as part of strategies to promote economic development. As a result, afforestation there has been associated with land retention or acquisition rather than land abandonment, and can be better characterized as a potential means of economic development rather than a response to or consequence of it. This suggests that, in this case, the economic development path can be used to describe forest transition, but it can be conceived of as a "reverse economic development path" that seeks rather than responds to economic development. The second hypothesized path to forest transition—that of forest scarcity—provides an accurate but partial description of the process of forest transition in highland Ecuador. In a globalized economy, this path also may be conceptualized somewhat differently, with scarcity increasingly viewed in the context of the ability of the forest sector to maintain sources of timber for both internal and external markets. In this sense, the forest scarcity path could be viewed as a component of, or complementary to, the reverse economic development path. These conclusions suggest that both the developing country context and the role of plantations may need to be reconsidered in forest transition theory, with plantations recharacterized as a separate category of transition rather than being grouped as part of a general trend of increasing forest cover.

This distinction between afforestation and reforestation is important not only in terms of the causes of increased forest cover, but also the consequences. In the case of páramo to pine transitions, the validity of some of the assumptions of forest transition theory with respect to the environmental benefits of transition is brought into question. Plantations are qualitatively quite different from natural forests and it has been demonstrated here that, in the case of páramo to pine transitions, they lead to a different biophysical response than those expected within the forest transition framework. It should be noted that, in the examination of páramo to pine at Cotopaxi, the grasslands were not in a highly degraded state and a comparison of pine with degraded grasslands would likely yield different results, making both the type of cover and the condition of the cover prior to transition important determinants of whether environmental benefits will result. However, although not all grasslands have the same soil carbon and water retention functions as páramos, plantations have been found to reduce streamflow, cause soil acidification and salinization, and have some other detrimental effects on soil, water, and biodiversity in a variety of grassland, shrubland, and other ecosystems (Carrere and Lohmann 1996; Farley, Jobbágy, and Jackson 2005; Jackson et al. 2005), suggesting that the assumptions of environmental benefits may be questioned for other locations as well. These results indicate that both the existing land cover prior to transition and the type of forest cover established during transition merit more attention in forest transition theory.

The páramo to pine transition also illustrates that the concept of nature as an agent that shapes human-environmental outcomes (in the sense of Zimmerer 2007a) could be a useful addition to forest transition theory. Páramo grasslands and pine plantations are two different versions of "nature" that lead to very different outcomes in terms of how they can be used by humans, the types of human needs that can be met with them, and how they can contribute to livelihoods. Increasing forest cover has been viewed as a way to improve environmental outcomes, and the perception of "good" forests that dominated forest policy in Ecuador throughout the twentieth century continues to be influential. However, shifting values and perceptions of páramo grasslands have led to an increasing recognition that they are valuable sources of water, soil formation, biodiversity, and other ecosystem services. In order to address the critique that it inadequately distinguishes among transition types and the benefits derived from them (Perz 2007), forest transition theory might benefit from a more explicit inclusion of the agency of nature in contributing to the production of a range of potential human-environmental outcomes.

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Notes

1. Such valuation is by no means universally applauded and has been characterized by some as an attempt to have nature "earn its own right to survive in a world market economy" (McAfee 1999, 134; for criticisms of the ecosystem services approach from an ecology perspective, see McCauley 2006). In the context of this article, the framing of ecosystem processes and functions in these terms is intended as a means to compare the "services" that society obtains from different landscapes, whether those services have actually been ascribed a monetary value or not.
2. The ACOSA plantation was chosen, in part, due to the fact that it included a variety of age classes. However, there were no stands younger than 5 years old or between 10–15 years old, so these age classes could not be included.
3. An estimated 80 percent of activities in the forestry sector are informal, so they are not recorded in official statistics and the role of forestry in the economy is probably substantially underestimated (Ponce 2000).
4. These rates are based on FAO (Food and Agriculture Organization) model estimates of the change in forest cover during these time periods.
5. As an example of this type of view, in their discussion of land reform in the Ecuadorian highlands, Haney and Haney (1989, 74) describe one-third of the land in the province of Chimborazo as "páramo and wasteland" and one-third as pasture, most of which was "unimproved."

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