Modeling Spatially Differentiated Environmental Policy in a Philippine Watershed: Tradeoffs between Environmental Protection and Poverty Reduction

Bayou Demeke

Pennsylvania State University (bed155@psu.edu)

Ian Coxhead

University of Wisconsin-Madison (Coxhead@wisc.edu)

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Abstract

Erosion and sediments are among the most important externalities in the developing world. These sediments negatively affect the quantity and quality of water in the downstream regions of watersheds. In light with the growing interest in many developing countries to use market-based instruments, this paper develops a model for designing efficient environmental policy at a watershed scale. Because farm households are heterogeneous in a given watershed, we develop a spatially explicit, heterogeneous watershed scale environmental policy to lesson watershed degradation. We use GIS data and geo-referenced household plots to populate the watershed with the heterogeneous households. Heterogeneity also implies that the impact of environmental tax policy on poverty varies among households. The model results confirm the benefits of differentiating policy based on the spatial attributes of the watershed. Our study proposes the possibility of funding poverty reduction using the revenues from environmental taxes. The results show that, for a moderate reduction in soil erosion, revenues from environmental taxes could be used for poverty reduction. However, for larger improvement in environmental quality, the efficient environmental tax would not be sufficient to compensate the poor. Our findings reveal the extent of tradeoffs between poverty reduction and environmental protection. In other words, tighter environmental policies could exacerbate poverty unless assistance is provided to the poor.

Key Words: Environmental Policy, Heterogeneous, Philippines, Poverty, Watershed

1. Introduction

Erosion and sediments are among the most important externalities in the developing world (Shively, 2003). Agricultural runoff negatively affects the quantity and quality of water in the downstream regions. Sedimentation in streams increases the risk of flash floods. Accumulation of silt in reservoirs reduces the supply of irrigation water, and also increases the cost of hydroelectric power generation. Given these downstream effects of agricultural runoff, improved watershed management is a major policy goal in many erosion-prone uplands of developing countries. However, empirical studies of watershed level environmental policy are lacking in developing countries, despite the growing problems of severe watershed degradation, and despite the need for appropriate policies to reverse the downward spiral of environmental quality. In the Philippines –the country from which data used in this study are drawn– an estimate by the Environment and Natural Resource Accounting Project (ENRAP) put the costs of deforestation, loss of watershed function, and treatment of polluted water supplies at 7,680 Million Pesos in 1992 - approximately 0.9% of GDP (Rola et al.). Despite significant efforts made to facilitate soil conservation at the farm level, there still remain significant external costs that need to be addressed with appropriate environmental policies. Land use planning and other regulatory approaches have had little success in South East Asia (Tomich et.al 2004).

The economics literature has contributed quite significantly to the understanding of on-site incentives to adopt improved land management practices in a dynamic context and under various assumptions about the economic environment of the decision makers (Burt 1981, McConnell 1983; Barbier 1990, Barrett 1991; LaFrance 1992; Demeke et. al. 1998). Besides the onsite impacts, the consideration of offsite damages from runoff has received significant attention, mostly in the developed countries. Due to external effects, on site impacts of soil erosion are not usually sufficient to induce socially optimal adoption of better land management practices. Due to the increasing problems of pollution-externalities form agricultural fields, there is growing interest in regulating pollution that is delivered from agricultural sources through runoffs. Initiatives aimed at reducing these forms of agricultural externalities often need to have significant spatial components (Hochman and Zilberman 1977, Shortle and Horan 1998, Schwabe, 1999).

