“Investigating the role and scale of transactions costs of incentive-based programs for provision of environmental services in developing countries.”

DISSERTATION

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Daniel Vicente Ortega-Pacheco

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The Ohio State University

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Dissertation Committee:

Anand Desai, Advisor

Andrew Keeler

Brent Sohngen

Trevor Brown

Abstract

The use of incentives for the provision of environmental services occupies a critically important place in the international development agenda. The use of local approaches for watershed services provision and international efforts for global provision of climate change land-based mitigation services are among promising management options. These two options have the potential to significantly reduce the overall costs of meeting environmental targets through market-based institutional arrangements. Despite widespread agreement that transaction costs are important, existing research has not yet considered the scale and role of transaction costs in determining: 1) the rate of adoption of incentive-based schemes for the provision of watershed services in the developing world, and 2) the supply of mitigation services associated with avoided emissions from deforestation, particularly in a developing country context. To address the former, the first chapter of this dissertation identifies patterns of adoption and the exogenous and endogenous factors that help to explain the number of incentive-based programs adopted during the last decade. Using an econometric model, it suggests that the degree of adoption can be interpreted as diffusion of interdependent induced institutional innovations. The second chapter presents a conceptual framework for transaction costs and reports results of field data collection and empirical estimates of the scale of transaction costs of mitigation through land-based activities in Ecuador taking place under alternative incentive-based institutional arrangements. The third chapter of this dissertation develops an econometric model to estimate the elasticity of land supply in Ecuador and evaluates the effect that transaction costs have on incentive-based mitigation activities. Knowledge generated from this research aims at enriching the scholarly debate on policy diffusion and climate change policy and provides critical insights for policymakers interested in incentive-based institutional arrangements for the provision of environmental services.

Keywords: water, climate change, mitigation, institutions, Payment for environmental services, transaction costs.

Dedication

A mis abuelitas Carmelina y Oferlina y ancestros familiares, por heredarme la más pura y valiosa de las enseñanzas -el amor a la vida y la perseverancia para vivirla.

To my grandmothers Carmelina and Oferlina and family ancestors, for inheriting me with the most pure and valuable lesson - the love to life and the perseverance for living it.

A.M.D.G.

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Vita

1999..Colegio San Francisco Xavier

2003..Licenciatura en Ciencia Agronómicas, Univesidad EARTH

2005..M.Sc. Agricultural Economics, Michigan State University

2005 to present ....Graduate Research Associate, John Glenn School of Public Affairs, The Ohio State University

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Fields of Study

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1.1. Introduction

Water resources conservation and management occupy a critically important place in the international development agenda. Tropical deforestation, advanced technologies for resource use, and population growth all represent serious threats to ecosystems (Dietz *et al.,*, 2003). Alternatives for addressing these rising pressures have received attention from researchers, public administrators and international organizations. The use of local approaches and incentive-based programs are among promising management options. Recent literature has confirmed that local management is an essential strategy for addressing site-specific pressures on natural resources that externally imposed management systems would likely fail to appreciate (see Baland and Platteau, 1996; Agrawal and Gibson, 1999; and Kerr, 2002). International organizations increasingly promote the use of incentive-based approaches for watershed protection and management (World Bank, 2003).

Self-financing Payment for Environmental Services [PES] programming is one market-based approach that integrates the use of economic incentives and is often being consider in local efforts to protect watersheds and watershed services such as water quality and quantity (FAO, 2007; Herrador and Dimas, 2000; Kaplowitz *et al.,* 2008;

Pagiola, 2002). Despite the interest in local PES approaches (e.g., Postel and Thompson, 2005; Ferraro and Kiss, 2002), previous research has not systematically examined their widespread adoption in developing countries during the last decade[i]. In this paper, I examine this phenomenon through the lenses of competing theories of policy diffusion. That is, I focus on the process by which the probability of adoption of an innovation in one location is conditioned by prior adoptions of analogous units elsewhere in a system (Manski, 1993; Berry and Berry, 2007; Braun and Gilardi, 2006).

Improved management of ecosystems, especially those that support highly valuable hydrological services such as drinking water, might be advanced by identifying and understanding the factors influencing the diffusion of local adoptions of PES watershed protection and management programs. Previous research suggests that institutional changes within the broader water sector occur due to the role of both endogenous and exogenous factors acting together to raise the opportunity costs of institutional change, reduce the corresponding transaction costs, and create a pro-reform climate (Saleth and Dinar, 2000). Following this line of work in water institutions (e.g., Saleth and Dinar, 2004 and 2005), this paper empirically investigates the diffusion of local adoptions of watershed-based PES programs. This paper argues that diffusion of PES across developing countries in Latina America, Africa and Asia can be thought of as an interdependent adoption of innovations, which is a process conditioned by identifiable institutional factors.

A significant strand of comparative studies has contributed with important insights into the nature, causes and process mediating international policy diffusion. As valuable as these contributions have been, there remain important questions unanswered that this research will address. First, previous research has mostly focused on the subset of diffusion processes concerning liberal trade and social reforms (e.g., Simmons and Elkins, 2004; Brooks, 2007). In this paper I attempt to extend the applicable domain of diffusion theories by assessing whether it becomes causally relevant for explaining the adoption of market-based environmental programs. Second, diffusion research has identified different exogenous and endogenous mechanisms (i.e., information and competition) that link innovation decisions across space and time (Weyland, 2005). In this study, I focus on domestic factors influencing adoption and draw from institutional economics and transaction costs theory to derive an alternative mechanism to explain diffusion of PES programs. Institutional economics theory proposes that institutional innovation - defined as changes in private property rights and the development of institutions (Ruttan and Hayami, 1984) - is induced by changes in resource and cultural endowments, and technology (Ruttan, 2006). It has been argued that these changes improve the viability of adoptions because they tend to increase demand for institutional innovation while reducing transaction costs (McCann, 2004). In this paper I evaluate PES programs adoption as the diffusion of induced institutional innovation. I examine how the significance of diffusion mechanisms is mediated in domestic adoption choices by (1) characteristics of the innovation itself, namely the transaction costs it entails, and (2) a country's institutional attributes associated with resources, cultural endowments, and technology. I expect that as the financial burdens of protecting water resources for the central government rise, mechanisms of diffusion (i.e., informational, competitive and institutional) should weigh more heavily in adoption choices. I also expect diffusion to be relatively less influential in explaining PES adoption in nations with a low level of resource and cultural endowments and technical-administrative capacity (i.e., institutional factors).

Existing comparative efforts have largely relied on discrete dependent variable models (e.g., event history and hazard analysis) for examining the probability of adoption for policies or administrative innovations across countries (e.g., Jordana and Levi-Faur, 2005; Meseguer, 2004). I develop an *inter*-country model of adoption that differs from previous efforts in that it incorporates a continuous dependent variable for PES adoption. The model is used to explain adoption for a given country as an aggregated and interdependent measure. Specifically, the model accounts for the effect of adoptions in close neighboring jurisdictions on the total number of local PES programs adopted in each country.

This paper proceeds by describing observed patterns of adoption that may help to support the argument that wide spread of PES programs in the developing world during the last decade is governed by a process of diffusion. Next, I discuss competing theoretical models of policy diffusion and derive testable hypothesis for explaining adoption of PES programs. The paper then moves to a statistical evaluation of the degree of adoption of PES and estimates an empirical model to assess the relative explanatory power of competing diffusion theories. Results of this study provide empirical evidence supporting the proposition that adoption of PES programs in the developing world is strongly influenced by diffusion. I then discuss the results in the context of the scholarly debate on policy diffusion. The paper concludes by providing critical insights for public managers and researchers interested in the diffusion of market-based environmental programs for the conservation and development of water resources.

1.2. Market-based Environmental Policies Patterns

PES have been widely implemented as means for promoting conservation and development of hydrological services (Echavarria, 2002; FAO, 2004; Postel and Thompson, 2005; Southgate, Haab, and Rodríguez, 2005). PES approaches can be targeted for the provision of different types of services, individually or as a bundle, (e.g., carbon sequestration, watershed services, biodiversity conservation, and scenic beauty), and implemented at different scales (e.g., national or local levels). Initially, PES efforts were limited to a national level approach that, among other conservation and poverty goals, focused on protecting upstream forests. The poster child case of adoption for a national level scheme is Costa Rica (see Pagiola, Landell-Mills, and Bishop, 2002;

Pagiola, 2006).

Relevant to this study, however, is the increasing adoption of local PES programs for protecting and managing watersheds and their services. Watershed management is an area that lends itself to incentive-based approaches (Kerr, 2002). A watershed may be thought of as a special kind of common pool resource that requires coordinated use of natural resources by all users for its optimal management (Kerr, 2007). Management complexities arise, however, due to multiple, conflicting land uses that may take place within a watershed (e.g., deforestation, intensive agriculture, population settlements). Drinking water[ii] is one service that can be affected by different land uses but can also managed and enjoyed in common by the households within the watershed. In local PES programs for watershed protection, upstream land users (e.g., farmers) are paid or rewarded by downstream water users (e.g., households or firms) for adopting land use practices (e.g., forest land uses) that are likely to limit adverse effects on the quantity and quality of services provided (Wunder, 2006).

In order to capture most of the adopted innovations I use the definition of local PES program proposed by Wunder (2007) and extended by Ferraro (2009). PES program is defined as a coordinating mechanism of voluntary transactions in which an environmental service buyer(s), who does not control the environmental factors of production, pays an environmental service provider(s), who controls the environmental factors of production, for a well-defined environmental service using a cash or in-kind payment that may vary conditionally on the quantity and quality of the environmental service generated. This PES definition is important because it allows for the negotiated agreement to take the form of a contractual arrangement that generally incorporates responsibilities and penalties for the transacting parties, identifies an enforcing authority and suggest alternatives for conflict resolution, which contribute to order by service-buyer interactions, and therefore minimize transaction costs.

Significant diffusion of local, watershed-based PES schemes across countries and regions has occurred around the world in the past decade (See Figure 1.1). These PES schemes range from formally proposed innovations where payments or rewards are not yet flowing to ongoing programs where payment or rewards have begun to flow. For the purpose of this study, both categories are considered as PES adoption.

In Latin America, the PROCUENCAS program in Costa Rica was the first local PES programs and was adopted *circa* 2000. PES program adoption was reported during subsequent years in Ecuador and Colombia (FAO, 2004). By 2004, these countries and their neighbors reported additional implemented programs or under design. Adoption of new programs has continued up to the present. A recent assessment reports about 90 local PES programs in watersheds across the region (Southgate and Wunder, 2009).

Similar patterns have also been reported for Asia and Africa, although in a delayed fashion and in much lower magnitude for the latter. In Asia, Indonesia and the Philippines are the main leaders in the early adoption of local PES programs, followed more recently by India and China. By 2007, 34 programs were reported for the region (Huang *et al.,* 2009). At the same time, two operating and eight formally planned PES programs were reported for Africa. South Africa has been the main innovator in the region followed by Tanzania and Kenya and more recently by Uganda (Ferraro, 2009).

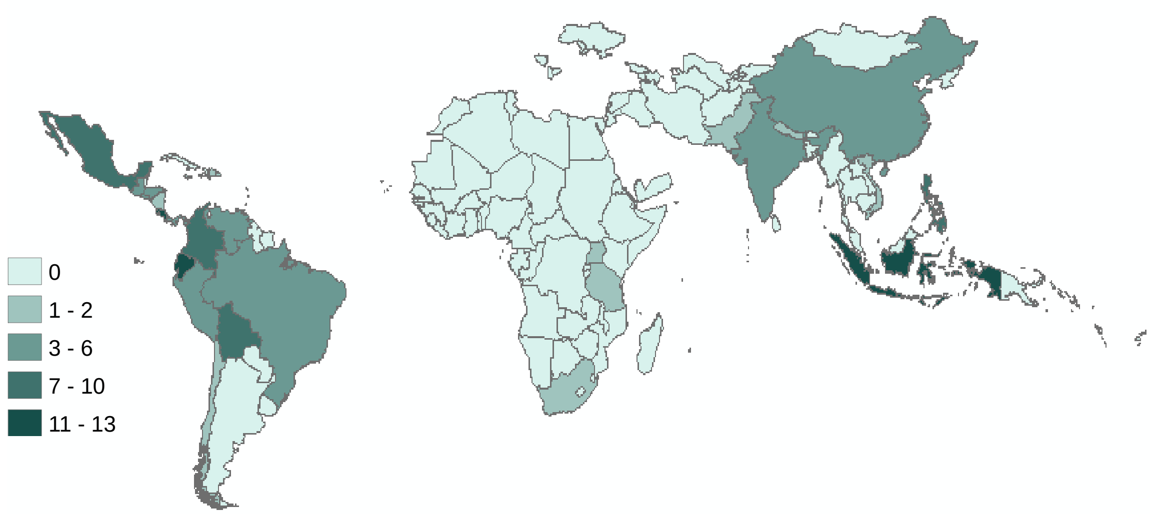

Figure 1.1 Diffusion of local PES programs in the developing world

Figure 1.1 reveals striking geographic patterns of adoption in Latin America and South-East Asia. Given the potential that local PES program seem to offer and the increasing interest in this innovation, the challenge of this study is to explain the observed geographic correlations of a country's degree of adoption and the extent of interdependence among domestic administrative choices across adopters and non-adopters. The concept of diffusion is at the center of this analysis, so I turn next to an examination of the underlying logic of this theoretical construct and the different models explaining its causes and mediating processes.

1.3. Theories of Policy Diffusion

Although most actions by governments are incremental in that they marginally modify existing programs or practices, ultimately every government program can be traced back to some non-incremental innovation (Berry, 1990; and Berry and Berry, 2007). At the heart of diffusion research is the concept of interdependent decision making. In this process, a decision in one location is conditioned to some degree by similar choices made elsewhere in the social system (Brooks, 2007).The logic of diffusion is that adoptions occur relatively infrequently near the beginning of the process. The rate of adoption then increases dramatically but begins to taper off again as the pool of potential adopters becomes small[iii].

Administrative choices are actions undertaken by individuals. Local managers may or may not choose to adopt a local PES program within each country provided that the prevalent legal framework does not prohibit it. Therefore, I continue by examining the principal actors involve in the decision to adopt PES programs and their motives in the decision-making process. In this paper, local watershed or water system administrators have the final responsibility for identifying and analyzing alternative mechanisms by which they could address conservation and its necessary funding needs. The technical capacity of water resources administrators varies and is conditioned by social context[iv]. In this view, local managers can evaluate available alternative innovations in a cost-benefit fashion wherein their potential to address pressing problems and to cope with institutional regularities are weighted against potential negative consequences of adoption. Whether and how such judgments are conditioned by analogous decisions in neighboring nations is central to understanding the channels and importance of diffusion in domestic decision-making processes. As much of the literature has a pronounced emphasis on both learning and competition as the basis for assuming that diffusion channels are regional in nature, in the next section, I discuss these concepts and point the reader to more detailed literature. Our focus is on describing an alternative domestic institutional channel for diffusion that then is used as building block for the empirical analysis.

1.3.1 Information

The core of comparative diffusion literature rests upon the idea that potential adopters of an innovation draw lessons from previous experiences (i.e., sources) from which information is credible and relevant to in-situ characteristics of the problem at hand. In this view, resource-constrained decision-makers that are uncertain about an innovation will examine the outcome of previous adopters in order to gain information about its likely effects. As the number of previous adoptions of an innovation rises, information becomes more readily available, reducing uncertainty and increasing the likelihood of subsequent adoptions of the innovation (Jordana and Levi-Faur, 2005; Simmons and Elkins, 2004) by changing total transaction costs[v]. Further, public administrators are unlikely to give equal consideration to all prior innovation experiences. Instead, informational sources have been shown to be jurisdictions that are geographically proximate and sharing political, economic and social linkages (Berry and Berry, 1992; Mintrom and Vergari, 1998). Relevant informational sources may be also defined by cultural ties and shared status (Simmons and Elkins, 2004).

1.3.2. Competition

Innovations can be adopted not only for their direct ends, but also for competitive signals that such decisions transmit about the decision-maker (e.g., environmental-friendliness and support for governance and decentralization). Competitive rewards such as improved status or material gains associated with the adoption of an innovation can introduce incentives for its diffusion (Berry and Berry, 1992; Gray, 1973; Mintrom, 1997; Simmons and Elkins, 2004; Walker, 1969). This is essentially because one country, community, or local manager's standing is conditioned by that of others in similar social units. For instance, competitive benefits at the local level can arise from an innovation's effect on reputation or status and the resulting ability to capture shares of central's government funds to support provision of services or support from governmental or non-governmental international organizations[vi].

Moreover, competitive considerations can also influence the decision of local administrators whether they are appointed or elected. The theory of public choice suggests that the aim of public employees is not only to fulfill the wishes of the community. They also look after their private interests. Appointed employees can gain stability by raising fees higher than is necessary to finance the optimal level of public services because the power and status of a public employee is positively correlated with the size of their budget (Brennan and Buchanan, 1980).Conversely, the choice of adoption in the case of elected employees could be influenced by both the level of demand from the community for such administrative change and opportunities for political rent-seeking. It has been argue that when demand for change is high, the politician would likely rush to adopt the innovation in order to avoid negative consequences in future elections (Besley and Case, 1995)[vii].

Comparative studies have noted that uncertainty about the benefits and costs of new programs is likely to vary across types of innovations (e.g., Brooks, 2007), geographically defined areas (e.g., Berry and Baybeck, 2005) and with respect to who makes decision and at what level of government. As uncertainty and information drawn from prior experiences domestically and abroad affects adoption decisions by changing total transaction costs, the importance of diffusion positions itself in an induced institutional innovation process.

The next steps, then, are to examine how and when varying degree of perceived transaction costs affect adoption decisions, and to examine how the characteristics of the innovation shape institutional concerns that influence adoption.

1.3.3. Induced Institutional Innovation

Institutions structure incentives, shape people's beliefs and preferences, and introduce predictability to human interaction (Schmid, 2004). A broad definition of institutions includes both organizations and the rules of society that govern behavior. As such, human institutions (i.e., ways of organizing activities) may affect the resilience of the environment and environmental services (Dietz *et al.,* 2003) such as biodiversity (Barbier *et al.*, 1994).The literature suggests that cultural endowments (e.g., social norms) can also prevent moral hazard and adverse selection from imposing excessive costs on society (Ostrom, 2007).

Local payments for the provision of environmental services are an example of an institutional arrangement that may help achieve water source protection goals by reducing transaction costs (Kaplowitz *et al.,* 2008). Institutional innovation - defined as changes in private property rights and as the development of institutions –can be induced by changes in resource and cultural endowments, and technical changes (Ruttan, 2006).

As different endowment and technology change, transaction costs can be reduced and social demand for institutional innovation increased, which may enhance the viability of adoption (McCann, 2004)[viii].This latent demand is translated into actual demand via the functioning of the politico-bureaucratic system (Grabowski, 1995). Institutional changes can then be endogenous to the system and evolutionary in that, over time, institutions develop to satisfy increasing demand while minimizing transaction costs[ix].This view suggests that the key to diffusion of innovations lies in the nature of preexisting institutions reflecting the accumulated level of endowments and technology, and correspondingly de-emphasizes the importance of exogenous competitive or informational forces. Preexisting culture and ideology may make institutional innovation more or less costly (Hayami and Ruttan, 1995). Institutional innovations are dependent upon enforcement costs (Grabowski, 1995), which are also subject to preexisting culture, ideology, and social structure.

Institutional economic theorists (e.g., Coase, 1961; Williamson, 1979) have argued that organization and institutions are the result of transaction costs minimization. In this context, institutional innovation can be understood as a strategy to economize both on relative scarce inputs available to the decision-maker but, ultimately on transaction costs affecting the coordination of human interaction for the protection of ecosystems and their services. If so, the degree of transaction costs can play an important role in conditioning adoption of institutional innovations.

Moreover, it could be argued that the supply of institutional innovations can be viewed as depending on the cost of achieving social consensus which may be reduced by advances in cultural endowments. This proposition is consistent with the notion that mental models and ideology are the basis for humans to construct explanations in the face of ambiguity and uncertainty, and to enable action upon them. Mental models[x] are the internal representations necessary to interpret the environment whereas the institutions are the external constructs individuals create to structure and order such environment. Institutions arise in this context to narrow a gap between an agent's competence and the difficulty of the decision problem to be solved.

The next section offers two hypotheses derived from these theoretical propositions. The first posits that as the magnitude of transaction costs of adopting an innovation rise, the informational and competitive mechanisms of diffusion will more powerfully shape domestic policy choices in cases where resource and cultural endowments and technology are not in and of themselves enough to bring about institutional change. The second hypothesis suggests that institutional diffusion forces should be more influential among developing countries with higher levels of resource and cultural endowments and technology. Whereas the first hypothesis conditions diffusion effects to specific levels of domestic institutional factors, the second narrows the domain of interdependent effects even further among potential adopters of an innovation. In this sense, I posit that the increase adoption over time and space of PES programs can be interpreted as diffusion of induced institutional innovations.

1.4 When and Where is Adoption of Institutional Innovations Conditioned by Transaction Costs?

Previous studies argued that transaction cost economizing can help explain decision-making of agents as they pertain to public organization (e.g., Maser, 1988; Williamson, 1979; 1981; 1997). Conceptual elements of transactions costs theory have been employed to address issues in the public sector such as those associated to the policy process (Bryson, 1984), implementation (Calista, 1987a; Calista, 1987b), government regulation (Heckathorn and Maser, 1987) and for explaining changes in formal rules produced by political organizations (Eggertson, 1996). More recently, literature has used transaction costs theory to empirically explain government's decision about public service provision (Brown and Potoski, 2003; Levin and Tadelis, 2007), and as a framework to policy intervention (Bryson and Ring, 1990). These previous studies inform our research as they explore whether, or in what degree, transaction cost economics is pertinent to public choices about PES programs.

The magnitude of transaction costs can affect induced institutional innovation. Innovation is a costly and risky endeavor, not just financially, but in political terms as well (Rose-Ackerman, 1980). The nature and magnitude of such risks are likely to vary systematically across types of innovations with the cost and ease of reversing the decision

(Brooks, 2007). For this paper, transaction costs can be defined as the expenses of organizing a market or implementing a government policy (Gordon, 1994). Transaction costs are costs borne by the government for program implementation and administration (see Falconer *et al.,* 2001). Transaction costs of decisions to adopt an innovation encompass both the financial expense associated with implementing the program, the disruption in current administrative practices, and the outlay of political capital to build public support for the new policy as well as the uncertain consequences of such actions. The variation in the relative shares of these transaction costs is partly related to the nature of the program, but also reflects the stage of its adoption and overall institutional framework in which they take place (Dorward, 2001). Examples of types of transaction costs include search and information costs, bargaining and political decision costs, and monitoring and enforcement costs (Dahlman, 1979).

Transaction costs and the uncertainty of consequences of implementing an innovation can be critical in the decision to adopt new policies and programs (Brooks, 2007). Once committed, relational specific investments associated to program implementation are fixed because the costs of reversing or undoing such decision are exceedingly high as they can potentially alter the next best policy choice (e.g., opportunity cost). All of these add to the transaction costs associated with administrative choices to implement contract-based public service provision (Brown and Potoski, 2003).

Decisions about the form of administrative organization and contracting for the provision of ecosystem services can be thought as programs strategies for coping with transaction costs. PES programs that protect ecosystems and provide hydrological services are infeasible without a coordinating program. This is because making conditional payments becomes expensive and limited when contract negotiating costs are high and the transmission of information is costly. Therefore, local water resources managers may have a strong incentive to reduce transaction costs through the adoption of PES programs for provision of hydrological services. In next section, a number of institutional innovations are described which would tend to reduce transaction costs and consequently increase the likelihood of PES adoption.

1.5. The Case of Local Payment for Environmental Services Programs

There has been global experimentation with local PES schemes in watersheds for almost a decade[xi]. The key elements that determine whether the provision of the service may lend itself to a local PES approach are the characteristics of the service, institutions and the individuals involved in these transactions. Characteristics of the ecosystem service can be observed in physical and economic dimensions. Physically, the production function of watershed services such as drinking water can be derived to some extent from the costs of monitoring and enforcing given levels of quantity and quality. Economically, drinking water can be thought as a rival and excludable good. Moreover, the service itself may be costly to observe and thus the payment will have to be tied to observable variables that are correlated and conditional with the quality and quantity of the desired service (e.g., paying landowners to create riparian buffers that reduce runoff into nearby surface waters).

Adoption of direct watershed-based PES programs enables transacting parties to capture gains from the exercise of property rights in ways that were not previously feasible. PES programs are one institutional arrangement by which contractual transactions over the provision of watershed services in exchange of conditional payments is coordinated. It is conditional because service providers must demonstrate continuous fulfillment of specific requirements such as quantity or quantity of the services.

In the absence of PES programs, coordination costs for reaching agreement between interesting parties can be prohibitively high. Eggertsson (1996) indicates that administrative choice can play an important role in reducing transaction costs involved in voluntary exchange by providing clear and stable property rights. The adoption of local PES programs allocates power and property rights to service beneficiaries and providers. The decision to adopt a PES is treated in this analysis as a choice dependent on different levels of endowments and technology that enable local managers to identify potential benefit from implementing a PES programs and to assess its feasibility. Local managers have a strong incentive to adopt a PES program if perceived transaction costs are low. The magnitude of transaction costs will depend on a number of factors, some of which relate to the institutional environment and norms, and some of which are specific to information associated to the services under consideration available from prior adoptions of the innovation or from external sources. The key factor is whether or not, and to what extent, information about the provision for watershed services is made readily available to suppliers and administrators.

PES programs are regularly designed so that local managers receive voluntary applications from landowners interested in becoming service providers and accept them into an administrative process under most circumstances. This is a condition for the right to negotiate regarding service contracts. In the case of agreements reached between PES administration and individual landowner, direct compensation is usually provided in exchange on pre-established services characteristics for a given period. Contracts also specify possibility for renewal, penalties in case of repetitive noncompliance and mechanism for conflict resolution. The PES program can thus be viewed as an alternative coordinating mechanism with many benefits including relatively lower cost burn by the participants in addition to the originally motivated outcomes of watershed protection and hydrological services provision.

Another task of the PES program is to provide information on the service provision process to stakeholders and the community, reducing transaction costs of information acquisition by the parties involved. One example is the development of model contractual agreements which lower the information costs involved with drawing up agreements. In this sense, non-for-profit and non-governmental organizations (NGO's) can play a critical role. Local staff may not have sufficient management capacity or resources. NGOs supplement these needs by holding low-cost or free seminars to explain procedures to stakeholders, conducting negotiation training or providing ad-hoc technical consulting services during program start-up. USDA and ICRISAT, among others, have organized various regional and international meetings to share what has been learned over time about the adoption of PES programs and alternative ways to improve adoption and performance (Kerr and Jindahl, 2008). Regional networks have emerged from these working sessions; RISAS and RUPES are two examples in Latin America and Asia, respectively.

A variety of factors increasing transaction costs may affect the decision to adopt local PES program for watershed services. In particular, I have noted that transaction costs can mediate the effect of both information and competitive concerns differently across countries. Formal institutions (i.e., laws) that limit overlapping and conflicting claims, place time limits on negotiation, allow regional agreements, improve coordination, and generally facilitate negotiation, would tend to decrease transaction costs. In most cases a contractual arrangement means little unless a formal or informal enforcement procedure and an enforcement authority are in place, otherwise negotiated agreements would tend to require higher enforcement costs. Therefore, in countries where legal frameworks are weak one would expect that a low number of PES would have been adopted as transaction costs would be higher. This essentially follows the notion that a local manager is likely to adopt if he or she has perceived costs of implementation and operation that are low relative to expected gains[xii].

Likewise, one also would expect adoption of PES programs to be lower in countries where the support of NGO for PES implementation is weaker. In countries where investments in water resources protection and management are lower, however, one would expect larger perceived benefits from implementing a PES programs. Perceived benefits could be associated with potential outcome and satisfaction of social claims, which may justify the decision to adopt[xiii].

Adoption of PES program can also reduce transaction costs by altering information sets. One example is the case of reductions in information asymmetries for contracting. Transaction cost economics assumes that economic agents exhibit opportunism (i.e., self-interest seeking with guile) (Williamson, 1985). Bromley and Cummings (1995) suggest that contracting parties will lower transaction costs in repeated negotiations since there is a perceived probability of reducing opportunism as information increases. This literature suggests that repeated interactions may promote the development of norms of trust, and that face to face interaction promotes cooperation. Consequently, one could expect that as more PES program are adopted within a country and subsequent contracts negotiated, available information could increase and positively affect the likelihood of adoption. In sum, the effect of transaction costs is expected to influence the number of local PES program adopted[xiv].

These predictions are tested by using cross-sectional data from 120 developing nations to explain how diffusion drives local PES program adoption. A list of the countries used in the analysis is included in Appendix A.

1.6. Empirical Model and Data

The econometric model estimates the causes of PES programs adoption at the national level. Diffusion models have been developed by various authors, mostly in the context of considering adoption of liberal trade and social reforms across time and space (see Simmons and Elkins, 2004; Brooks, 2007). Previous theoretical research on water institutions suggests that institutional changes within the broader water sector occur due to the role of both endogenous factors (e.g., water scarcity, performance deterioration, and financial non-viability) as well as exogenous factors (e.g., macro economic crisis, political reform, natural calamities, and technological progress) (Saleth and Dinar, 2000). Following this earlier line of work, this paper estimates a diffusion model that controls for three major types of diffusion mechanisms: competition, learning, and geographical proximity. Earlier econometric studies of diffusion have largely relied on discrete dependent variable models (e.g., event history and hazard analysis), whereas this study uses a continuous dependent variable and estimates a cross-sectional diffusion model of induced institutional innovations that predicts the magnitude of adoption of PES programs. The magnitude of adoption of PES programs in each country is expressed as a linear function of different endowments and technology, as well as political, economic and institutional factors likely to affect transaction costs. Following Simmons and Elkins (2004), the functional form for the spatially dependent diffusion can be expressed as a spatial lag model[xv].The model can be expressed as

$$Y = \rho Wy + X\beta + \varepsilon, \tag{1}$$

where ρ is a spatial autoregressive coefficient, W is the n × n spatial weights matrix, X is a vector of non-diffusion regressors with coefficients β, and ε is a vector of error terms. The left hand side of eq. 1 is the accumulated number of local PES programs adopted over time in each country interpreted as an induced institutional innovation[xvi].

Our model specification allows us to account for the influence of accumulated adoption in country i on country j transmitted along a region or neighbor. The spatial lag is the weighted average of the dependent variable in the actor's "neighborhood." The neighborhood is mapped by W. Thus, the spatial lag for country i can be written as

$$Wy_i = \sum_{j=1,\ldots,N} W_{ij} \cdot y_j, \tag{2}$$

where W is the spatial weights matrix and y_j is the dependent variable for country j . In matrix form, this relationship can be written as W_y, where y is an N × 1 vector of observations on the dependent variable.

$$W_{ij} = 1/d_{ij}^2, \tag{3}$$

In eq. (3) W_i is a $N \times N$ matrix whose off-diagonal elements are 0 and the diagonal elements are the weights of all observations relative to i, (as opposed to implicit equal weights in ordinary least square). This specification of W allows for a gradual rate of decay in the lag by weighting the dependent variable by the distance to all other observations in the sample (Anselin, 2001). Thus, W is row standardized, distance-weighted average of the neighboring i values. Where, d_{ij}^2 is the distance in Km between the centroids of countries i and j ($w_{ii} = 0$). As noted by Fotheringham, Brunsdon, and Charlton (2002) this distance d can be determined by selecting the model with the lowest Akaike Information Criterion (AIC) value.

Our model specification allows for transaction costs to decrease as the number of adoptions reached increases because of learning that occurs over time and space. Even with high transaction costs, the model allows adoptions to occur where potential large gains from adoption are perceived by local managers. The model can be used to project future adoption by changing the vector X. For example, future changes in legislatures improving enforcement (i.e., quality of institutions) can be projected, and used to predict the number of PES program adopted by different countries and regions.

Table 1.1 and Table 1.2 present a list of definitions and sources for each variable included in the empirical model. Otherwise indicated, variables values are obtained for year 2000.

Variable	Description	Mean	Standard Deviation
Dependent			
PES adoption	Number of local PES implemented in each country during the 2000-2007 period	1.008	2.643
Independent			
Policy index	Quality of policymaking: created based on all of the variables utilized for the indexes presented in Table 1.2.	1.287	0.722
Institutional Indexes	As presented in Table 1.2, Institutional quality include variables for		
	Party institutionalization	1.105	0.759
	Judicial independence	1.198	0.911
	Bureaucracy quality	1.049	0.790
Controls			
Legal Origin	Dummy =1 Common law tradition.	0.210	0.409
	Dummy =1 Civil law tradition.	0.452	0.500
Environmental Expenditures	Defined as the share of government budget allocated to public investments in each country. It is measured as the average expenditures as share of GDP in 2000. Expenditures on public health are used as proxy.	2.719	1.708
Development Flows	Defined as a proxy for green international pressures. It is measured as the gross official development assistance flows in millions USD$ received by each nation as share of total government expenditure.	16.071	33.552
NGO Support	Defined as the principal source of information access and green capacity building. It is measured as the number of NGO working in each country	827.895	658.442
World Bank	Defined as the principal instrument of policy transfer by which implicit or explicit conditionality attached to development loans and assistance. It is measured as the level of World Bank loans and credits in million USD$ weighted by total area of each country in thousand hectares.	0.036	0.138
Country wealth	Defined as overall proxy of country wealth. Measured as per capita gross domestic product.	1424.371	1664.194
Water withdrawal	Defined as demand for water resources. Measured as annual per capita water withdrawal in cubic meters at year 2000.	478.395	687.876
Forest cover change 1990-2000	Defined as environmental threat to natural resources proxy for pressures on improved resources management. Measured as average percentage change.	-3.006	13.356

Table 1.1 Variables description and summary statistics

Variable	Description
Institutional Indexes	Institutional quality
Judicial Independence	- *GCR jud:* The judiciary in the country is independent and not subject to interference by the government and/or parties to disputes. - *BTI jud:* Does an independent judiciary exist? - *Fraser:* Rating of independence of judiciary.
Bureaucracy Quality	- *CUSCS merit:* the degree to which effective guarantees of professionalism in the civil service are in place and the degree to which civil servants are effectively protected from arbitrariness, politicization, and rent-seeking. Average of all years for which data are available. - *CUSCS perform:* the degree to which the bureaucracy has salary compensation systems and systems for evaluating the performance of public officials. Average of all years for which data are available. - *CUSCS efficiency:* the degree to which the bureaucracy is efficient in assigning human capital, given a fiscal policy constraint. Average of all years for which data are available. - *ICRG strength:* High points are given to countries where the bureaucracy has the strength and expertise to govern without drastic changes in policy or interruptions in government services.
Party Institutionalization	- *BTI stable:* To what extent is there a stable, moderate and socially rooted party system to articulate and aggregate societal interests? - *Confidence:* How much confidence do you have in the Political Parties? - *Vote volatility:* Volatility is calculated by subtracting the percentage of the vote/seats won by every party in an election from that won in the previous election, taking the absolute value of this result, summing the results for all parties, and then dividing this total by two. - *PI elect:* To what extent are political leaders determined by general, free and fair elections? - *DPI party age:* The average of the ages of the 1st government party, 2nd government party, and 1st opposition party, or the subset of these for which age of party is known.

Table 1.2 Description of Institutional and Policy Indexes.

“continued”

Table 1.2 continued

Variable	Description
Policy index	Quality of policymaking. Constructed as a unified measure of:
Stability	- *Fraser*: Standard deviation of detrended Fraser Index of Economic Freedom (quadratic trend) (1999-2004). - GCR Plan: measures whether legal or political changes over the past five years have (1=severely undermined your firm's planning capacity, 7=had no effect) - GCR Commit: Measures whether new governments honor the contractual commitments and obligations of previous regimes. - PI: Experts evaluation of the "Consistency and continuity of government action in economic matters".
Adaptability	- *BTI flexibility:* the ability of the political leadership to act flexibly, political leaders capability for learning, and whether political leaders can replace failing measures with innovative policy. - *CUSCS 29:* experts (from academia, government and the media) rate the states' ability to respond effectively to domestic economic problems. - PI capacity: employers and experts evaluate the decision-making capacity of the political authorities in economic matters (responsibility, rapidity, etc.) - CUSCS 21: rating the states' ability to formulate and implement national policy initiatives.
Coordination and Coherence	- CUSCS 18: rating the effectiveness of coordination between the central government and local-level government organizations. - PI coordination: rating co-ordination between ministries and within administrations.
Implementation and Enforcement	- GCR wage: Expert evaluation of whether the minimum wage set by law in the country is (1=never enforced, 7=strongly enforced) - GCR tax: Expert evaluation of whether tax evasion in the country is (1=rampant, 7=minimal) - GCR environmental: Expert evaluation of whether environmental regulation in the country is (1=not enforced or enforced erratically, 7=enforced consistently and fairly) - BTI imp: Analysts' estimate of the following question: Does the government implement its reform policy effectively? - CUSCS 21: rating states' ability to formulate and implement national policy initiatives.
Efficiency	- CUSCS 22: rating states' effectiveness at collecting taxes or other forms of government revenue. - GCR spending: Expert rating of the composition of public spending and whether it is wasteful. - *BTI eff:* Expert evaluation of whether the government makes efficient use of available economic and human resources. - EIU eff: Expert assessment of the effectiveness of the political system in formulating and executing policy.
Public-Regardedness	- GCR favor: Expert rating of whether when deciding upon policies and contracts, government officials (1=usually favor well-connected firms and individuals, 7=are neutral among firms and individuals) - GCR poor: Expert evaluation of whether government social transfers go primarily to (1=poor people, 7=rich people). - Corruption: Perceptions of the degree of corruption as seen by business people and country analysts ranging between 10 (highly clean) and 0 (highly corrupt).

Note: Modified from Berkman et al. (2008).

1.6.1 Dependent Variable

I defined this as a continuous variable representing the country's internal degree of adoption (i.e., aggregated measure of number of PES programs adopted through 2007). Adoption of PES schemes range from formally proposed innovations where payments or rewards are not yet flowing to ongoing innovations where payment or rewards have begun to flow. This dependent variable captures adopted innovations across Latin America, Africa and Asia.