The objective of this study is to design a spatially explicit efficient environmental policy at a landscape scale, following the methodology used in economic studies of non-point source pollution. It is a well-documented fact that soil erosion from a given agricultural field depends on the physical characteristics of the field and the choice of land use. The contribution of an agricultural field to quantity of sediments that reach down streams sites is also affected by the topographic attributes such distance to the nearest stream, slope and, and land use choices When farm fields are heterogeneous, uniform targeting policies would not be economically efficient. By economic efficiency we mean the ability to attain a given environmental quality at the least cost. The main objective of this study is, therefore, to design a least cost, spatially explicit watershed policy to reduce the downstream effects of sediments leaving agricultural lands. Empirical evidence has shown that producers in the study area do respond to price incentives and to their household level resource constraints (e.g. Coxhead and Demeke 2004; Coxhead et.al. 2002, Demeke and Coxhead 2005). Thus the empirical question that follows from there are: i) What policies should be used to efficiently limit the level of environmental degradation to an acceptable level? ii) How large is the benefit of heterogeneous policies as opposed to of uniform policies? iii) Which mix of land use/conservation strategies should be adopted for each type of land in a watershed to maximize economic returns from the whole watershed for any given level soil erosion? iv) To what extent could environmental taxes be used as a poverty reduction strategy when there are significant income disparities among farm households?

This study adds to the existing literature on spatial modeling, but also makes contributions in other areas. First, this study contributes to the literature on environmental policy by empirically analyzing policy in a developing country context. The literature on non-point source pollution focuses heavily on developed countries' agriculture, even though water pollution in developing countries is increasingly becoming one of the major environmental problems. In addition, the models used in a developed country context typically assume complete markets for factors and outputs. However, it is widely known that households in developing countries operate under incomplete factor and output markets. Their decisions are constrained by incomplete credit markets, which limit their ability to use purchased inputs; imperfect labor markets, which limit their ability to hire labor or find employment outside their farm; constrained land markets that limit their ability to sell or rent their plots, and underdeveloped infrastructure that raises the costs of marketing (De Janvry, et.al. 1991; Carter and Yao, 2002; Demeke 2004). In many developing countries, households also face significant land tenure insecurities that limit their incentives to invest in long-term land improvements (Larson and Bromley 1990; Smith 2004). On top of these constraints, farm households in developing countries are poor. Thus policies aimed at changing the production environment of these households will most likely have a consequential effect on poverty. Our model incorporates many of these features of a typical developing country decision making environment by incorporating additional constraints such as capital availability, household labor force, and land availability.

Second, we use detailed topographic information from GIS data and additional geo-referenced information to link production technologies and heterogeneous household characteristics to spatially differentiated landscape features. We have a decade of survey data on our sample households in the watershed and their farms have been geo-referenced. The survey data, together with the topographic information, is used in identifying technology parameters and other household characteristics for all parts of the watershed. Our model includes not only differentiated geographical features but also differentiated household characteristics with their corresponding production technologies. Thus the study includes household characteristics in addition to spatial physical characteristics. This paper will therefore contribute to the growing literature on the economics of non-point source pollution, and even more significantly, to the limited literature on empirical application of watershed-level spatial policies in developing countries. The literature on heterogeneous environmental policies in developing countries is limited; to our knowledge, this is the first application of an economy-environment model of this type in a developing country context. Thus the methodology will add quite significantly to the economics of public policy and environmental policy in developing countries.

Third and finally, with growing income disparities among rural households and with the continuing prevalence of acute poverty, environmental policy cannot be made in isolation, and indeed could be used to address broader welfare problems. In this context we discuss the distributional and welfare implications of alternative policies including marketable permits, taxes, subsidies and auctions. In particular, this paper explores the extent to which the revenues from are capable of reducing acute poverty.

The remainder of the paper is organized as follows. Section 2 presents a literature review on non-point source pollution. In section 3 we present an economic model of pollution control. In section 4 the spatial model of the Manupali watershed is presented. In section 6, we present and discuss the results. Section 7 discusses income distribution, economic welfare and the environment. Section 8 presents extensions to the model. Section 9 concludes.

2. Policies to reduce agricultural externalities: review of the literature

Market mechanisms have been widely promoted by economists in the past decades to address environmental problems, and there is a growing interest by policy makers in the U.S., Europe, and in some developing countries to use such instruments (Shortle and Horan, 2001). The importance of market-based mechanisms in environmental policy making is thus growing. In theory, market-based mechanisms equalize marginal abatement costs between polluting firms, thereby allocating the control of emissions at least cost. Despite the efficiency advantage of economic instruments over "command and control", in many developing countries authorities commonly prefer the former due to the difficulties associated with implementing economic instruments. Economic incentives differ from command and control regulation in two main aspects: 1) differences in firms' marginal abatement costs determine the allocation of pollution reduction responsibility, and firms have both the right and the incentive to shift their marginal abatement cost curves through innovation. 2) The marginal abatement costs are equalized in equilibrium across all polluting firms. Often command and control policies by definition set emission levels (or standards) at an arbitrary level without considering the abatement cost differences of the firms- except in rare cases where the command control choice is set to the efficient abetment level.