1.6.2. Independent Variables

Domestic adoptions of institutional innovations such as PES programs are hypothesized to be governed by the logic of transaction costs economizing, wherein local managers seek alternatives for achieving service provision while minimizing costs associated to establishing, enforcing and monitoring contracts and perceived risks of such decision affecting the stability of their posts. The principal independent variables of this analysis are, therefore, indices[xvii] that reflect the magnitude of transactions costs. The effect of transaction costs affecting the adoption decision is captured in the empirical model through the overall quality of institutions (including legislatures affecting enforcing) in a given country. The stronger the overall national institutional framework, the lower transaction costs required for local adoptions. Policy and Institutional Indexes are drawn from an international dataset recently developed by the Research Department at the Inter American Development Bank (Berkman *et al.,* 2008).

1.6.3. Control Variables

Additional variables are included to control for two alternative explanations for the observed patterns of adoption. The first, public expenditures on environmental protection can be thought as a major source of pressure on adoption of self-financing institutional innovations such as PES programs. Lower expenditures allocated to protection of ecosystems from already limited governments accounts mean that local water resource management must be financed in-situ, raising the incentives of greater degree of aggregated country adoption. Previous research points to international forces as the greatest source of incentives for adopting market-based PES programs (Wunder, 2007). In this view, the decrease of international green aid constrains governments' ability to finance resource protection, while strengthening the need to implement alternative means to achieve service provision sustainability. To control for these pressures, I include the share of government budget allocated to public health investments in each country as a proxy for green public investments (Environmental Expenditures)[xviii]. I also include a proxy for international environmental pressures, measured as gross official environmental-based aid flows to each nation (Green Flows).

Institutional innovation may be transferred within and across nations from a common source, such as by international financial institutions or NGOs. International institutions have long been shown to lower the cost of acquiring knowledge about policy innovations abroad and to provide direct financial support for adoption (Dolowitz and Marsh, 2000). In the area of incentive-based policy instruments for environmental protection, the World Bank has been at the forefront of active dissemination of models such as PES programs (World Bank, 2003) and thus is a likely source for the transfer of these innovations. For some scholars, the principal instrument of policy transfer is through the implicit or explicit conditionality attached to development loans and assistance (e.g., Huber and Stephens, 2000). In this view, as the value of World Bank development loans to a given country increases, the likelihood and degree of PES programs adoption should also increase.

Although financial green aid may have decreased as government or multilateral-lead international development assistance is shifting to fulfill poverty alleviation goals, in recent years non-public organizations (NGOs) have increased their environmental activities, which could have a positive implication in adoption decision. The international donor community has been shifting from environmental protection towards poverty alleviation strategies (Wunder, 2006). During the 1990s, the bilateral and multilateral agencies' forest-sector funding dropped by approximately 25% and 60%, respectively (Molnar *et al.,* 2003).As noted earlier in this paper, NGO tend to support local efforts with information and capacity building. To control for these effects, I include a measure of the level of World Bank loans and credits for water resources protection and management (World Bank) and the number of non-public organizations working in each nation (NGO support).

Similarly, the adoption of PES programs seems to respond to features of country wealth, legal origin, water resources demand and perceived environmental threats. Country wealth is measured as per capita gross domestic product (GDP per capita).Water demand is measured as cubic meter consumed per person per year. Perceived environmental threats are likely to result in increase demand for institutional innovations. Specific linkages between changing land uses and water quality have been reported (Bruijnzeel, 2004). Kosoy *et al.,* (2007) suggest that Central American rural populations widely perceive water provision to be a primary forest benefit, and high quality and quantities of water to be a function of large forest cover upstream. This argument is consistent with empirical findings of Máñez Costa & Zeller (2005), Johnson & Baltodano (2004) and Ortega-Pacheco *et al.,* (2009) for Guatemala, Salvador and Costa Rica, respectively. Following this previous line of work, I use the rate of forest cover change as a proxy for environmental forcing on water resources likely to affect demand for institutional innovations. Further, to account for interrelated criticisms in the comparative politics literature which suggest that legal origin is one of the main determinants of actual policymaking & institutions today (and economic outcomes) (e.g., Posner, 1973; North and Weingast; 1989, Acemoglu *et al.*, 2001; Beck *et al.,* 2000) I control for the basic institutional setting at the time of colonization (Legal Origin).

1.7. Results and Discussion

A spatial lag model was estimated using cross-sectional observations for 124 countries to explain country's internal degree of adoption of PES schemes. Coefficients, test statistics, robust standard errors for the aggregated measure of adoption of PES programs are reported in Table 1.2. Overall, the Lagrange multiplier (LM) test of the model indicates that spatial dependence is significant ($p < 0.000$).This latter test provides evidence that a country's aggregated measured of adoption of local PES program is sensitive to adoptions elsewhere in the system. This spatial interdependence, however, only partially explain the problem as the weight matrix only incorporates a distance measure and does not accounts for information of interdependence (or the lack thereof) of other factors such as neighboring boundaries, political backgrounds, etc.

Some key variables in the model were significant and of the expected sign. For instance, as expected, Institutional Indexes (i.e., Party Institutionalization) had a positive effect ($p = 0.030$). The measure of NGO Support also had positive effect ($p < 0.000$) on estimated country's interdependent adoption of local PES programs. These two estimates suggest that institutional, competitive, and informational factors play a role on the observed interdependent adoption of PES programs. Specifically, the results indicate that the higher the overall quality of institutions and strength of the overall national institutional framework, and the number of non-public organizations working in each nation, the higher the magnitude of country's internal degree of adoption of PES schemes.

Rate of forest cover change for the 1990-2000 period and the allegedly associated beliefs regarding the potential for water sources to be threatened had a significant negative effect ($p = 0.009$) on interdependent PES adoption. Likewise, having a legal system in the common law tradition had a significant negative effect ($p=0.039$). Surprisingly, the direction of the effect for the former variables is not as expected. This effect means that, on average, countries in which the deforestation rate was higher have adopted a lower number of local PES programs.

Accounting for scale effects and spatial non-stationarity can help explain the observed effect of deforestation which seems to be at odds with our prior expectations. Scale effect, that is, the influence of scale on the outputs of a model (e.g., strength of the relationship, parameter values and direction, prediction accuracy) can be a consequence of the relationship between the dependent and independent variables varying in space (Foody, 2004). Non-stationarity means that the relationship between variables under study can vary from one location to another depending on socio-economic, bio-physical or institutional factors, which can also be spatially correlated. In fact, scale-dependent results can be expected with a change in the spatial specificity if a relationship is spatially non-stationary (Openshaw, 1984). The major empirical problem from ignoring scale effects and non-stationarity is that predictions based on parameters derived from global non-spatial models may be biased and misrepresent country specific effects.

	Coefficient	Robust Standard Error	P-value
Dependent Variable			
PES adoption			
Independent Variables			
Policy Index	-0.619	0.547	0.258
Institutional Indexes			
Party Institutionalization	0.796	0.367	0.030++
Judicial Independence	0.141	0.417	0.735
Bureaucracy Quality	-0.345	0.552	0.532
Control Variables			
Legal Origin (Common Law)	-1.191	0.578	0.039+
Legal Origin (Civil Law)	0.653	0.443	0.141
Environmental Expenditure	0.138	0.125	0.268
Development Flows	-0.008	0.005	0.072
NGO Support	0.002	0.000	0.000+++
World Bank	0.087	0.722	0.904
Country Wealth	0.000	0.000	0.400
Water withdrawal	0.000	0.000	0.213
Forest cover change 1990-2000	-0.040	0.015	0.009++
Constant	-1.079	0.589	0.067
N	124		
Log Likelihood	-258.949		
Wald test			0.001
Lagrange multiplier (LM) test			0.000

Table 1.3 Model results, tests and summary statistics

Notes: A +++, ++, or + indicates that the parameter is significantly different from zero at 1%, 5%, or 10% levels. Fixed bandwidth 43

In this context, the negative effect obtained for deforestation from the estimation of the spatial lag model is an estimated global effect. That is, the effect of explanatory variables is evaluated in a fixed form at the mean value. By doing so, the effects of variables that have large variability across countries are not being fully captured. For instance, the rate of forest cover change can be substantially different between two countries. In a country with high rate of deforestation it is likely that the effect could be positive, whereas a country with low rates could have negative rates. However, the average effect denoted by our model only reflects the estimated effect evaluated at the mean rates of deforestation for our entire sample. This is particularly problematic given that in Table 1.1 it can be noted that the mean value of deforestation rate is negative (-3.006).

Similar issues are likely to arise due to the selected modeling scale. Note that I have estimated a country-level model of adoption of local PES programs. This decision rested on the assumption that an aggregated measure of total number of local PES programs adopted at each country allows me to capture on average the heterogeneity of forces likely to influence adoption across localities within each country. However, this may not be necessarily the case. Unfortunately, lack of access to reliable watershed level statistics or information about political, economic and institutional factor for each local jurisdiction at each country prevents a more disaggregated analysis.

1.7.1 Spatial variability analysis

The preceding discussion shows that scaling down and dealing with spatial non-stationarity are critical remaining tasks for understanding the widespread adoption of local PES programs. Locally representative regression coefficients are desirable for micro-level policy decisions (Ali, Partridge, and Olfert, 2007). Spatial explicit approaches can provide a solution to problems of non-stationarity while increasing accuracy and prediction power in diffusion modeling. Geographically weighted regression (GWR) is one such approach proposed to address the problem of non-stationarity and estimate the regression model parameters varying in space (Fotheringham *et al.,* 2002; Calvo and Escolar, 2003). The GWR functional form allows us to identify spatial patterns that global non-spatial (i.e., OLS) and spatial error and lag models (SEM and SLM) are not able to capture. This is because the GWR model allows the researchers to estimates the effect on the dependent variable resulting not only from variations in the geographical characteristics and relationships of the independent variables, but also from variations in the estimated coefficients (Ali, Partridge, and Olfert, 2007). I now proceed to estimate such a model in an effort to better understand the spatial explicit effect of deforestation on the magnitude of country's' adoption of PES schemes.

The total number of local PES programs adopted during 2000-2007 in each country of the developing world is expressed as a linear function with same explanatory variables included in our spatial lag model. Following Fotheringham, Brunsdon, and

Charlton (2002), the functional form for the geographically weighted model can be expressed as,

$$y_i = \beta_{i0} + \beta_{i1}x_{i1} + \beta_{i2}x_{i2} + \cdots + \beta_{ik}x_{ik} + \varepsilon_i ; \varepsilon_i \sim N(0,\sigma^2), i = 1,2,\cdots,N, \qquad (4)$$

$$\hat{\beta}_i = (X'W_iX)^{-1}X'W_iY; i = 1,2,\cdots,N, \qquad (5)$$

$$W_{ij} = 1/d_{ij}^2, \qquad (6)$$

β is the vector of coefficients to be estimated for each unit of observation *i* (i.e., country in our case). *i* subscripts on the parameters indicate that there is a separate set of *K + 1* parameters for each of the *N* observations (i.e., *N*= 124). A separate regression is estimated for each y_i in which the sub-sample is composed of each *i* within some specific bandwidth *d*. Note that the assumption that the ε errors are normally and identically distributed ($\hat{\beta} = (X'X)^{-1}X'Y$) is not longer necessary for identification purposes. GWR specification allows us to capture spatial variations in the regression coefficients by introducing a weight matrix *W* in the estimation procedure. W_i follows the same form used in our spatial lag model.

Variation in the total response from a particular variable in eq. 4 would be caused by variation in β_{ik}, variation in X_{ik}, and covariance between the two. If the variation in the expected impact of an explanatory variable is mostly due to spatial heterogeneity of β_{ik}, this would suggest that global approaches would produce a misleading picture of the impact of that variable. This is because in spatial lag models, the only spatial variation is due to differing X_{ik}, as the coefficients are fixed across the sample. For the expected impact across the 124 countries in the GWR model, consider the expansion of Var($\beta_1 X_1$) following Kmenta (1986):

$$Var(z_1 = \beta_1 X_1) = \left(\frac{\partial z_1}{\partial X_1}\right)^2 Var(X_1) + \left(\frac{\partial z_1}{\partial \beta_1}\right)^2 Var(\beta_1) + 2Cov(\beta_1, X_1)\left(\frac{\partial z_1}{\partial X_1}\right)\left(\frac{\partial z_1}{\partial \beta_1}\right) \quad (7)$$

The partial derivatives in eq.7 can be evaluated at the corresponding mean values following Ali, Partridge, and Olfert (2007). The first component of this formula can be referred to as the contribution of the spatial variation in X_1, the second component as the contribution of the spatial variation in β_1, and the third component as the contribution of covariance between $\beta_1$and X_1. Known individual components' contributions can be expressed as a percentage of total variations (Var(z_1)).

Coefficients for the adoption on local PES programs for the period 2000-2007 are reported in Table 1.4. Overall, the Akaike Information Criterion (AIC) values for all models indicates that GWR exhibits a higher descriptive power of the tradeoff between bias and variance in model construction relative to global non-spatial (OLS) and spatial (SML) counterparts. The selected bandwidth for local neighboring is d=43 km. The Robust LM test of the OLS model indicates that significant spatial lag dependence is found ($p < 0.000$). Although the approach followed for constructing our model may raise questions about multicollinearity, one advantage of the GWR results is that they reflect the compilation of nearly 124 sets of regression coefficients in which the entire set of results should offset the influence of outliers or multicollinearity.

The rates of forest cover change coefficients are found to be significantly different across countries, indicating spatial variation in the marginal impact of perceived environmental threats of the total number of local PES programs adopted. For example, at the lower quartile, the GWR coefficient indicates that at the mean rate of -3.006, the typical country would have experienced less adoption of local PES programs relative to the corresponding country at the upper quartile. Comparing the median GWR estimate to the SLM and OLS estimates indicates where the two standard global estimates fall relative to the median GWR parameter estimates. One weakness of the standard and the spatial econometric approaches is that an analyst would have likely argued that rate of forest cover change has no impact, when in fact its impact spatially varies between the upper and lower tails of the distribution (and the tails significantly differ from one another).

Variable	Min.	Lower Quartile	Median	Global (OLS)	SLM (ML)	Upper Quartile	Max.
	Dependent Variable: PES adoption						
Independent							
Policy Index	-4.025	-0.903	0.002	-0.637	-0.619	0.167	0.833
Institutional Indexes							
Party Institutionalization	-0.202	0.014	0.250	0.835	0.796	1.172	3.150
Judicial Independence	-0.147	0.012	0.137	0.144	0.141	0.425	1.065
Bureaucracy Quality	-2.849	-0.478	-0.281	-0.354	-0.345	0.051	2.777
Controls							
Legal Origin (Common Law)	-7.719	-1.458	-0.489	-1.183	-1.191	-0.230	0.099
Legal Origin (Civil Law)	-0.216	-0.087	0.271	0.640	0.653	1.371	3.895
Environmental Expenditure	-0.137	-0.052	0.022	0.141	0.138	0.239	0.494
Development Flows	-0.062	-0.013	-0.003	-0.008	-0.008	-0.002	0.000
NGO Support	0.001	0.001	0.001	0.002	0.002	0.002	0.004
World Bank	-0.920	-0.242	0.037	-0.347	0.087	0.582	6.143
Country Wealth	-0.001	0.000	0.000	0.000	0.000	0.000	0.000
Water withdrawal	0.000	0.000	0.000	0.000	0.000	0.000	0.001
Forest cover change 1990-2000	-0.075	-0.050	-0.018	-0.039	-0.040	-0.011	-0.010
Constant	-2.427	-1.457	-0.496	-1.056	-1.079	-0.232	0.560
No. of observations	124						
Adjusted R^2					0.396		
Akaike Information Criterion (AIC)	448.522				543.898		

Table 1.4 Parameter Summary of GWR, OLS, and SLM Estimation

Note: Unless otherwise indicated, all variables are measured in the initial period 2000. See the appendix for countries included and Table 1.1 for variable definitions. OLS = Ordinary Least Squares, SLM = Spatial Lag Model, ML = Maximum Likelihood, GWR = Geographically Weighted Regression. Fixed bandwidth 43 km. Regression Diagnostics: Multicollinearity Condition Number 17.168 >15 indicates multicollinearity; Jarque-Bera (test on normality of errors) 0.000; Diagnostics for Heteroskedasticity: Breusch-Pagan (random coefficients) OLS = 0.000, SLM = 0.983; AIC = -2 * log-likelihood + 2 * (number of parameters). Adjusted R^2 for SLM refers to the squared correlation between the observed and predicted values of the dependent variable.

	Variables (X)		Parameters (β)		Pred. comp. (Xβ)a		Percent of Variance (Xβ) Attributable to		
Variables	**M (1)**	**SD (2)**	**M (3)**	**SD (4)**	**M (5)**	**SD (6)**	**Var (X) (7)**	**Var (β) (8)**	**Cov (Xβ) (9)**
Total variation in y (PES Program adoption 2000-2007)	1.008	2.643							
Policy Index	1.287	0.722	-0.785	1.914	0.362	11.885	34.36	90.81	-25.16
Party Institutionalization	1.105	0.759	0.877	1.374	2.396	11.000	29.64	48.01	22.35
Judicial Independence	1.198	0.911	0.298	0.477	0.900	1.539	29.45	47.83	22.72
Bureaucracy Quality	1.049	0.790	-0.156	2.001	1.626	14.651	31.78	72.28	-4.06
Legal Origin (Common Law)	0.210	0.409	-1.960	3.272	0.020	4.283	57.92	49.98	-7.90
Legal Origin (Civil Law)	0.452	0.500	1.047	1.710	1.053	2.801	36.80	41.78	21.42
Environmental Expenditure	2.719	1.708	0.113	0.254	1.323	3.993	35.79	50.19	14.02
Development Flows	16.071	33.552	-0.016	0.026	-0.011	7.375	55.56	48.20	-3.77
NGO Support	827.895	658.442	0.002	0.001	2.805	10.854	53.36	20.47	26.17
World Bank	0.036	0.138	1.120	2.860	1.450	4.524	47.62	50.28	2.10
Country Wealth	1,424.371	1,664.194	0.000	0.000	0.136	3.424	50.95	58.86	-9.82
Water withdrawal	478.395	687.876	0.000	0.000	0.820	2.775	49.61	42.33	8.06
Forest cover change 1990-2000	-3.006	13.356	-0.033	0.029	0.483	2.648	69.36	26.46	4.18

Table 1.5 Decomposition of Variance of the Predicted Components due to Individual Factors

Notes : a. Contribution of variable or variable groups to the predicted values of the dependent variable. Standard deviations of the predicted component in column 6 are calculated using the typical procedure to estimate a standard deviation (for the 124 observations). For individual explanatory variables in column 6, standard deviations are calculated using the following formula from Kmenta (1986). The partials can be evaluated to the corresponding mean values. Therefore, the partial in the first component is equivalent to the mean of the squared β_1 values, the partial in the second component is equivalent to the mean of the squared X_1 values, and the partials in the third component are equivalent to the means of β_1 and X_1, respectively.

The bandwidth equals approximately 43 kilometers, after which w_{ij} is set equal to zero. W is then row standardized such that the sum of each row equals one, or the spatial-lagged variables can be interpreted as a distance-weighted average of the neighboring parish values. W was calculated using the Stata 10.0 software using a program written by Maurizio Pisati of the Department of Sociology and Social Research, University of Milano Bicocca, Italy.

Moreover, GWR allows observing the heterogeneity of variable across space and total contribution to predicted impacts. Table 1.5 presents the decomposition of contribution from observed characteristics of the variables (*X*), and GWR parameter estimates (β) across the 124 countries. The important interpretation of this result is that if Var ($X\beta$) is mostly due to Var (βik), OLS approaches would produce a misleading picture of the impact of that variable. The variable NGO support has important impacts on adoption on local PES programs, yet the variation of its impacts across the 124 observations is caused by different components. For example, variation across developing countries in the total impact of having NGO support is almost entirely because of variation in the response to NGO support (Varβ) rather than variation in the number of NGOs. Thus, standard approaches would greatly underestimate its true spatial variation because they would primarily focus on variability in the number of NGOs. Conversely, spatial variation in the impact of the indexes of institutional strength (e.g., Party institutionalization) is much more due to its observed shares (Var*X*) versus its marginal impacts, which means that the spatial variation of the impact of institutional characteristics on the adoption of local PES programs is likely to have been captured by standard SLM approaches.

1.8 Conclusion

Natural resources management occupies an increasingly important place in public administrators' agenda given the rising pressures on important ecosystems and communities relying on their goods and services. Payment for environmental services (PES) programming is one market-based approach that increasingly has been used in local efforts to protect watersheds and watershed services such as water quality and quantity (e.g., FAO, 2007; Ortega-Pacheco *et al.,* 2009). Despite the increasing interest in the literature for the potential that using self-financing, contract-based PES approaches offer to achieve environmental and natural resource goals (e.g., Postel and Thompson, 2005; Ferraro and Kiss, 2002), previous research has not systematically examined their wide spread adoption in developing countries during the last decade. The results of this research provide preliminary evidence that this phenomenon can be explained through the lenses of competing theories of diffusion. From a policy perspective, the synergy from endogenous and exogenous factors that seem to affect diffusion of local PES program can be exploited to realize their relative performance impact and fiscal significance for the protection and management of water resources. Our estimated model suggests that international efforts to increase PES adoption could be enhanced by increasing the quality of institutional endowments and the work of NGOs—especially when neighboring countries are also adopting PES schemes.

Using country level data, this paper studies developing countries interdependent degree of adoption of PES approaches for managing watersheds services. From a survey of recent assessments of PES, I identified patterns of adoption and the exogenous and endogenous economic, political and institutional factors explaining the number of PES that have been implemented by developing countries in Latin America, Africa and Asia during the last decade. By drawing from field of institutional economics and transaction costs theory, I derived an alternative theoretical explanation for diffusion of institutional innovations and testable hypothesis: that the increased adoptions over time and space of PES programs can be interpreted as diffusion of interdependent induced institutional innovations. The process of interdependent institutional innovation is argued to be induced by varying levels in resource and cultural endowments, and technology across countries. These factors reduce transaction costs, increase social demand for improved water quality, and enable PES adoption. An empirical model was estimated to test how well this theoretical proposition can empirically explain the observed pattern of PES adoption. One indicator of the level of overall institutional strength (i.e., Party institutionalization) is found to have a positive significant effect on a country's total number of local PES programs; the number of NGOs in a country also positively correlates with PES adoption.

Although this paper provides insights for public managers and researchers interested in market-based administrative alternatives for water resources conservation and development, there are major limitations that prohibit an accurate assessment of the relative explanatory power of competing diffusion theories for explaining adoption of local PES programs. One key limitation is the impossibility of accounting for spatially explicit non-stationarity. This limitation is the result of the unavailability of local-level data which ultimately pointed to the use of a global spatial lag model. By construction, this model specification captures the spatial dependence of observed realizations of variables under study. However, this model ignores non-stationarity, which results in biased predictions and misrepresentation of country specific effects. This can help explain the inconsistency found between the expected and estimated direction of effects for some control variables, particularly in the case of deforestation. In this context, geographically weighed regression analysis seems to offer valuable tools to account for spatial non-stationarity across countries. Further research should account, however, for non-stationarity of local contributions of different factors at watershed level.

Moreover, in our models, the spatial weights matrix, W, is a matrix of geographic distances among units. In studies of diffusion, it is also important to measure influence along other channels such as competition and learning (e.g., measures of neighboring boundaries, political backgrounds, etc.) while controlling for geographical interdependence. Note that single or group membership in regional or international network of support for PES program would tend to reduce transaction costs associated with the implementation and operation as it is likely to increase available information and, thus, positively affect the number of adoptions. However, PES networks seemed to have become stronger only in the most recent year. In this study, it is assumed that the network effect would become relevant in future studies as the diffusion of PES programs continues to spread.

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Chapter 2: Measuring transaction costs and comparing its role on mitigation policy designs.

2.1 Introduction

Transaction costs are not usually included in climate change mitigation policy evaluation. Incorporating them may increase the efficiency and sustainability of policies by adjusting the optimal amount of emissions control, the optimal resource allocation, or the choice of policy instrument. This may be especially true for mitigations actions in developing countries. However, transaction costs must first be measured to be included in the evaluation of alternative mitigation policies. The objective of this essay is to report empirical measures of transaction costs resulting from alternative designs of land-based mitigation activities. Specifically, two designs introducing incentives will be scrutinized and decomposed into main components in order to compare trade-offs concerning the resulting magnitude of different types of transaction costs and overall environmental outcomes.

2.1.1 Research Question

Reducing greenhouse gas (GHG) emissions is the focus of domestic and international policies to reduce the risks of anthropogenic climate change. A number of studies have now suggested that land-based carbon credits can reduce the costs of meeting stringent GHG goals by developed nations (e.g., Sohngen and Mendelsohn, 2003; Tavoni et al., 2007, Nabuurs et al, 2007; Kindermann et al., 2008). Many of the credits that these studies anticipate are derived from actions undertaken in developing countries through afforestation/reforestation activities or reductions in emissions from deforestation or forest degradation. One critical concern about the cost estimates in these studies has arisen. These studies assume that international credits can be generated for the exact cost incurred by land owners and managers. However, it has been noted that transaction costs could raise the total costs of generating land-based carbon credits to a level significantly higher than that set of on-the-ground costs (Antinori and Sathaye, 2007).

The policy need for incorporating transaction costs of mitigation policies is at the center of the international negotiations of the climate regime. The Bali Action Plan (BAP), adopted by the Conference of the Parties (COP) as decision 1/CP.13 in 2007, launched a comprehensive process to enable the full, effective and sustained implementation of the Convention through long-term cooperative action. As part of the BAP, parties agreed that subject for the negotiations include approaches to enhance the cost-effectiveness of mitigation actions, including market mechanisms. The Kyoto Protocol's Clean Development Mechanism (CDM) – in whichever form (project-based, programmatic, or sectoral) – or alternative designs of mechanisms for reduced emissions from deforestation and forest degradation (REDD) are among possibly several policy alternatives under negotiation. These options are likely to be associated with different levels of transaction costs that would affect cost-effectiveness of mitigation policy as well as the relationship envisaged in the BAP with respect of the provision of finance from developed country parties for implementation of mitigation actions in developing countries.

2.1.2 The transaction cost-effective strategy

The literature on analysis and measurement of transaction costs for evaluating environmental policy is rapidly expanding (e.g., Dorward, 2001; Falconer and Saunders, 2002; Krutilla, 1999; McCann et al., 2005), particularly in the context of tradable pollution permits (e.g., Hahn and Hester, 1989; Keeler, 2004; Stavins, 1995) and climate change (e.g., Antinori and Sathaye, 2007; Antle et al., 2003; Cacho et al., 2005; Chadwick, 2006; Michaelowa et al., 2003; Woerdman, 2005). While the body of descriptive and theoretical literature is growing, the empirical literature is far more limited. This empirical literature focuses on costs borne by services providers (i.e., market transaction costs) while largely ignoring costs faced by administrators (i.e., political transaction costs). The few studies exploring the magnitude and effects of transaction costs in climate policy recognize the methodological difficulty of measuring and comparing the transaction costs. Cacho et al. (2005) reported that their attempt to measure transaction costs was unsuccessful, among other reasons, because of data limitations. They argued that transaction cost data unavailability can be associated with the lack of standard accounting practices or reporting mechanism to collect this information.

Measurement limitations seem to be exacerbated in the context of political transaction costs. For instance, Cacho et al. (2005) attempted to measure market transaction costs of project-based emission trading scheme, although they did not fully consider the implication of political transaction costs. Some observers have argued that political transaction costs should be optimized jointly with other costs (i.e., opportunity costs of generating land-based carbon credits) across time to fulfill the policy objectives (Falconer et al., 2001). The underlying concept is that efficient policy design requires knowledge about how government choices on expenditures relate to the quantity and quality of land-based carbon credits generated. Recent research has found that the design of a reduced emissions from deforestation and forest degradation (REDD) policy framework can have a significant impact on monitoring costs as they seem to follow economies of scale (Bottcher et al., 2009). Costs may vary from 0.5 to 550 US$ per square kilometer depending on the required precision of carbon stock and area change detection.

Similarly, the scale of transaction costs can affect choices in policy design and implementation. In fact, the importance of filling the gap for sufficient and comparable empirical measures and measurement methods for estimating transaction costs cannot be understated. Tavoni et al. (2007) suggest that reductions in deforestation can reduce costs of meeting stringent carbon targets by 40% in coming years. These estimates, though, do not account for transaction costs. Antinori and Sathaye (2007) reported that average transaction costs are about 19% of total carbon sequestration projects costs. The effect on credit supply is potentially dramatic: as a rough standard of comparison, a supply elasticity of 0.5 implies that transaction costs of 19% would reduce the supply of credits by about 10% from the predictions of current models that ignore such costs.

This paper reports empirical measures of transaction costs from land-based mitigation actions implemented in Ecuador. This paper begins by sketching the relevant theory, and then move to the specifics of our measurement strategy. Then the paper continues to describe the method use for primary data collection and to present results. The discussion will focus on comparison of magnitudes of transaction costs types across alternative designs. The paper concludes by drawing some general lessons from this investigation.

2.2 Mitigation policy design

A useful point of departure for understanding transaction costs effects is the general literature on transaction costs in economics (e.g., Hodgson, 2000; North, 1990; Schimd, 2004; Williamson, 1981). This literature provides a framework for analyzing the most significant sources of transaction costs, and assessing their magnitude and effect on the efficiency of public policies. This framework has been employed in the context of broad environmental policy evaluation; however, its usefulness in the context of climate policy has not being fully explored. The research reported here has also been informed by suggestions in the literature on innovative methods (e.g., McCann et al., 2005) and field research (e.g., Adhikari and Lovett 2006; Kuperan et al., 2008; Meshack, et al. 2006; and Mburu et al., 2003) attempting to measure different types of transaction costs.

2.2.1 Influence of institutional design and transaction costs on mitigation decisions

Institutional economics posits that transacting agents making decisions do so in a costly way (e.g., Coase, 1960; Commons, 1931; Dixit, 1996; North, 1987; Williamson, 1979). For example, farmers deciding on implementing land uses for providing carbon credits or a broker/dealer interested on executing carbon credit trades base their decisions not only on the price they expect to receive for each credit, but also on additional costs related to transacting in these markets. Similarly, regulators responsible for facilitating transactions in this market base their decisions on monitoring and management of carbon programs not only on the social benefit associated with specific levels of pollution control but also on the costs associated to achieving such target. For the purpose of this paper, the transaction of carbon mitigation services is defined as consisting of the provision, verification, and use of land-based mitigation credits generated in developing countries and accepted for compliance purposes in international emissions markets or domestic cap-and-trade programs. This transaction can be thought as having three major components: 1) *implementation and administration*, 2) *mitigation activities*, and 3) *credit trading*. Transaction costs in these components can occur simultaneously (see Figure 2.1). This paper focuses on measuring transaction costs for the first two components. The third component, although important, should be treated in a systematic fashion in separate work. It is useful to analyze the transaction costs occurring within each of the components of the transaction in a framework of different types and categories. The table below summarizes transaction costs types and categories taking place within each of the components of the transaction of carbon mitigation services.

Component	Categories	Types
1. Implementation and Administration	a. Political and Market	*i. Search*
2. Mitigation activities	b. Fixed and Variable	*ii. Negotiation*
3. Credit Trading	c. Explicit and Implicit	*iii. Enforcement*

Table 2.1 Costs types and categories across components of the transaction.

North and Thomas (1973) break down transaction costs into three main types: search, negotiation, and enforcement costs. Information costs occur before the exchange takes place and include aspects such as searching for attributes that could facilitate the transactions, seeking better prices, and looking for potential buyers (Key et al., 2000). Bargaining or negotiation costs are incurred during the exchange and include the time to negotiate a contract, reach an agreement, and make arrangements for payment. Monitoring and evaluations costs are incurred to ensure that the conditions of an exchange are met (e.g., enforcing the specified attributes of the carbon credit).

In addition, transaction costs types are categorized in three dimensions: fixed versus variable costs, explicit versus implicit costs, and political versus market costs. The first transaction component, *Implementation and Administration*, can be mainly thought of as in the domain of political transaction costs, that is, all costs that are borne by the government in order to facilitate market transactions across time. The second and third components, *Mitigation Activities, and Credit Trading*, are in the domain of market transaction costs, meaning the costs borne by private agents to execute market transactions within a given institutional setting. North (1990, p. 51) suggests that political transaction costs are higher (and more difficult to measure) than market transaction costs. Generally, transaction costs borne by the government are not fully covered in economic analysis of climate policy instruments because they are different to those costs conventionally considered as transaction costs (Banuri, 2001, p. 52; Barker et al., 2007).

Market and political transaction costs can be fixed or variable. Fixed transaction costs are those independent of the mitigation credit quantities sold or bought as well as the size of land parcels involve in the transaction. Variable transaction costs change according to how much credit quantities sold or bought in the transaction (e.g., price premiums deriving from bargaining capacity) (Fabozzi et al., 2006) and the number of contracts and area of land associated to such contracts. Transaction costs also depend on observability. Explicit transaction costs are those cost that are observable and known upfront such as commissions, fees, and taxes. Implicit transaction costs, on the other hand, are unobservable and unknown in advance. Some transaction costs that fall in this category are credit market impact costs and credit trading opportunity cost as well as those costs associated to risks of unexpected events affecting land use such as fires. The implicit costs can account for a large part of the total transaction costs in some cases (Fabozzi et al., 2006).

2.2.2 Design 1: focus on environmental integrity

Domestic and international climate policies focus in reducing GHG emissions for stabilizing global temperatures. Providing an apparatus for financial transfers from the North to the South, the United Nations Clean Development Mechanism (CDM) has rapidly become the poster child case for the use of project-based policy instruments. One objective of the CDM is to fund land-based (e.g., afforestation/reforestation, and Land Use, Land-Use Change and Forestry) projects jointly achieving: 1) environmental integrity - represent real, measurable reductions of GHGs beyond what could be expected to occur in the absence of a project, and 2) sustainable development. The CDM is supposed to enable developed country parties of the Kyoto Protocol to meet their emission requirements at lower cost by investing in GHG emission reductions in developing countries. However, implementation of CDM projects entails significant costs (Stavins, 1995). This is particularly true for land-based projects because of the difficulty of establishing baselines and the cost of measuring or estimating CO_2 fluxes – on a per-ton basis - from diverse land parcels. In addition, researchers have pointed out that, to ensure environmental integrity (i.e., determining what GHG reductions are truly additional, permanent and without leakage), CDM projects are exposed to burdensome approval, monitoring and evaluation procedures (Chadwick, 2006).

In fact, CDM projects will pass through several stages and a well-established governance structure from initial conception through approval and the issuance of marketable Certified Emissions Reduction [CER] credits. In terms of the governance structure, the CDM requires a National Designated Authority [NDA] to monitor and to facilitate the transaction. In Ecuador, the NDA is the Ministry of Environment. In addition, the CDM Executive Board [EB] exercise some control by screening CDM projects and requiring validation of the methodology and baseline used for the project by a designated operational entity (DOE). Ecuador DOE promotes the CDM program and provides support for the full arrange of project types feasible under the CDM including afforestation/reforestation activities.

Moreover, each CDM development stage may be supported by in-house staff, but usually involves hired consultants who are specialists in CDM technologies and legal requirements. The process begins with a project concept and initial design effort. The

Project Idea Note [PIN] identifies opportunities to use the CDM, the main participants, necessary technologies, the scale and magnitude of expenses, and estimated payoff streams and revenues. Subsequently, projects developers must decide on a methodology to use to determine the level of greenhouse gas reductions envisaged. If existing methodologies are somehow not useful or appropriate for the project, project developers may propose their own methodology and submit it for CDM EB approval. A third stage requires developers to prepare a CDM project design document (PDD), which is an official United Nations CDM document. The PDD will then be evaluated by DOE, DNA and the EB before it can be approved, registered and receive CERs for emission reductions. As a matter of fact, a project can receive saleable CERs only after it is registered, underway, has achieved emission reductions as planned and had those reductions verified.

2.2.3 Design 2: focus on environmental outcomes

An alternative to domestic implementation of mitigation actions under United Nations framework has received increasing attention, Reduced Emissions from Deforestation and Forest Degradation in Developing Countries (REDD). In this regard, Ecuador has recently implemented a programmatic approach, Socio Bosque, that could serve as a model for REDD implementation in Developing Countries. Socio Bosque is based on a single conditional payment covering multiple environmental services. The program has targeted low-income landholders and paid particular attention to informing participants and to simple contract language. As REDD has not yet been officially adopted as a mechanism under the United Nations framework, Socio Bosque has been designed, implemented, and supported first at a regional scale and later at the national scale with the help of public domestic financing. This national incentive-based scheme is significantly different than CDM because it does not focus on compensating landowners for the provision of individual ton-per-ton mitigation services. Instead it offers a conditional single payment for land that is managed in ways consistent with conservation and the associated avoidance of emissions. Socio Bosque also has an explicit objective to contribute to poverty alleviation.

To attain this mix of objectives, Socio Bosque has operated with the central government as a single payer for multiple service contractors. The program uses a single contract that specifies a voluntary action in exchange of a conditional payment. Similar schemes have been argued to reduce transaction costs associated with equity concerns (e.g., minimize contractual stages and facilitate access to information among landowners groups, small farmers in particular) (e.g., Zbinden and Lee, 2005). Socio Bosque has also targeted payments, on a priority basis, to encourage the sustainability and recuperation of forests in selected areas (e.g., Sánchez-Azofeifa et al., 2002), but also uses differentiated payments to address equity considerations and budget constraints.

Socio Bosque is managed and operated by an administrative unit under the umbrella of the Ministry of Environment of Ecuador. The program emphasizes giving full information to landowners about general provisions and individual contract terms. Submitted applications are verified for compliance with requirements and targeted priorities. Priority areas are set based upon deforestation pressure, ecosystem services (i.e., carbon, water and biodiversity) productivity, and the socioeconomic status of participants. Landowners sign contracts after both they and the government have verified information and approved the terms. Once contracts are settled, program payments are made bi-annually directly to beneficiaries' bank accounts (See Table 2.2).