Recent developments in the study of agricultural non-point pollution policies emphasize the use of site-specific heterogeneous tax policies (e.g. Fleming and Adams 1996, Schwabe 1999; Braden et al 1989, Coxhead 2002; Khanna et.al 2003; among others). This study will build on this body of literature to allow for integrated, site-specific, land-use specific environmental policy to address the problem of runoff externalities in the Manupali watershed of the Philippines. We will make use of GIS data to incorporate spatial heterogeneity that affect sediment delivery and a suitable biophysical model to predict environmental outcomes. We will also address the effect of different environmental policies on income distribution and poverty, and the nature of any tradeoffs involved.

There is an extensive literature in environmental economics on the theory of negative externalities and policies to minimize their effects. By definition, the existence of a pollution externality calls for government intervention in order to improve social welfare (Baumol and Oats 1971; Hochman and Zilberman 1978.) The most frequently cited remedy, in a first best world, is the adoption of emission taxes where the taxes are set equal to the marginal social damage -at a point where marginal benefit from production is equal to marginal social damage from pollution. Even though such Pigovian taxes produce a Pareto-optimal resource allocation, they are rarely used, mainly because of the difficulty in measuring the social damages; i.e. the cost of pollution. An alternative practical remedy, cited in much of the environmental economics literature, is to set a desired environmental quality standard and then impose a tax to attain it (e.g. Baumol and Oats, 1988; Griffin and Bromley, 1982; Ribaudo, Horan and Smith, 1999). Such a policy, though less efficient, can in principle attain the environmental standard at a minimum cost to society. The reason such policies cannot attain the full efficiency is the lack of information on damage cost. Equally efficient remedies, under a wide range of conditions, are marketable pollution permits, and subsidies on pollution reductions (Baumol and Oats 1988; Griffin and Bromley 1982). The cost effectiveness methodology that we have adopted has also been widely used in air pollution literature (e.g. Atkinsn and Lewis 1974; Tietenberg 1995; Newell and Stavins 2003). The global warming literature also uses cost effectiveness due to differences in production technologies and abatement costs.

Several studies have considered the spatial aspect of agricultural pollution externalities. Hochman, Pines and Zilberman (1977) developed a theoretical model for a rectangular river basin that is used for agricultural production along the river, and a city at the bottom of the river basin that suffers from agricultural pollutants. Their model allows for natural absorption of the pollutants along the river, implying that the pollutant coming from the topmost agricultural land will contribute least to deposition of the pollutant at the city center. Pollutants can be abated using labor in addition to natural absorption. The damage cost to society is assumed to be known and linearly increasing in the amount of pollutant deposited. Transportation cost is assumed to be increasing in distance from the city center, where producers sell their output. Without any environmental policy the model predicts that, because of transportation costs, both the intensity of cultivation and land rents are decreasing functions of distance from the city center- the standard von Thünen result. When optimal environmental policy is adopted through taxes per unit of pollutant, the von Thünen result could be reversed, depending on the natural absorption rate and pollution to output ratio. The optimal pollution tax decreases with distance and hence could offset the effects of transportation cost in the standard von Thünen prediction.

This model shows how heterogeneous taxes that depend on the distance from the bottom of the river basin could be used to reduce agricultural externalities. Though an efficient outcome is in theory possible, in practice it is usually impossible to know the exact cost of the externality. The model also assumes that only one good is produced, and does not allow for the impact of crop choice on pollution. In the model spatial variation across farms is assumed to be one-dimensional (e.g. distance from the center of the city), although in reality land is heterogeneous in terms of slope, soil type, elevation and other relevant characteristics. The other difficulty with the application of this model is the prohibitive cost of measuring pollutant loadings from each farm along the riverbank.