A typical contracting process takes six to eight months. Landowners must put together a binder with documentation that includes a formal land title, certification from estate registry, proof that the land is not currently under financial obligation with a commercial or development bank (i.e., if the land is mortgaged, proof of payment fulfillment is required), proof of holding of bank account were the payments can be transferred, and a basic investment plan for Socio Bosque resources. Landowners must list intended uses in their investment plans.

Administrative processes	Landowners activities
1. Call for voluntarily expressions of interests	1. Compilation and presentation of submissions with required documents (land title, bank account, etc)
2. Validation: a) Check of documentation, b) Geographic prioritization upon ecological and socio-economic criteria. c) Field inspections.	2. Compilation and presentation of submissions with required additional documentation
3. Contract settlement	3. Contract signing
4. Incentives payment - transfer	4. Reception of payments
5. Monitoring	5. Adjust activities and use of incentives as necessary according to investment plan and contract
6. Evaluation	6. Across time would decide to renew contract

Table 2.2 Socio Bosque Program activities

The term of the contracts is for twenty years. The amount to be transferred per hectare per year depends on the area an individual landowner or community desires to conserve. For areas smaller than 50 hectares, the payment is $30 per hectare per year. The payments per hectare decrease for larger areas and are set to change across time. During the term of the contract, the program plans for compliance to be regularly monitored at an individual contract level through the use of remote sensing and field visits. Evaluation at a broader scale spanned for through the establishment of a national accounting system for determining baselines and measuring changes in forest coverage, deforestation rates, and emissions of GHGs from deforestation. A monitoring scheme has also been implemented to measure social and economic impacts of program payments. Annual frequency and level of detail of monitoring is based upon available information on determinants of land use change: accessibility and distance to markets.

2.3 Empirical design

For simplicity, here we assume that the provision of land-base credits is derived from landowners' voluntary participation and the performance of the resulting contractual agreements between the government (buyer) and landowners (seller). Therefore both the landowners providing the services and a government facilitating the contracts face transaction costs.

2.3.1 Analytical Model

The magnitude of market and political transaction costs can be estimated a) across time, b) within each component of the transaction (i.e., *Program Implementation & Administration* and *Mitigation Activities*), and c) by design characteristics such as:

a. scale, scope and stage of implementation (i.e., project-based or programmatic approach; national or regional; and whether it has been recently established or is operating for sometime)

b. characteristics of parties involved, and organizational procedures:

    i. characteristics associated with the contract (e.g., period), the service buyer (e.g., government personnel training and staff size, organizational structure and tasks division), the service provider (e.g., size of parcels, accessibility, proximity to local populated areas with services, individual preferences)

    ii. type and number of stages for approval and operation

In particular, this research assumes that the more precise the design of the management prescriptions and more demanding the verification of generated credits[xix], the more effective will be project or the program, and the lower the potential for contractors to avoid contracting terms. However, this effectiveness will come with high transaction costs across components. Table 2.3 lists sources, categories and types of transaction costs relevant for this study.

Implementation & Administration

The variation in the relative shares of costs faced by the government is partly related to the nature of the program, but also reflects the stage of its development (Dorward, 2001). Land-based mitigation actions may require fixed cost implementation-type activities in their firsts years, as the details of implementation are finalized and the program is set up; administration-related transactional activities then may rise in relative importance. The political transaction costs can be thought as comprising implementation and administrative costs. The *Implementation and Administration* component of the transaction can involve recurrent or annual resource needs relative to the likely workload in each year.

The analytical model allows changes over time in the mix of administrative activities linked to the time profile of the activity cycle. In particular, this paper posits that at some point the balance will switch from "implementation or set-up" activities (such as promoting the program and entering into contracts) to more routine "maintenance" activities (like those related to checking compliance with contract terms). As the transaction costs of the *implementation and administration* component include overhead costs -- which are partly fixed -- the administrative cost functions are expected to exhibit size economies (falling marginal costs as a project or contract encompasses a larger area). Inflexibility in administrative structures is also considered (e.g., planned staffing adjustments and public budget setting process are likely to be made only on a yearly basis).

Sources (Scale/ Implementation Phase/ time)	Component	Costs category	Costs type	Fixed costs	Variable costs
Local, regional or national / Pilot or full scale phase (year *t*)	Administration & Implementation	Political	Information	Equipment - Supplies & administrative tasks	Initial Contact, Submission reception & registry
			Negotiation		Inspection, info validation and contract negotiation
			Monitoring Evaluation		Field visits, and field & office evaluation
		Market	Information	Initial Contact, Application submission (*explicit*: fees on gov agencies).	(*implicit*: transportation and opportunity costs of time involved)
			Negotiation	Inspection and contract negotiation	
			Monitoring Evaluation		Field visits, and evaluation
	Mitigation activities	Political	Information	Equipment - Supplies & administrative tasks	Geo-coding and conservation activities info
			Negotiation		Conflict resolution
			Monitoring Evaluation		Field visits & evaluation
		Market	Information	Info on types of activities	
			Negotiation	Conflict resolution	
			Monitoring &Evaluation		Field visits & evaluation

Table 2.3 Transaction costs categories and types across components and time

Mitigation Activities

Information and bargaining costs are expected to be mostly fixed at the project level, although there may be economies of scope in doing multiple projects at the same time. Contracting costs have both fixed and variable components. For example, the negotiation costs for participants include a fixed cost of contacting the government agency implementing the scheme, to indicate the landowner's wish to negotiate entry. However, there is also a degree of variability in costs because the scope of negotiation will vary with the size of the land parcel under consideration. Monitoring costs can have both a fixed component for each project and a variable component depending on the size of the areas that must be measured.

2.3.2 Data

This research attempts to compare the magnitude of transaction costs associated to mitigation activities both in project-based against program-based approaches, across stages of development and within their scopes of implementation. Given that there are not project-based afforestation/reforestation CDM activities registered and verified in Ecuador, this paper presents data on transaction costs affecting project-based CDM-like initiatives in Ecuador reported in the literature. Secondary data on transaction costs associated to project-based design approaches focused on environmental integrity is identified on published information. The project-based reported initiative offers a useful point of comparison as it has marked design differences relative to the Socio Bosque programmatic approach.

This research combined both qualitative and quantitative methods to generate primary data on transaction costs affecting the provision of mitigation services within a programmatic activity in Ecuador. Data on transaction costs across components were collected from landowners and government officials to assess market and political transaction costs, respectively. The research team used an iterative design approach including focus groups and pretesting to develop instruments (Desvousges and Smith, 1996; Kaplowitz et al., 2004) that respondents understood and accepted as plausible, and shared consistent interpretations of implemented policies (Johnston et al., 1995). The initial experimental design of this research drew information from previous research in Ecuador offering measures of costs for the supply of credits as a function of contracting costs (e.g., de Koning et al. 2004). During a first stage, information was collected from stakeholders to refine the initial experimental design using qualitative methods (e.g., focus groups). The second stage generated the necessary information to identify market and political transaction costs and empirically explore their magnitude at one point in time. During the second stage secondary data was gathered in order to explore changes across time in political transaction costs.

The application of semi-structured interviews (e.g., Adhikari and Lovett 2006; Kuperan et al., 2008, Meshack, et al. 2006; and Mburu et al., 2003) and in-person questionnaires have been identified as promising options in empirical analysis of transaction costs and can generally provide useful information for environmental policy evaluation (McCann et al., 2005). Consequently, given the relative importance and small number of program staff this research used semi-structured interviews to collect data on political transaction costs. In-person questionnaires were designed to elicit info on market transaction costs from landowners.

A complete enumeration was conducted to examine how landowners with heterogeneous socio-economic and geographical characteristics perceive transaction costs associated with a programmatic approach for the *Implementation and Administration* and *Mitigation activities* components of the transaction. Landowners who participated in this exercise are located in the province of Esmeraldas, a rain forest region of Northwestern Ecuador where the establishment of palm plantations and pervasive deforestation has created land use conflict. This site has received increasing attention as the likely demonstration area for Ecuador's emerging climate mitigation and energy policies. In fact, Esmeraldas province posses the larger share of individual contracts under the programmatic approach, Socio Bosque.

The differences across landowner characteristics provide useful variation in a number of key attributes related to transaction costs. Elements of the questionnaire characterize transaction components[xx] and the aforementioned sources of transaction costs identified in the literature and identified as relevant to market transaction costs by focus groups and key stakeholders (e.g., regional experts and decision-makers). Interviewers canvassed all landowners in the selected region and, after obtaining informed consent, administered an in-person questionnaire. The information elicited from questionnaires was entered into a database and analyzed.

Measures of political transaction costs from program staff were collected using in-depth cognitive interviews (semi-structured interviews), (see Kaplowitz and Hoehn, 2001; Patton, 1990; Weiss, 1994). Semi-structured interviews were conducted with government officials using a discussion guide and following standard practices including audio-recording the sessions. The in-depth interviews were transcribed and, together with the interviewers' notes, systematically coded and analyzed following generally accepted qualitative data analysis methods (Strauss and Corbin 1998). Political transaction costs measures were derived from information assessed through semi-structured interviews and supplemental official secondary data provided by program staff.

2.4 Results

2.4.1 Empirical measures of Transaction Costs for project-based design

Wunder and Albán (2009) reports updated transaction costs measures for a Afforestation/Reforestation (A/R) CDM-like initiative in Ecuador - PROFAFOR[xxi], which has been establishing plantations to sequester carbon since 1993 through signed contracts (i.e., a total of 152) with private owners and local communities. Although at the aggregated level this initiative has a monitoring process certified by the Swiss company *Société Générale de Surveillance* [SGS], the captured carbon is in part not eligible for the issuance of credits under the Kyoto Protocol framework as its year of launching is before the established in the Protocol (i.e., about 80% of projected emissions). There are various concerns associated to this initiative. This includes not only additionality issues associated to the established baseline but also a lax incentive-conditionality scheme, absence of explicit development aspects in its design, and permanence and leakage considerations.

Wunder and Albán (2009) updated and supplemented data from Albán and Arguello (2004) gathered in 2002 and 2003 for a study on the socioeconomic impacts of payment for environmental services [PES] systems from six community plantation contracts through community-based workshops and family-level interviews. Data was updated in a selective way, and new data was gathered on the implementation and administration costs of the projects.

The transaction costs reported are mostly the costs incurred by the project. For the purpose of this paper, Wunder and Albán (2009) transaction measures have been name-matched in a way consistent to the underlying concepts of types and categories presented here. PROFAFOR political transaction costs are reported in Table 2.4. The total project costs are around US $6.54 million for the period 1994-2005. The total project costs per hectare arrive at US $ 293. It is distributed as follows: 6% for the launching costs and

94% for operational expenses. The costs of design and implementation has been estimated in about US $ 375,000 of which 86% are the design costs of the project (e.g., contracts, design of the outline, design of the monitoring, modification recommendations by the certification agency, negotiation processes). The operation or running costs are of US$ 6.7 million, and the largest item corresponds to payments (74%) and political transaction costs (26%) comprising administration, monitoring, promotion, and certification costs. Administration and monitoring costs contribute with 76% and 13% of total transaction costs, respectively. PROFAFOR suspended the signing of new contracts in year 2000 and began a process of administration cost decrease (see Figure 2.1). Political transaction costs for implementation are reported to be about US$ 17 per hectare and average annual administrative transaction costs are about US $ 6 per hectare. FACE (2004) estimated a total fixation amount of about 2.23 million tons CO_2 for the first 10 years of the plantations' operation[xxii].

	PROFAFOR (Afforestation / reforestation) (US $ in millions)	
Implementation & Administration		
Implementation or set-up		0.37
Administrative *		0.74
Sub-total	*1.11*	
*Mitigation activities***	*0.18*	
Total Political Transaction costs	1.29	
Total projected CO_2 tons for 10 years (in million tons)***	2.23	
Political transaction cost per ton CO_2	0.58	

Table 2.4 PROFAFOR political transaction costs during 1994 - 2005.

* Present value for 1994-2005, comprises management recurrent costs, ** present value for 1994-2005, comprises monitoring, certification, and promotion recurrent costs. Different from running costs reported by Wunder and Albán (2008) as costs of direct payments have been removed given that are not consider transaction costs in the present study,

*** average capture or fixation potential of 180 tons CO_2 per hectare.

Sources: (Wunder and Albán 2009).

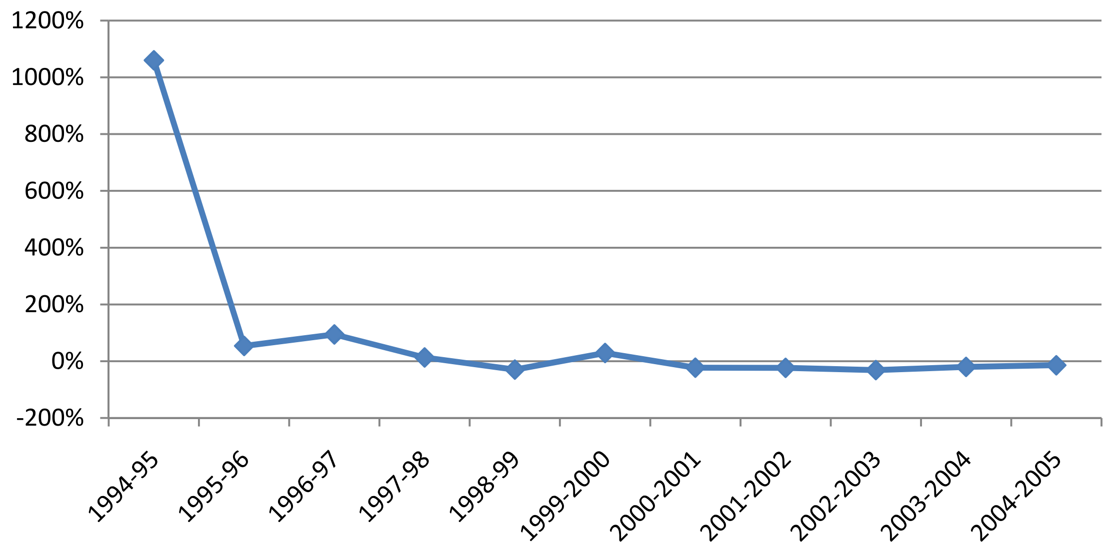

Figure 2.1 PROFAFOR administrative transaction costs change relative to previous year during 1994 - 2005.

2.4.2 Empirical measures of Transaction Costs for program-based design

Political and market transaction costs of the programmatic approach, Socio Bosque, were collected for the purpose of this study. Political transaction costs reported represent data and information facilitated by official sources during focus groups, voluntary and inform-consented semi-structured interviews. Supplementary information in the form of budgetary forms has also been used to explore changes in political transaction costs across time and scale. Two government officials were chosen for semi-structured interviews given their high discretionary power over budget decisions and level of involvement on the different components of the transaction since the inception of the REED-like program.

An official database of total 93 landowners in Esmeraldas was used to explore market transaction costs. This group of consists of the landowners who had been contacted by, or who have voluntarily contacted, the Socio Bosque program in order to explore the possibility of joining. This group includes landowners with different ethnic and socio-economic characteristics, and is geographically distributed across the agro-climatically heterogeneous landscape of Esmeraldas. Only three contacted landowners reported that they have decided not join the program. This decision was based on the perceived cumbersome approval process and the absence of formal tenure over land.

A complete enumeration of landowners through in-person interviews was conducted. Out of those who have decided to join Socio Bosque program, this research collected 73 complete questionnaires. According to AAPOR (2008), this study reports a response, refusal and noncontact rates of 0.82, 0.13 and 0.5, respectively.

On the one hand, reported measures of political transaction costs represent government disbursements for program establishment at different scales across time. On the other hand, measures of market transaction costs represent average market transaction costs per contract in Esmeraldas, Ecuador for year 2010. Political transaction costs measures together with average annual transaction costs are then used to make projections of the net present value of total program transaction costs for the 20-year contract period.

According to official budgetary information, total program costs borne by the government during 2008-2010 are about US $7.9 million. It is distributed as follows: 12% for the launching costs and 88% for operational costs expenses. The costs of design and set-up has been estimated in about US $ 0.47 million of which about 51% are the design costs of the project (e.g., contracts, design of the outline, design of the monitoring). The operation or running costs are of US$ 7.43 million, and the largest item is payments (64%), with the balance going to political transaction costs (36%) comprising administration, monitoring, promotion, negotiation and baseline establishment activities across transaction components. Costs associated to *implementation and administration* and *mitigation activities* contribute with 76% and 24% of total political transaction costs. The total program costs borne by the government per hectare is calculated about US $ 14.65. There are approximately 540 thousand hectares registered under the program. Political transaction costs for set-up are reported on about US$ 2.64 per hectare and average annual political transaction costs are about US $ 2.76 per hectare. Figure 2.2 shows a process of change in the distribution of political transaction costs shares across scales of implementation and a tendency of implementation and administrative costs to decrease relative to mitigation activities over time.

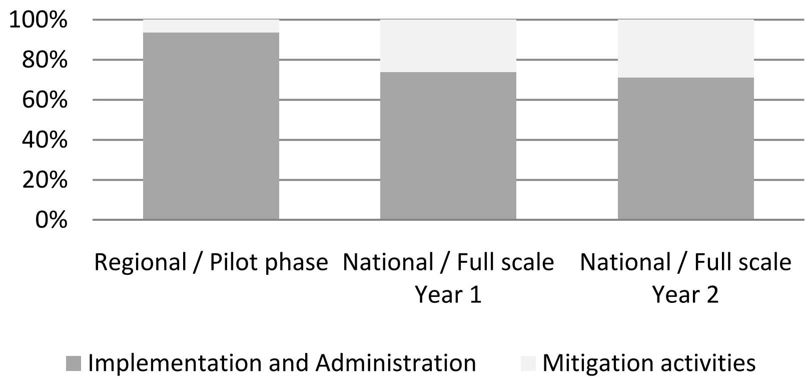

Figure 2.2 Changes in shares of political transaction costs type across scales and implementation phases over time.

Based on this research field work, Table 2.5 summarizes calculated measures of average market transaction costs per contract in Esmeraldas across transaction components. On average, total market transaction costs per hectare for contracts established with individual landowners are estimated at about $USD 5.53. See Annex B for a distribution of per-hectare market transaction costs for Socio Bosque program in Esmeraldas. The number of individual contracts in Northwest Ecuador adds up to approximately 16 % of total overall individual contracts under Socio Bosque program.

	Costs ($) per contract	Share of total market transaction costs
Information costs	287.0	57%
Negotiation costs	89.5	18%
Monitoring costs	127.3	25%

Table 2.5 Socio Bosque average market transaction costs per individual contracts in Esmeraldas, Ecuador for year 2010.

Reported figures for transaction costs in year 2010 are used to project total program costs across 20-year contract period (See Table 2.6). The Socio Bosque program established 574 individual contracts across the country in 2010. These contracts represent an area equivalent to 67.3 thousand hectares. Areas supported under Socio Bosque are restricted to native, primary or secondary forests in an advance stage of ecological succession.

To calculate total avoided emissions from deforestation, we follow Archard et al.'s 2002 approach. First, we consider existing regional figures of total carbon vegetation biomass derived from the actual biomass density without roots (i.e., 182 t per hectare of aboveground biomass for moist closed forest in Ecuador) (Brown, 1997). These figures can be increased by 20% to account for belowground vegetation (root) biomass, accepting that root biomass varies considerably in tropical forests[xxiii]. Second, carbon can be assumed to be 50% of total biomass (Watson et al., 2000). The resulting computation indicates 109.2[xxiv] tons of carbon (tC) per hectare[xxv]. Therefore, it could be roughly estimated that Socio Bosque is helping to avoid a total of 26.91 million tons of CO_2[xxvi] through 20-year period contracts established with individual landowners.

Transaction costs types		Costs (US $ in millions)
Implementation & Administrative	Information	0.16
	Negotiation	0.05
	Monitoring	0.07
Mitigation Activities	Monitoring (present value for 20 year contract-period)	0.56
Total market transaction costs[+]		0.85
Implementation & Administrative	Information*	0.46
	Negotiation	0.23
	Monitoring **	1.80
Mitigation Activities	Monitoring ***	3.17
Total political transaction costs		5.67
Total projected CO_2 tons for 20 years[+] (in million tons)		26.9
Total transaction cost per ton CO_2 (in $)		0.24
Political transaction costs share (% of total transaction cost)		87
Market transaction costs share (% of total transaction cost)		13

Table 2.6 Projected total Socio Bosque transaction costs types and magnitudes

* Comprises set-up costs for baseline establishment, and promotion. ** Projected present value for 20 year contract-period represents management and financial recurrent costs. *** Projected present value for 20 year contract-period, represents monitoring recurrent costs. [+] Note that market transaction costs for communal contracts are not reported in this study.

+ The baseline scenario is that there would be full deforestation in absence of incentive policy. As it will be shown in Chapter 3, this baseline is empirically observed through a period of 30 years. This essentially means that independently from using a 20 a 30 year period, the idea that in the absence of the incentive-based policy, forest land area is likely to be significantly reduced and, therefore, using the value of total forest carbon stock in standing forest is a consistent approach to conduct this analysis.

2.5 Discussion

To begin the analysis of reported results, it is important to keep in mind two important considerations. First, the two designs explored have different periods of implementation. Transaction costs magnitudes for the CDM-like design takes in to account reported figures for a 10-year period project whereas reported results for the REDD-like design are based upon projections for a program involving a 20-year contract period using reported figures of costs for the second year of full-scale implementation. Therefore, the following discussion will focus on comparisons between designs in reported figures for political transaction costs and on differences among market transaction costs within the REDD-like program. Discussion of costs measures based on projections will be limited to a general discussion on environmental cost-effectiveness of alternative designs.

Second, the expected environmental outcome for both designs shares some commonalities but involves one major critical difference. Although both programs aim at climate change mitigation through the implementation of land-based activities, the service provided is substantially different. In the case of CDM-like design the service involves sequestration of GHG emissions from the atmosphere whereas the REDD-like design is targeted to avoid emissions into the atmosphere from deforestation and forest degradation. This major difference is associated to the potential crediting period and economic value that could be attached to these services. Given that there are not current market specifications yet in place for these potentially different carbon assets, this analysis continues under the assumption that the outcome of both designs may receive equivalent market economic value and, therefore, comments with respect to their transaction cost-effectiveness may be made.

2.5.1 Comparison of magnitudes of transaction costs

Keeping in mind the aforementioned limitations, this study shows major differences on the distribution of political transaction costs for the two designs under examination (See Table 2.7). For political transaction costs associated to activities set-up and launching, the CDM-like project involves a relative magnitude of costs two times greater that the share of costs associated to the REDD-like program. In terms of operational expenditure, however, a relative larger share of budget is allocated to incentives instead of toward political transaction costs in the CDM-like project compared to the REDD-like program. This may indicate the potential for a larger share of costs invested during activities set-up and launching to result in a lower relative magnitude of recurrent political transaction costs during the activity life-cycle. The ultimate importance of this potential linkage, however, would have to keep in to consideration the absolute environmental outcome achieved and the associated costs. Lastly, the observed relative higher share of political transaction costs associated with the REDD-like program can be explained by the bureaucratic process and the associated resources that a centralized, national level scheme entails. It is also relevant to note that political transaction costs may decrease across time. This trend seems to be consistent in both the CDM-like and the REDD-like designs.

Costs Types	PROFAFOR (A/R CDM-like)	Socio Bosque (REDD-like)
Total Costs	- 6% launching costs + 86% design costs - 94% operational expenses + 74% Payments + 26 political transaction costs	- 12% launching costs + 51% design - 78% operational expenses + 64% Payments + 36% Political transaction costs
Political Transaction Costs	- 76% Administration - 13 % Monitoring	Components - *76% implementation and administration* - 24% *mitigation activities*
Market Transaction Costs		- 57% Information - 25% Monitoring
Average set-up Political Transaction Costs per hectare	$ 17	$ 2.64
Annual Political Transaction Costs per hectare	$ 6	$ 2.76
Average set-up Market Transaction Costs per hectare		$ 5.53
Annual Market Transaction Costs per hectare		$1.40
Transaction cost per ton CO_2	Political Transaction Costs - $ 0.58	Total Costs - $ 0.24 Political Transaction Costs - 87% of total Market Transaction Costs - 13% of total

Table 2.7 Comparative measures of transaction costs for reported activities

With respect to market transaction costs associated with contract set-up for the REDD-like program, search or information costs are observed to represent a larger share of total market transaction costs (i.e., about 57%) followed by monitoring and enforcement (i.e., a about 25%). As expected, this relative distribution of costs can be explained by the fact that during contract implementation landowner activities and efforts are mostly focused on collecting all necessary information about the program and fulfilling all required process in order to enter the program. Given that information and search costs were reported to be mostly implicit costs, it would be useful to continue yearly surveys to explore potential changes in this observed distribution of market transaction costs shares.

In addition, it could be argued that there is some cost advantages to the programmatic approach over the project-based scheme reported. One critical difference is observed on fixed-costs processes such as that those associated with monitoring during the implementation and administration component. It seems particularly relevant in the case of the CDM-like project-based approach that it demands a relatively higher initial investment. This is different than the national level programmatic approach, as the fixed costs of implementing the monitoring system are distributed among a larger number of individual contracts and associated area. In fact, average set-up political transaction costs per hectare are substantially higher for the CDM-like project-based design relative to the REDD-like programmatic approach (i.e., about a six-fold difference). Likewise, annual political transaction costs of CDM-like design, PROFAFOR, are more than twice as large relative to those observed for the REDD-like program, Socio Bosque.

With respect to transaction cost-effectiveness, the REDD-like national level design approach seems to pose large advantages relative to the CDM-like project-based approach for achieving real emissions reductions. Total transaction costs for the former are about less than half of the political transaction costs reported for the latter. This is particularly important as political transaction costs per ton of carbon dioxide for the REDD-like design represent about to 87% of total transaction costs. Consequently, based on this conservative scenario and keeping in mind the aforementioned limitations, it could be argue that REDD-like programmatic designs are characterized by lower transaction costs than CDM-like approaches for afforestation and reforestation activities.

Mitigating this cost advantage is the fact that the programmatic approach may face difficult design choices that will affect the trade-off between cost-effectiveness, poverty alleviation objectives, regional equity, and the ability to secure sources of international finance. The current arrangement of the REDD-like program, Socio Bosque, has allowed the rapid, streamlined enrollment of vast areas during a relative short span of the program existence. Some of the same observed characteristics that make the program so attractive for individual landowners, however, may also detract from its environmental integrity. In order to avoid this outcome, it is particularly important to design and implement effective and transparent systems for both monitoring and verification that comply and satisfy international expectations under REDD but also follow the same transaction costs minimization approach that currently characterizes Socio Bosque.

Transaction costs represent an important share of total project costs for both the CDM and the REDD-like initiatives explored in this paper. However, there are some differences in this study's estimates compared with those previously reported by Benitez et al. (2001). Using a benefit-cost analysis of the carbon sequestration potential of afforestation projects and secondary forests in North Western Ecuador, Benitez et al. (2001) estimated transaction costs to be about \$60 to \$80 per hectare for projects with 20-year rotation cycles. Transaction costs were considered to be mainly certification costs borne by the landowners and calculated at annual \$0.5 per hectare and about US\$0.5 of each Certified Emissions Reduction (i.e., one ton of carbon dioxide or equivalent GHGs). Additional differences with reported measures in the literature are presented in Table 2.8.

	Country	Transaction costs type				TOTAL	Source
		Implementation	**Administrative**	**Implementation**	**Administration**		
		($/Ha)		**($/ton CO_2)**			
Socio Bosque (REDD-like)	Ecuador	4.38	1.54			0.24	Ortega-Pacheco et al. (2010)
PROFAFOR (A/R CDM-like)	Ecuador	184	3			1.42	Wunder and Alban (2008)
Full implementation REDD in Amazon	Brazil			0.58			Nepstad et al. (2007).
Forestry offset projects	Ecuador (Global)					1.22 (0.38*)	Antinori and Sathaye (2007)
National level PES in Central and Latin America	Americas				0.01-0.04		Grieg-Gran (2006).
Aggregation of costs						1**	Boucher (2008)
U.S. Conservation Reserve Program (CRP)	United States					1**	Sohngen (2008).

Table 2.8 Summary of transaction costs for land-based mitigation (modified from Olsen and Bishop 2009)

* average min 0.03 and max 1.23; ** This estimate is based on the aggregation of sub-sets of implementation and transaction costs from a range of studies: Antinori and Sathaye's (2007) average estimate, Nepstad et al.'s (2007) implementation cost estimate - including project and national level costs- and Grieg- Gran's (2006) highest administrative cost estimate - includes Costa Rica, Mexico, and Ecuador.

2.6 Conclusions

2.6.1 The role of transaction costs

The purpose of this paper is to provide empirical measures of land-based mitigation activities in Ecuador. Using primary and secondary data, we have presented transaction costs types and magnitudes associated to alternative designs of land-based mitigation actions. This study has identified preliminary differences between transaction costs affecting project and programmatic approaches in a developing country context. Although it is difficult to reach definite conclusions on the likely impact of transaction costs, the above discussion shows that there are a relatively large magnitudes and differences among types of transaction costs across transaction components. In fact, the results point to a likely reduction across time of costs facing implementation and administration component in both design explored. Decrease in implementation and administrative costs may be explained by a combined effect of learning both on administrator and landowners. One would expect to observe the same behavior of changes in costs on approaches sharing similar design characteristics.

This study shows that it may be a useful practice to identify who incurs transaction costs as a result of specific design characteristics as it can lead to costs reductions over time. A call of caution is in order, however. Design choices may not necessarily lead to a cost reduction strategy but to a redistribution of the transaction costs across transaction components or from one transacting party to another. This fact points to the relatively important notion emphasized at the beginning of this paper for the need to consider overall transaction costs measurement is mitigation policy evaluation.

The results presented in this study results may help to identify specific institutions that can lead to reductions in transaction costs and to raise the cost-effectiveness of mitigation actions. This can be particularly relevant considering the current policy need in the context of international climate negotiations. With regard to information, high agency costs for developing and making available information on land use practices for conservation, both under CDM-like PROFAFOR and REDD-like Socio Bosque, reduces individual transaction costs on gathering such data and increases the possibility for achieving expected outcomes. Negotiation costs can be reduced by observing standardized and simplified procedures for contracting and bargaining, which in turn facilitate the understanding of the incentive-conditionality scheme and time-length of commitments, and decentralizes the signing process under the REDD-like program, Socio Bosque. Well-identified operational entities and trained staff can reduce costs for individual contracts and raise the level of confidence between transacting parties. Improved monitoring, centralizing technical, legal and economic expertise and standardizing data collection and reporting methodologies developed and implemented by administrators reduces mitigation activities costs for landowners and assures information reliability for international reviews and potential funding sources.

In addition, the land-based mitigation actions explored in this study are relatively new. Therefore, it is not clear whether results can lead to general conclusions in terms of negotiation and enforcement costs. In particular for the REDD-like program, Socio Bosque, it is yet to be tested whether contract conditions can be legally enforced, and at what cost to the project managers across time.

2.6.2 Future research

As transaction costs could raise the total costs of generating land-based carbon credits and keeping in mind the possibility for international registry systems, future research should direct efforts to generating instruments that could lead to streamlining robust measurement methods. Moreover, note that this research has not reported measures of transaction costs associated to the third identified component of this transaction, *Credit Trading.* As these costs may be mostly implicit and represent a large share of transaction costs for other activities in the financial sector such as futures and commodity trading, future efforts may attempt to address the need for empirical measures of these transaction costs, their sources and magnitudes, and how they could affect overall cost-effectiveness of land-based mitigation activities and climate policy. Finally, as discussed above, measures of transaction costs reported in this study do not incorporate market transaction costs for communal contracts. Although they may be important given that such contracts cover about 88% of total area under the program, a thorough analysis considering communal tenure regimes and group decision-making goes beyond the scope of this paper.

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Chapter 3: Land use competition and the effect of transactions costs on land-based mitigation activities in a developing country

3.1 Introduction

Reducing GHG emissions – particularly from fossil fuel combustion – is the focus of domestic and international policies to reduce the risks of anthropogenic climate change. Emissions trading programs are likely to be the central incentive-based policy tool used to accomplish this task in developed nations. These programs have the capacity to bring about greenhouse gas (GHG) reductions in a highly efficient way – but for ambitious targets they will still impose substantial economic costs. A number of studies have now suggested that land-based carbon credits can reduce the costs of meeting stringent GHG goals (e.g., Sohngen and Mendelsohn, 2003; Tavoni et al., 2007, Nabuurs et al, 2007; Kindermann et al., 2008). Many of the credits that these studies anticipate are derived from actions undertaken in developing countries.

Two critical concerns about the cost estimates in these studies have arisen. First, the individual studies described above assume that international credits can be costlessly generated, however, it has been noted that transactions costs could raise the total costs of generating land-based carbon credits, such as through reductions in deforestation or forest degradation (see Antinori and Sathaye, 2007). Second, many large-scale modeling studies used to estimate costs of land-based credits have made critical assumptions about the elasticity of land supply in developing countries without having solid empirical evidence. Specifically, some studies (e.g., Sohngen and Mendelsohn, 2003) have used elasticity estimates from empirical studies in the United States, and the authors have transferred those to elasticity estimates to other regions. As shown in Sohngen and Mendelsohn (2007), assumptions about land supply can have large effects on the resulting supply of carbon credits available.

The importance of estimating transaction costs and the elasticity of land supply cannot be overstated. Tavoni et al. (2007) suggest that reductions in deforestation can reduce costs of meeting stringent carbon targets by 40% in coming years. These estimates, though, do not account for transactions costs and they use the same land supply elasticity for all regions of the world. This paper uses an econometric model to estimate the elasticity of land supply in one developing country, Ecuador. It also evaluates the effect that transaction costs have on land-based mitigation activities based on new research.

Ecuador is chosen for a number of substantive and practical reasons. Substantively, the implementation of this research in northwestern Ecuador is particularly important because: i) the large potential for biofuels production given agroclimatic conditions, the relative amount of land area available for conversion that could be linked to deforestation, existing processing capacity and excess supply of vegetable oil (ECLAC and GTZ 2004; Ludena, Razo, and Saucedo 2007; González 2007), and ii) the significant institutional reforms implemented and the on-going capacity building towards climate change mitigation (MMRREE 2007) and energy self-sufficiency through biofuels (Jull et al. 2007; Pelaez-Samaniego et al. 2007; República del Ecuador 2004, 2007, 2007; Rothkopf 2007) as well as the National Strategy for Reducing Emissions from Deforestation and Forest Degradation and the on-going national incentive-based program for forest conservation - Socio Bosque (MAE 2010). Practically, first, it contains a number of eco-zones, ranging from tropical lowland forests to dry highland forests offering sources for variability for our model. Second, there exist good data sources for estimating spatially explicit models of land use change.

Our methodology will build upon earlier work by Plantinga et al. (1999) and Sohngen and Brown (2006). These earlier efforts have developed land supply functions for the United States, using either county level data or more specific plot level data. This analysis will use similar methods, but focus on the Northwestern region of Ecuador. Other studies have examined land use change in the Amazon basin (e.g., Nelson and Hellerstein, 1997, and Pfaff, 1998), but these studies were conducted at a lower level of resolution than we propose, and they did not estimate land supply elasticity.

Similarly, based on maps (biophysical factors) and census data (land use data and socio-economic factors) previous studies in Ecuador have explored land use change dynamics at different levels of specificity, determined land use drivers and land use patterns, and considered interdependent effects on food production, natural resources and provision of environmental services (e.g., de Koning et al. 1998; de Koning et al. 1999a;

de Koning et al. 2003; de Koning et al. 1999b; Overmars et al. 2003). These previous studies offer a useful point of departure for our modeling of land use change, however, they did not estimate models that can be used to calculate land supply elasticity, and do not account for transaction costs as does this paper.

Although the international context seems to indicate that greater resources will be available to match overlapping climate policy demands, policymakers in developing countries are left to confront essential yet difficult policy choices. One key issue is the extent of competition with respect to the use of land between mitigation options such as crop production and land-based carbon mitigation. Using Ecuador as case study, this paper presents a spatially explicit land use model that can be used to examine the interdependencies arising at the intersection of policy options for land use changes (i.e., deforestation and agriculture) and climate change mitigation. The land use model generated from this research may help establish whether or not policy alternatives and the associated transactions costs in developing countries may have a value in mitigating greenhouse gases (GHG) concentrations. By estimating changes in forest land use shares, projections are used to assess the implications that shifts on land use shares may have on baseline carbon storage in the region. Finally, the land-use share and simulation model is used to examine the types of incentives that could be used to maintain forest lands in this region.

3.2 Empirical land use share model

The econometric model estimates the proportion of land in two different land uses. The central conclusion reached in theoretical economic analyses of land use change is that land use patterns are determined by relative rents, relevant policy variables, and land characteristics such as soil fertility (Miller and Plantinga 1999). This result is also supported by empirical studies exploring main determinants of land use change in Ecuador (see Sierra 2001 and Sierra, Stalling 1998, Southgate et al. 2000, de Koning et al. 1998; de Koning et al. 1999a; de Koning et al. 1999b). This paper attempts to use a land-use share model, following Hardie and Parks (1997) and Sohngen and Brown (2006), to examine the mix of agricultural and forest land in a province region of the Northwestern Ecuador: Esmeraldas. Given that we will only be considering two land uses, we adopt a logit model specification. Unlike most previous logit models exploring the conversion of agricultural land to forestry in the United States (i.e., Hardie and Parks, 1997; Plantinga et al., 1999; Ahn et al., 2000), this paper focuses explicitly on natural occurring forest land conversion into agriculture in a tropical developing country context. Moreover, differently from Sohngen and Brown's (2006) multinomial logit model, the proposed research does not disaggregate land use into types. Sohngen and Brown's (2006) model analyzed different types of cultivated forest whereas we propose to study aggregated agriculture land shares into one single management type (i.e. African palm) and compare it to natural occurring forest (i.e. primary and secondary forest). Note that the limitation to only two types of land uses is critical for our decision to adopt a logit model.