To overcome the prohibitive cost of measuring emissions by individual firms, Griffin and Bromley (1982), in their theoretical analysis of agricultural runoff control policy, suggest the use of inputs and crop choices that are correlated with emissions. For the case of point-source pollution, where pollution from each firm can be measured directly, the design of optimal pollution taxes is straightforward. For the case of non-point-source pollution, however, incentive policies should be directed at pollution-generating activities or inputs, and with appropriate policy design, an efficient outcome can be attained just as in the case of point-source pollution. The degree of efficiency, of course, depends largely on how well the input choices, outputs levels and crop choices predict environmental outcome. Coxhead (2002) similarly developed a model of a tax policy that incorporates spatial characteristics of agricultural pollution, and the role that devolution plays in the design of local environmental policy. The model also provides a framework for including taxes on land use decisions such as crop choices.

Other policy instruments proposed for controlling agricultural pollution include ambient taxes, random fines, and type-specific contracts (Shortle and Dunn 1986; Segerson 1988; Xepapadeas 1992; Cabe and Herriges, 1992). The high costs of acquiring information on farm-level characteristics and transaction costs have led to suggestions of using such second best policy instruments (e.g. Cabe and Herriges 1992; Helfand and House, 1995; Wu and Babcock, 1996). In practice, in the USA, the emphasis has been on voluntary compliance approaches, which combine public persuasion with technical support to help farmers adopt environment-friendly practices. (Shortle and Horan, 2001). However, such approaches are generally not very effective as there is not enough incentive to adopt costly conservation practices without being compensated for doing so.

There is a growing empirical literature on the use of spatially-explicit environmental policy at a watershed scale. Most empirical studies used constrained optimization (i.e. to maximize total profits subject to a limit on total pollution level such as sediment and soil loss, nitrogen loading, phosphorous pollution) in the design of pollution control policies. Several studies have concluded that highly targeted information-intensive strategies outperform uniform strategies in the design of non-point control policies (Braden et. al. 1989; Babcock, et al. 1997; Schwabe 1999, Classen and Horan 2001; Khanna et al. 2003). For example, Khanna et al. (2003), in their study on cost-effectiveness of the land retirement reserve program in Illinois, find that the costs of achieving an abatement goal of 20% are much lower with differentiated standards than with uniform standards. Under the differentiated standard some watersheds should abate much less than the 20%, while some others need to abate much more.

In contrast, other empirical studies suggest that that the benefits of heterogeneous policies over those of uniform policies are not very significant. For example Fleming and Adams (1997) assess the importance of spatial variability in the design of a tax policy to control groundwater nitrate concentration from irrigated agriculture. Their results indicate that a detailed accounting of spatial heterogeneity had little effect on the selection of a cost effective tax policy. The differentiated tax policy yields a higher aggregated farm profit than the uniform tax does, but the difference is about $7 per acre. In another study, using data from California's Salinas Valley, Helfand and House (1995) compare several instruments to reduce nitrate leaching by 20% from two soils used for lettuce production. They evaluate the cost effectiveness of taxes and restrictions on nitrogen inputs and on irrigation water inputs. They find that uniform policies are slightly less cost-effective, but not by much. Their results that are consistent with the theory. However, the magnitude of the gains from the differentiated policies generally depends on the extent of spatial heterogeneity. For example, Fleming and Adams (1997) classified their study area into 4 sub-areas based on soil type. The four soil types might not have significant impact in terms of environmental outcomes, and thus the gains from differentiated policies are limited. However, when there is significant spatial variation in terms of pollution contribution, due to physical characteristics of individual farms and land management, use of a uniform policy is known to be inefficient (Shortle and Horan, 2001). When farms at different site in a watershed differ in their sediment contributions to a common catchments area, Braden et.al. (1989) show that an optimal regional sediment plan has to take into account the effects of land management at locations between any given source and the catchment area. Thus, whether the benefits of differentiated policies are large enough to warrant spatially-specific policies is an empirical question whose answer depends on the characteristics of the watershed under consideration.

Most of the empirical literature is also confined to the control of agricultural non-point sources in North America and Western Europe (Shortle and Horan, 2001), and empirical studies in developing countries are lacking. The modeling approach for control of non-point source pollution in developing countries should differ for a number of reasons, among which, the absence of complete markets, the absence of well-defined property rights, weak legal and institutional structures, and the prevalence of widespread poverty are the major ones.