The proportion of land in one of these uses (i.e., forest and agriculture) in each parish is expressed as a binomial logistic function with explanatory variables[xxvii]. According to previous work, land use shares in the selected region can be estimated to be a function of explanatory variables including proxies for land rents from alternative uses, and institutional (i.e., tenure formalization, immigration rate and population density), and bio-physical factors (i.e., soil agricultural potential). Following Maddala (1983), the functional form for the binomial logit can be expressed as,

$$
\begin{aligned}
y &= \frac{1}{1+e^{-f(X)}}, i.e., \\
P_i &= \frac{1}{1+e^{-(\beta_0+\beta_1 x_{1,i}+\beta_2 x_{2,i}+\cdots+\beta_k x_{k,i})}}
\end{aligned}
\qquad (1)
$$

The left-hand side of Eq. (1) is the proportion of land allocated to usage *j*. *X* is the vector of independent variables and *β* is the vector of coefficients to be estimated. Under the assumption that P_j is distributed as a generalized extreme value distribution, the log-odds ratio (the ratio of P_j/P_m, for example) can be derived as a linear function of the parameters and expressed as follows:

$$
\ln\left(\frac{p_j}{p_m}\right) = (\beta_j - \beta_m)X \qquad (2)
$$

As noted in Hardie and Parks (1997), and Plantinga et al. (1999), parameter estimates for β_j can be obtained by setting β_m =0, and assuming that the errors are normally and identically distributed. Thus, Eq. (1) can be transformed into a linear form with two different land uses (j and m) and expressed as follows:

$$L(P_j) = \ln(\frac{P_j}{1-P_j}) = \beta_0 + \beta_1 x_{1,j} + \beta_2 x_{2,j} + \cdots + \beta_k x_{k,j}$$

$$\ln\left(\frac{p_j}{p_m}\right) = \beta X + u_i \tag{3}$$

For the land uses proportion considered in our model, the specific equation estimated is:

$$\ln\left(\frac{Forest_i}{Agriculture_i}\right) = \beta_{0Forest} + \beta_{1Forest} X_{1i} + \beta_{2Forest} X_{2i} + \ldots\ldots + \varepsilon_{ForestAgriculturei} \tag{4}$$

where $Forest_i$: share of land in natural occurring forest; $Agriculture_i$: share of land in agricultural (aggregated cropland dedicated to African Palm cultivation only) uses; X_i: independent explanatory variables indexed to parish i such as agriculture and forest extraction rents, soil agricultural potential, population density, immigration rate and tenure; β is a vector of unknown parameters to be estimated; ε : normally distributed, i.i.d error terms.

With the parameter estimates, the proportion of land allocated to the two land uses can be predicted for each unit of observation (parishes in our case). The model can also be used to project future land uses by changing the vector *Xi*. For instances, future rental values resulting from the introduction of direct payments to incentivize forest conservation can be projected, and used to predict the resulting area of land allocated to forest and agricultural land uses.

3.3 Data Sources

Data for this model were drawn from several sources. Data on the proportion of cropland from agricultural production units were obtained from the Ecuador Census of Agriculture (Censo Nacional Agropecuario - CNA). This data was collected during the period 1999 – 2000, and provides information on numerous attributes for the agricultural production units (e.g., land use, tenure, and agricultural yields) (INEC-MAG-SICA, 2003). Data is aggregated into parishes and there are 62 parishes in the province of Esmeraldas. The CNA survey data was originally collected at farm level, thus it is factored using their original expansion factors. Data on forest shares were drawn from land use maps developed by the Center for Integrated Inventory of Natural Resources (CLIRSEN, 2008).

A number of additional variables were also collected for the analysis. All of the variables used and the sources for the data are presented in Table 3.1.Data on agricultural rents will be derived from the Census of Agriculture. While the Census did not specifically measure land values or rents, it did measure inputs and outputs from the agricultural production units. These variables can be used to estimate net returns per hectare of land. Agricultural rents per hectare of land will be estimated as Net Present Value. Annual net returns values for African Palm cultivation can be imputed using harvested quantity and prices. While the Census did not specifically measure land values or rents, it did elicit inputs and outputs values from the agricultural production units (i.e., farm level). Thus, harvested quantity in each farm for each crop is available from the Agricultural Census data set. Annual average farm-gate prices (i.e., output prices) have been collected from the Ministry of Agriculture. To estimate production costs, information was assessed from regional budgets from Ministry of Agriculture. Although total quantity of labor and agricultural equipment is available from the Agricultural Census Data Set, we do not incorporate them directly and rather follow budget allocations per hectare of production in order to avoid making major assumptions on how inputs were allocated within each farm. Agricultural wages were drawn from government statistics. The budget approach also allows us to include information from the mix of pesticides and fertilizers to calculate variable costs, which is information not available from the Agricultural Census. This is particularly important in the case of a commercial crop such as African Palm because costs associated with pesticides and fertilizers could represent a significant share of total variable costs.

Variable	Description
AgRent	Rental values for cropland (estimated from net present value analysis)
ForestRent	Rental values for forestland
Soil	Soil agricultural potential
PopD	Population density (Ecuador Population and Household Census)
ImRate	Immigration rate (Ecuador Population and Household Census)
Tenure	Dummy variable representing parishes with aggregated measure for land with and without title

Table 3.1 Variables used in the regression analysis

Estimating forest rents for natural occurring tropical forest is a difficult task. Previous studies on land use change conversion have mostly look at cultivated forest in the United States. Net present values for each timber type, site class, and price region were calculated by Sohngen and Brown (2006) using the Faustmann formula, adjusted for management. Annual rents were then imputed using the interest rate for timber investments. Rental values for major forest types in each county were estimated as a weighted average across current site classes. In this study, we follow Wunder's (2000) finding that deforestation is driven by farmer's adoption of land myopic strategies focused on short-run returns (i.e., wood extraction) as source of basic capital accumulation, which are subsequently reinvested in other activities (i.e., agriculture)[xxviii].

Following Sierra (2001) who suggests that a large proportion of total harvests in Ecuador are locally consumed (>50%), forest rents per hectare will be calculated as:

$$ForestRents_i = Roundwood \; * P_i^s, i : 1 \ldots . n$$

Where, ***ForestRents_i*** indicates rental value per hectare of forest land in parish *i*, ***Roundwood_q*** is the average measure of roundwood per hectare of tropical forest in the region*, and* P_i^S the stumpage price for parish *i.*

The stumpage price, P_i^S , can further be broken down as,

$$P^S = (P^L - C^h - C^m - C^t - C^k)*\eta$$

where, P^L is the end product price of the lumber, C^h are harvesting costs, C^m are milling costs, C^t are transportation costs, and C^k are marketing costs. *η is processing efficiency rate.* Data for estimating forest rents per hectare has been drawn from Benitez (2005). This info has been supplemented from government statistics, peer-reviewed publications and interviews with local forestry experts.

Data on agricultural soil types can be drawn from basic thematic cartography and integrated geographic information data sets provided for this research initiative by the Center for Integrated Inventory of Natural Resources and the Geographic Information System Unit of the Ministry of Agriculture (CLIRSEN, 2008; and SIG-Agro, 2008). Data on population density and immigration rate for each parish is obtained from 2001[xxix] Population and Household Census of Ecuador (CPV – Censo de Población y Vivienda) (INEC 2001). Soil agricultural potential is extracted from SIG-Agro (2008) agricultural potential map. The 18 types of agricultural potential are set to value range from 0 to 10 based on their agricultural limitation, texture, slope, and irrigation condition. Land tenure status is classified into two groups, identified by dummy variables of 0, 1, indicating whether or not landowners had tenure or not. Four types of tenure statuses as originally collected in CAN (i.e., land ownership with officially registered title, renter, tenant, communal or group ownership), are given a value of 0. Alternatively, tenure statuses originally coded as land occupied without title, and other, are assigned a value of 1.

3.4 Econometric results

The results of the econometric analysis, based on 62 observations, are shown in Table 3.2. Many of the parameters are significant at the 1% or 5% level. As expected, higher rental values for forest land decreases the proportion of land devoted to the activity because they increase deforestation and land conversion. Higher cropland rents increase the proportion of land devoted to agriculture. Parishes with a higher soil agricultural potential have a lower proportion of forestland, as do parishes with a higher immigration rate.

	Estimate[+]	**Std. Error**	**t value**	**Pr(>|t|)**	
(Intercept)	17.8987	5.1051	3.5061	0.0009	***
AgRent	-0.0035	0.0013	-2.7332	0.0084	**
ForestRent	-0.0011	0.0005	-2.0574	0.0444	*
Soil	-0.3669	0.1010	-3.6326	0.0006	***
PopD	0.0068	0.0016	4.3355	0.0001	***
ImRate	-1.2042	0.4009	-3.0034	0.0040	**
Tenure	1.6221	0.9474	1.7123	0.0925	.

Table 3.2 Parameter estimates of econometric land use model
Significance: '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
[+] White's estimator is used to adjust the estimated coefficient standard errors using a variance–covariance matrix addressing heteroskedasticity (Zeileis 2004).

Population density and tenure title have a positive effect on the proportion of forestland relative to cropland. This indicates that parishes with higher population density and tenure title have a higher proportion of forestland. Moreover, the average proportion of land dedicated to forest in the region is estimated to be about six times the area relative to that dedicated to agriculture in the baseline period. In general, the model projects proportions of land area that are consistent with the observed pattern.

3.5 Forest area and carbon projections

Projections of future land uses are made by adjusting the rental rates for future time periods, and re-projecting the area of land in alternative land uses. During 2000-2005, the deforestation rate in Esmeraldas Province as a whole increased up to 7.3%, and deforested land was mostly converted to African Palm cultivation (Ortega-Pacheco and Jiang, 2010). Based on field observations, forest in the region can be described as native, primary or secondary forest, rich in wood resources. Wood extraction is a highly labor-intensive activity. During the forest conversion cycle, the initial phase of land-use change (i.e., forest clearing) on a one-hectare plot over a one-year period is assumed to be dedicated to wood extraction. For this study we assume that the processing efficiency continues to improve, and that prices rise at relatively modest rates so that forest land rental rates increase at 1% per year.

By contrast, African Palm rents as proxy for agricultural rents are assumed to increase at 2.0% per year. While crop yields for African Palm in this region have remained relatively stable according to the National Association of African Palm Growers and Processors (ANCUPA) 2005 Census, input prices have risen due to dollarization by more than 4-5%. Similarly, market prices have steadily risen about 1–2% per year. This is mainly due to increased world demand on vegetable oils for biodiesel production. Consequently overall returns to growing crops in this region have risen over the period 2000–2010. For this analysis, therefore, we assume that African Palm rental values are rising at least 2.0% per year over the next 20 years.

Table 3.3 presents the projected land areas for forest and agriculture and the change relative to the baseline projected value for the years 2010, and 2020, and 2030. Because this study only considers aboveground carbon storage, and ignores soil carbon sequestration, agricultural land areas and carbon storage on agricultural lands are not shown in Table 3.3. The model predicts land use proportions, so total land in forest and agriculture remains constant. The results suggest that large areas of land are expected to convert to agricultural activities associated with African Palm cultivation over the next 20 years, rising from 1,338 million ha in year 2000 to about 10,168 million ha by 2030. Forest land use is projected to decline during the period. Results in Table 3.3 indicate that more forestland shifts to agriculture in the future than in the past. This is an indication that the existing incentives strongly favor conversion. Overall, results denote the high extent of competition with respect to the use of land between mitigation options such as crops production for biofuels processing and land-based carbon mitigation through avoided emissions from deforestation.

To calculate total emissions from forest land conversion, we follow Archard et al.'s 2002 approach. First, we consider existing regional figures of total carbon vegetation biomass derived from the actual biomass density without roots (i.e., 182 t per hectare of aboveground biomass for moist closed forest in Ecuador) (Brown, 1997). These figures can be added 20% for belowground vegetation (root) biomass, accepting that root biomass varies considerably in tropical forests[xxx]. Second, carbon can be assumed to be 50% of total biomass (Watson et al., 2000). Under these assumptions, carbon stock projections for the baseline case are shown in Table 3.3.The resulting computation indicates 109.2[xxxi] tons of carbon (tC) per hectare[xxxii]. Therefore, it could be roughly estimated that a total of 352.92 Tg of CO_2[xxxiii] are currently in the form of a forest sink. If crediting systems eventually emerge to provide credit for avoided emissions from deforestation, these avoided emissions could be used as credits.

	2000	2010	2020	2030	Average annual change
Forestland area (in million hectares)	0.88	0.02	0.0001	0.0000003	0.03
Carbon stock in forests (in Tg CO_2)*	352.9	9.3	0.03	0.0001	11.76

Table 3.3 Forest area and carbon stocks
* (million tonnes carbon by the year given; 1 tonne=1 Mg=10^6 g; 1 Tg=10^6 Mg)

3.5.1 Sensitivity and policy analysis

Alternative scenarios of future expected land rental rates can also be considered. Following the Von Thünen approach, forest rents are determined by the opportunity costs of keeping land in forest. Therefore, the reported measure of forest rent also indicates "conservation opportunity costs", since it represents the minimum income stream that one would have to come up with to offer the landowner economically competitive alternatives to forest conversion. It is possible that conservation payments for forest land can be introduced. The following analysis rest under the assumption that the effect of introducing incentives for forest conservation (or avoided emissions) affects the forest land rent associated with logging (deforestation) activities. This analysis is particularly different from previous studies as much of the research focus has been allocated to understanding changing rents on timber land (plantation-based) whereas in this case the forest at hand is naturally occurring tropical forest which rents are associated to deforestation. Therefore, any incentives to keep forest standing would have to be subtracted from the potential rents of deforestation so that, when conservation flows equate the opportunity costs of deforestation, landowners may decide to keep standing forest and, consequently, keep the carbon stock available.

In fact, Ecuador has recently implemented a programmatic approach, Socio Bosque, that could serve as a model for domestic implementation of mitigation actions under United Nations framework for Reduced Emissions from Deforestation and Forest Degradation in Developing Countries (REDD). Socio Bosque is based on a single conditional payment covering multiple environmental services. The amount to be transferred per hectare per year depends on the area an individual landowner or community desires to conserve. For areas smaller than 50 hectares, the payment is $30 per hectare per year. The payments per hectare decrease for larger areas and are set to change across time. For the purpose of this study, a scenario where payments are assumed to grow across the 30-year analysis period is examined. The set of payments for conserving forest land examined is $30 per ha per year for 2000–2010, $40 per ha per year for 2010–2020, and $50 per ha per year for 2020–2030. The payments are set to grow over time given that the opportunity costs associated with maintaining forestland versus shifting land to agriculture rises as African Palm rental rates may raise.

The second policy considered assumes the same size conservation payments, but also deducts transaction costs of program implementation, administration and activities across 30-year analysis period. The variation in the relative shares of costs faced by the government is partly related to the nature of the program, but also reflects the stage of its development (Dorward, 2001). Land-based mitigation actions may require fixed cost implementation-type activities in their firsts years, as the details of implementation are finalized and the program is set up; administration-related transactional activities then may rise in relative importance. Here we allow changes over time in the mix of administrative activities linked to the time profile of take-up. In particular, we posit that at some point the balance will switch from "implementation or set-up" activities (such as promoting the program and entering into contracts) to more routine "maintenance" activities (like those related to checking compliance with contract terms). The set of transaction costs affecting programmatic approach is \$5.53 per hectare per year. Transaction costs are set to diminish at a rate of 1 percent per year relative to previous year. Transaction costs information used in this study are consistent with those empirically measured for the same study area by Ortega-Pacheco et al. (2010).

The third policy changes the size of payments. Here the structure of incentives simulates a REDD payment. It assumes a potential value of a REDD credit per ton of CO_2 avoided equal to a reference market value for sequestered emissions (i.e., \$25 per ton). Thus, the payment per hectare will be equal to the market value for the total amount of ton of CO_2. For simplicity, in this policy scenario, transaction costs of program implementation, administration and activities across 30-year analysis period are the same as those used in the second scenario.

	Baseline	Without conservation payments	Conservation payments only	Conservation payments and transaction costs	REDD payments and transaction costs
Forestland area (in thousand hectares)	883.03	0.0000002	0.0000006 (95.71%)*	0.0000004 (94.37%)	1,011.19
Carbon stock in forests (in thousand ton CO_2)	352,921	0.00001	0.002	0.0018	404,143.9
Avoided emissions (in thousand ton CO_2)			0.0022	0.017	404,143.9

Table 3.4 Changes in forest area and carbon stock between 2000 and 2030 under alternative scenarios.
* Quantities between parentheses indicate percentage change due to the policy (or lack of policy). There is also a -31.31% reduction on forest area due to consideration of policy transaction costs.

The results in Table 3.4 show that conservation payments currently in place are not sufficient to assure forest land use area. In fact, any potential positive effect that could be associated with the introduction of conservation payments is outweighed by the likely trend of steady growth assumed for African Palm rental rates across time. This finding points to a need for conservation payments schemes in the study region to attempt to equate at least the "conservation opportunity costs" otherwise they would not necessarily offer the landowner an economically competitive alternative to forest conversion.

Despite this fact, the results from the scenario show the negative effect of transaction costs on the incentive scheme. Transaction costs seem to reduce incentives to keep forest land use and allow associated additional losses from carbon storage. Thus, it could be argued that accounting for transaction costs could help better assess the cost-effectiveness of institutional arrangements attempting to introduce incentives for avoiding emissions from deforestation. As a matter of fact, the introduction of a larger payment based on REDD credits being valued more highly denotes the substantial difference that can be expected on forest land conservation.

3.6 Conclusions

Land-based mitigation actions undertaken in developing countries may generate carbon credits that can help reduce the costs of meeting stringent GHG goals. This paper reports empirical measures that address two critical concerns over the potential that land-based credits may have for climate policy. First, a logit share model is used to produce a model predicting the share of land in forest land and agricultural land in Northwestern

Ecuador. Second, using the estimated model this paper evaluates the effect that different levels of rents have on the allocation of land across land uses as well as the impact that conservation payments and associated transactions costs seem to have on land-based mitigation activities. In general, results denotes the high extent of competition with respect to the use of land between mitigation options such as crops production for biofuels processing and land-based carbon mitigation through avoided emissions from deforestation in Northwestern Ecuador. The net effect of transaction costs is negative, reducing the scale of incentives to stop land conversion into agriculture as well as environmental benefits from emissions avoidance by about 25%.

Using this paper's empirical model parameter estimates, projections and sensitivity and policy analysis demonstrate that emissions from forest sinks can be reduced if conservation payments are introduced. However, the results show that for the incentives produce a considerable effect, the incentive would have to be raised so as to match or surpass the opportunity costs for forest land conversion. This paper's discussion indicates that if we were to hold forests constant throughout the 30-year projection period so as to realize potential environmental benefits from emissions avoidance, payments required may be equivalent to or higher than $1,733.76 per ha. Holding the area of forest land constant through the projection period reduces the CO_2 emissions in about 11.76 Tg per year, suggesting that the incentives could have an environmental benefit. In fact, the results reached by simulating the introduction of REDD payments provide a benchmark for how an actual REDD program might positively affect incentives.

Finally, these results raise an interesting issue regarding the storage of carbon on the landscape and the extent of competition and complementarities regarding crediting schemes for net emissions from forest to agricultural conversion associated to biofuels crops. Currently, the Kyoto Protocol rules only consider credits for additional storage of carbon on the landscape, without considering emissions avoidance from forest sinks or net emissions from land conversion. Our sensitivity analysis on elasticity of land supply suggests that the results are highly sensitive to the estimated rental rates. These results illustrate the trade-offs that could arise when designing policies to enhance terrestrial sequestration. If only avoided emissions are credited or if net emissions from forest to agricultural conversion are credited, then incentives for establishment of cropland for biofuels processing can be a useful tool for enhancing carbon storage and emissions avoidance. If credits are also provided for emission offsets in the energy or transportation sectors (i.e., fuel - biofuel substitution), the analysis suggests that in the short term, there would be incentives to expand the stock of African Palm and, thus, additional incentives targeting net emissions levels would be increasingly important if holding forests constant is a goal. It is beyond the scope of this paper to conduct a full life-cycle analysis of energy uses during harvesting, transportation, and processing biodiesel products, however. Additional research would is needed to assess the full net mitigation potential of these alternative scenarios.

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Conclusions

This dissertation generated knowledge to enrich the scholarly debate on policy diffusion and climate change policy, and provides critical insights for policymakers interested in incentive-based institutional arrangements for the provision of environmental services. At the center of the analysis reported through these three chapters is the importance of transaction costs, their measurement and the effect they could have in determining: 1) the extent of adoption of incentive-based schemes for the provision of watershed services in the developing world, and 2) the supply of mitigation services associated with avoided emissions from deforestation, particularly in a developing country context.

The results of the first chapter provide preliminary evidence that the increasing adoption of incentive-based programs for provision of watershed services in the developing world can be explained through the lens of competing theories of diffusion. Using country-level data and drawing from field of institutional economics and transaction costs theory, I argued that the increased adoption over time and space of PES programs can be interpreted as diffusion of interdependent induced institutional innovations. The process of interdependent institutional innovation seems to be induced by varying levels in resource and cultural endowments and technology across countries. These factors reduce transaction costs, increase social demand for improved water quality, and enable PES adoption. Although this paper provides insights for public managers and researchers interested in market-based administrative alternatives for water resources conservation and development, there are major limitations that prohibit an accurate assessment of the relative explanatory power of competing diffusion theories for explaining adoption of local PES programs.

The second chapter provides empirical measures of transaction costs types and magnitudes associated with alternative designs of land-based mitigation actions. This research has identified preliminary differences between transaction costs affecting project and programmatic approaches in a developing country context. Although it is difficult to reach definite conclusions on the likely impact of transaction costs, the discussion shows that there are a relatively large magnitudes and differences among types of transaction costs across transaction components. In fact, the results point to a likely reduction across time inr rhw costs of implementation and administration component in both policy designs that I investigated. Decreases in implementation and administrative costs may be explained by learning effects on both administrators and landowners. One would expect to observe the same behavior of changes in costs for approaches sharing similar design characteristics.

My results support the argument that it may be a useful practice to identify who incurs transaction costs as a result of specific design characteristics, as it can lead to cost reductions over time. The results presented in this study may also help to identify specific institutions that can lead to reductions in transaction costs and to raise the cost-effectiveness of mitigation actions. This is particularly relevant for the policy debate taking place in the context of international climate negotiations. With regard to information, high agency costs for developing and making available information on land use practices for conservation reduce individual transaction costs on gathering such data and increase the probability of achieving expected outcomes. Negotiation costs can be reduced by observing standardized and simplified procedures for contracting and bargaining, which in turn facilitate the understanding of the incentive-conditionality scheme and time-length of commitments, and decentralizes the contracting process. Well-identified operational entities and trained staff can reduce costs for individual contracts and raise the level of confidence between transacting parties. Improved monitoring, centralizing technical, legal and economic expertise and standardized data collection and reporting methodologies developed and implemented by administrators all reduce mitigation costs for landowners and assure information reliability for international reviews and potential funding sources.

Chapter Three reports empirical measures that address two critical concerns for the potential that land-based credits may have for climate policy. First, a logit share model is used to estimate a model predicting the share of land in forest and agricultural in Northwestern Ecuador. Second, using the estimated model, I evaluate the effect that different levels of rents have on the allocation of land across land uses as well as the impact that conservation payments and associated transaction costs have on land-based mitigation activities. In general, the results show a high extent of competition for the use of land between crop production for biofuels processing and land-based carbon mitigation through avoided emissions from deforestation in Northwestern Ecuador. The net effect of transaction costs is negative, reducing the scale of incentives to stop land conversion into agriculture, as well as environmental benefits from emissions avoidance, by about 25%.

Using this chapter's empirical model parameter estimates, projections and sensitivity and policy analysis demonstrate that emissions from forest sinks can be reduced if conservation payments are introduced. However, the results show that for the policy to have a significant effect, the incentive would have to be raised enough to match or surpass the opportunity costs for forest land conversion. In fact, the results reached by simulating the introduction of Reducing Emissions from Deforestation Forest Degradation (REDD) payments provide a benchmark for how an actual REDD program might positively affect incentives. These results illustrate the trade-offs that could arise when designing policies to enhance terrestrial sequestration. Additional research is needed to assess the full net mitigation potential of alternative policy scenarios.

Appendix A: List of countries included in the analysis of Chapter 1.

Country	Region	Country	Region
Afghanistan	Asia	Haiti	Caribbean
Armenia	Asia	Jamaica	Caribbean
Azerbaijan	Asia	St. Lucia	Caribbean
Bangladesh	Asia	St. Vincent and the Grenadines	Caribbean
Bhutan	Asia	Argentina	Latin America
Cambodia	Asia	Belize	Latin America
China	Asia	Bolivia	Latin America
Fiji	Asia	Brazil	Latin America
Georgia	Asia	Chile	Latin America
India	Asia	Colombia	Latin America
Indonesia	Asia	Costa Rica	Latin America
Iran	Asia	Ecuador	Latin America
Iraq	Asia	El Salvador	Latin America
Jordan	Asia	Guatemala	Latin America
Kiribati	Asia	Guyana	Latin America
Korea	Asia	Honduras	Latin America
Kyrgyzstan	Asia	Mexico	Latin America
Laos	Asia	Nicaragua	Latin America
Lebanon	Asia	Panama	Latin America
Malaysia	Asia	Paraguay	Latin America
Mongolia	Asia	Peru	Latin America
Myanmar (Burma)	Asia	Suriname	Latin America
Nepal	Asia	Uruguay	Latin America
Pakistan	Asia	Venezuela	Latin America
Papua New Guinea	Asia	Algeria	North Africa
Philippines	Asia	Egypt	North Africa
Solomon Islands	Asia	Libya	North Africa
Sri Lanka	Asia	Morocco	North Africa

Syria	Asia	Tunisia	North Africa
Tajikistan	Asia	Maldives	Pacific
Thailand	Asia	Tonga	Pacific
Turkmenistan	Asia	Western Samoa	Pacific
Uzbekistan	Asia	Angola	Sub Saharan Africa
Vanuatu	Asia	Benin	Sub Saharan Africa
Vietnam	Asia	Botswana	Sub Saharan Africa
Yemen	Asia	Burkina Faso	Sub Saharan Africa
Cuba	Caribbean	Burundi	Sub Saharan Africa
Dominica	Caribbean	Cameroon	Sub Saharan Africa
Dominican Republic	Caribbean	Cape Verde	Sub Saharan Africa
Grenada	Caribbean	Central African Republic	Sub Saharan Africa

A

Country	Region
Chad	Sub Saharan Africa
Comoros	Sub Saharan Africa
Congo	Sub Saharan Africa
Djibouti	Sub Saharan Africa
Eritrea	Sub Saharan Africa
Ethiopia	Sub Saharan Africa
Gabon	Sub Saharan Africa
Gambia, The	Sub Saharan Africa
Ghana	Sub Saharan Africa
Guinea	Sub Saharan Africa
Guinea-Bissau	Sub Saharan Africa
Ivory Coast	Sub Saharan Africa
Kenya	Sub Saharan Africa
Lesotho	Sub Saharan Africa
Liberia	Sub Saharan Africa
Madagascar	Sub Saharan Africa
Malawi	Sub Saharan Africa

Mali	Sub Saharan Africa
Mauritania	Sub Saharan Africa
Mauritius	Sub Saharan Africa
Mozambique	Sub Saharan Africa
Namibia	Sub Saharan Africa
Niger	Sub Saharan Africa
Nigeria	Sub Saharan Africa
Rwanda	Sub Saharan Africa
Sao Tome and Principe	Sub Saharan Africa
Senegal	Sub Saharan Africa
Seychelles	Sub Saharan Africa
Sierra Leone	Sub Saharan Africa
Somalia	Sub Saharan Africa
South Africa	Sub Saharan Africa
Sudan	Sub Saharan Africa
Swaziland	Sub Saharan Africa
Tanzania, United Republic of	Sub Saharan Africa
Togo	Sub Saharan Africa
Uganda	Sub Saharan Africa
Zaire	Sub Saharan Africa
Zambia	Sub Saharan Africa
Zimbabwe	Sub Saharan Africa

B

Table A List of countries included in the analysis of Chapter 1.

Appendix B: Distribution of per-hectare market transaction costs in Northwest, Ecuador (Esmeraldas) for Socio Bosque program contracts in Chapter 2.

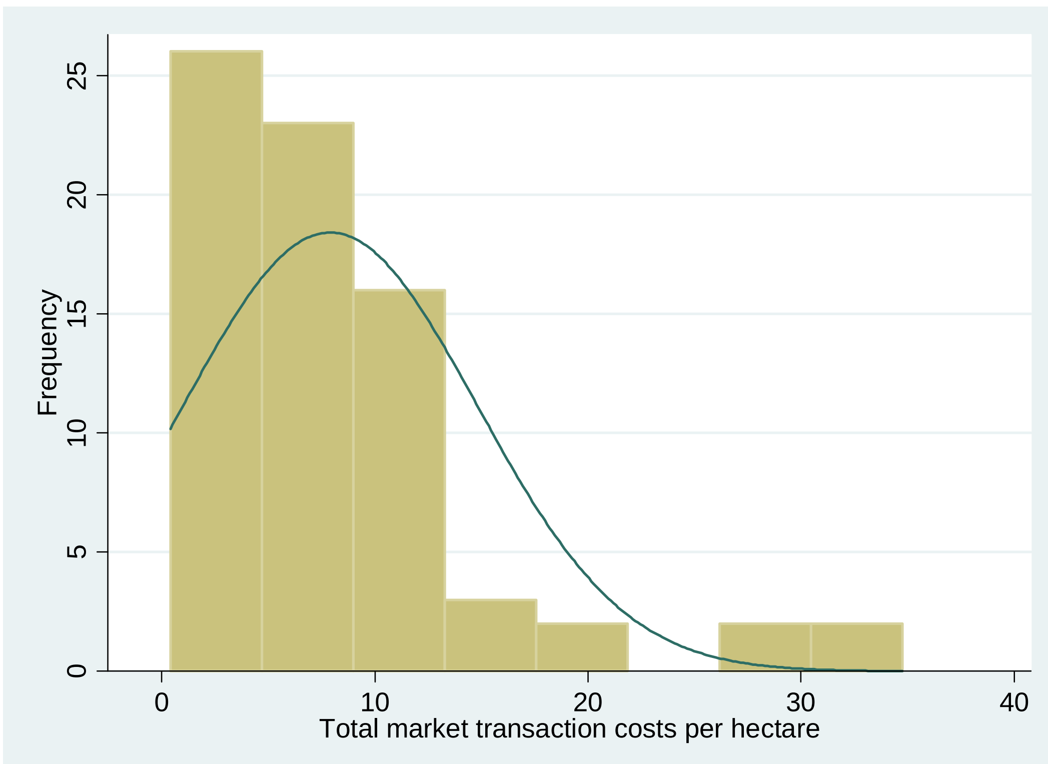

Table B Distribution of per-hectare market transaction costs in Northwest, Ecuador (Esmeraldas) for Socio Bosque program contracts in Chapter 2.

* Standard Deviation = 6.88 Min = 0.40 Max = 34.7

[i] Recent PES assessments mostly have a regional focus and heavily rely on numerous case studies to draw causal inferences with respect to constraints for adoption or to evaluate efficiency, effectiveness and equity implications.
[ii] Conversely, carbon sequestration services are produced locally but their beneficiaries are globally distributed and, thus, arranging terms for this transaction demands an institutional framework distributed across national and international governance structures.
[iii] This model is an extension of the general learning model developed and formalized by communication theorists analyzing the diffusion of an innovation through a fixed size social system (Berry 2008).
[iv] Denzau and North (2000) note that individuals with different learning experiences (both cultural and environmental) are thought as having common cultural backgrounds and experiences, sharing mental models, ideologies, and institutional preferences.
[v] This is because the aforementioned cues (Walker 1969; Weyland 2005) can help decisionmakers' satisfy their information needs (Simon 1976) and enable internal choice.
[vi] This notion follows the same logic described by Brooks (2007) by which competitive benefits at the national level often arise from an innovation's effect on the reputation or international status of the government and its ability to capture excludable goods such as domestic or international foreign investment, development aid or admission into international organizations.
[vii] Implementation of administrative innovations in neighboring jurisdictions can also have a positive effect on increasing social demand for adoption.
[viii] This notion of induced institutional innovation seems to address previous concerns raise by Grabowski (1988) about the necessary attention to changes in resource and cultural endowments and factors such as transaction costs.
[ix] This argument follows the logic of the process of upward and downward causation (Hodgson 2000:326): "reconstituive downward causation -individuals create and change institutions, just as institutions mold and constraint individuals".
[x] Denzau and North (2000) argue that shared mental models guide choices and shape the evolution of political-economic systems and societies.
[xi] Despite the increasing interest in the use of direct approaches to finance conservation (e.g., FAO 2004; Postel and Thompson 2005; Ferraro and Kiss 2002), the factors influencing adoption of watershed PES institutions at the local level within a country has not been fully explored.
[xii] Adoption of local PES program will occur until gains from adoption are no longer positive.
[xiii] Note that single or group membership in regional or international network of support for PES program would tend to reduce transaction costs associated with the implementation and operation as it is likely to increase available information and thus positively affect the number of adoptions. However, PES networks seemed to have become stronger only in the most recent year. In this study, it is assumed that the network effect would become relevant in future studies as the diffusion of PES continue to mature.
[xiv] This is similar to the effect of transaction costs on trading in environmental pollution (Stavins 1995).
[xv] Because I expect spatial effects to reverberate across a region and not just from the closest country, I adopt an autoregressive function. This spatial dependence is key for our analysis; I employ a spatial lag model because it allows observed features to be correlated while estimating the nature and significance of that dependence (Anselin 1998).
[xvi] This model assumes that transaction costs decrease as the number of adoptions reached increases because of learning that occurs over time and space. If so, adoptions with large gains occur earlier than

other adoptions. In fact, it is assumed that even with high transaction costs, adoptions will occur where potential large gains from adoption are perceived by local managers.

[xvii] Issues associated with the use of aggregated indicators (indices) for institutional and policy characteristics may raise concerns for external validity. Critics of qualitative indicators point to significant limitations. It has been argued that their use increases measurement error and sample selection bias, and also decreases transparency. Other criticisms include the creation of an arbitrary scale that does not allow for monitoring of changes over time (see Arndt and Oman, 2006; Glaeser *et al.*, 2004) and that fails to capture the reality between intended and actual outcomes (e.g., Woodruff, 2007). Given the large number of countries and the fact that most data is available only for a limited number of years, the dataset used for this study´s institutional and policy indexes were constructed by Berkman *et al.,* (2008) using averages of data, ranging from either 1980 to the present or 1990 to the present, conditional on the availability of data. Using average values of the indicators does not allow predicting changes in administrative outcomes over time. It does, however, ensure that the measures of institutional capacity are not influenced by the economic conditions of the country. Critics of qualitative indicators based their arguments in part on their belief that experts who provide data for some of the sources may be influenced by financial or political crises and by perceived changes or long-term trends in a country's economic performance (Kaufmann, *et al.,* 2005). Conversely, Berkman *et al.,* 2008 argued that by averaging the data available for their measurements of institutions, it can be ensured that inflated or deflated scores on certain measures are smoothed out over time and provide a useful picture of the state of institutions and related public administrative outcomes.

[xviii] In the absence of cross-sectional data on public environmental expenditures as excludable goods, the use of public health expenditures as proxy rest in an assumption that is generally acceptable if I imagine the differences between the kinds of problems related to drinking-water management in the developing world. In countries with low access to safe drinking-water there are mainly problems with infectious, communicable diseases, very often with large outbreaks of these diseases. The meeting point of public health and environmental expenditures in developing countries seems to be the water quality, especially in the area of the disinfection of water.

[xix] i.e., that credits are real, additional, permanent, and without leakage.

[xx] The questionnaire elicited information from respondents concerning several activities implemented for setting-up a contract. On information, costs measures are computed from collected data for initial contact, application submission, registry and validation. On negotiation, questions were presented regarding contract bargaining and signing. On monitoring, costs associated to field inspections were elicited from landowners. In general, questions on costs measures for every activity presented in Table 1 was presented to individuals interviewed.

[xxi] PROFAFOR has been described as an example of an Annex 1 partner acting as the project financer, developer, implementer, investor and seller of carbon credits. The Annex 1 partner has also established its own office in the host country leaving only the co-implementation and production of carbon to the host country (Milne 2002).

[xxii] PROFAFOR measures the service every year through fixed sample parcels extrapolated to the rest of the contracts.

[xxiii] The error range of such biomass estimates is suggested to be as high as ±30 to ±60%.

[xxiv] $109.2 = (182*1.2)/2$

[xxv] This result is consistent with Archard et al. 2002 resulting regional estimates: 129 tons of carbon (tC) ha^{-1}1 for the pan-Amazon and Central America region, 190 tC ha^{-1} for the Brazilian Amazon forests (23), 179 tC ha^{-1} for tropical moist Africa, and 151 tC ha^{-1}1 for Southeast Asia.

[xxvi] Carbon to CO_2 conversion factor is 3.6667. About 400 t CO_2 ha^{-1} .
[xxvii] For convenience, the share functions are specified in logistic form in order to restrict the estimated shares to the unit simplex.
[xxviii] Note that this is different from Wunder's net present value analysis approach to estimated rental values for tropical forests during a fifteen-year period. For Wunder, during full forest conversion cycle the first two years are assumed to be dedicated to wood extraction (timber and charcoal), the next four to agriculture and the last ten to cattle-ranching. According to field observation and our interest on comparing only forest conversion to agriculture, we only contemplate the first year dedicated to wood extraction. The specification of our land share model will allow us to explore how the landowner will compare forest rents versus agriculture rents.
[xxix] Although population and household census data were available for 1990, socio-economic data for each parish were not interpolated between the available years so that it would conform to the year of the CNA data.
Given the proximity of collection data, we chose to use 2000 data and assume that no major differences exist in months in between when the data were collected for agricultural and population census.
[xxx] The error range of such biomass estimates is suggested to be as high as ±30 to ±60%.
[xxxi] 109.2 = (182*1.2)/2
[xxxii] This result is consistent with Archard et al. 2002 resulting regional estimates: 129 tons of carbon (tC) ha^{-1}1 for the pan-Amazon and Central America region, 190 tC ha^{-1} for the Brazilian Amazon forests (23), 179 tC ha^{-1} for tropical moist Africa, and 151 tC ha^{-1}1 for Southeast Asia.
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