2.1. Developing countries and payments for environmental services

There is growing interest in developing countries to protect the environment using market-based instruments. Awareness of environmental services and land use change in Southeast Asia is high among scientists, policymakers, and society as a whole. And yet policy-relevant results are regarding sedimentation and other downstream effects of soil transfer are rare (Tomich et.al. 2004). Several initiatives are underway in South East Asia to reward the poor for environmental services they provide, for example by refraining from certain polluting farming practices. These alternatives are sought because, with the exception of large commercial farmers, many upland farmers are very poor, and implementation of punishing (tax) policies to protect the environment may be associated with aggravated poverty (Coxhead, 2004). Hence these new initiatives by definition give the poor the right to pollute, implying that the beneficiaries of improved environmental quality will have to pay for the environmental services (see Pagiola, 2002, for case studies). Such initiatives provide both environmental quality improvement and poverty alleviation in the uplands. For instance, in south East Asian countries, a pilot program called Rewarding Upland Poor for Environmental Services (RUPES), coordinated by the World Agroforestry Centre (ICRAF), and supported by the International Fund for Agricultural Development (IFAD) aims at developing mechanisms for rewarding the upland poor in Asia for the environmental services they provide. Some of the major environmental services identified are watershed protection, biodiversity protection and carbon sequestration. Possible sources of funding for the payment of the environmental services are NGOs, governments, international organizations such as the World Bank, Nature Conservancy Groups (REECS, 2003, Arocena-Francisco, 2002, Pagiola 2005). Based on the principle of Coase theorem, the direct beneficiaries of the environmental services such as hydroelectric generating companies in the downstream, irrigation water users, etc could pay for environmental services (see Kerr 2002; Pagiola 2005). There are several difficulties with such Environmental Services Payment (ESP). First, the obvious difficulty is finding the finances to pay these upland poor due to poverty of the downstream users or due to free rider problems. Outside finances, from NGOs for example, are not always sustainable. Second, the ESP is in principle uses subsidies to reduce pollution. The standard problem associated with subsidies, especially when property rights are not well defined, is that it will create incentives for countervailing entry of new firms (See Baumol and Oates 1988; Pagiola 2005). In addition, since not all upland farmers are poor, payment for the environmental services might not necessarily be the most efficient way to reduce poverty. When a subset of farmers is non poor, it is actually possible to use pollution tax revenues to compensate the poorer households.

With spatial variation in degree to pollute, a least cost policy calls for building a spatially-explicit integrated economy-environment model that involves quantifying environmental quality improvement and estimation of abatement costs under alternative scenarios. We will present the conceptual economic model in the next section.

3. An Economic Model of Pollution Control Policy

The conceptual economic model aims for a least cost means to reduce detrimental externalities, with explicit accounting for spatial dimensions. Assume that there are J unique areas (or zones) in a watershed, each with its corresponding biophysical attributes, and each operated by a number of uniformly similar farmers. Let the j^{th} farmer's vector of netputs be denoted by $\mathbf{y^j}$ with y_i^j being the i^{th} element of the vector. The production function is given by $f(\mathbf{y^j}) \leq 0$ with corresponding aggregate pollution function

(1) $$d^j = g^j(\mathbf{y^j}).$$

The social planner's problem is to maximize the sum of current profits consistent with a targeted pollution level, or

(2) $$\underset{y^j}{Max} \sum_j \mathbf{py^j}$$

subject to the constraint on total pollution.

(3) $$\sum_j d^j \leq \bar{d}$$

where $\mathbf{p}$ is a vector of prices and $\mathbf{py^j}$ represents profit for the j^{th} farmer.

Assuming concave production functions, and that all the constraints are binding, the Lagrangean of the problem is given by:

(4) $$L = \sum_j py^j - \sum_j \psi_j f(y^j) + \sum_j \eta_j [d^j - g^j(y^j)] + \lambda\left(\bar{d} - \sum_j d^j\right)$$

where ψ_j, η_j and λ are the Lagrangean multipliers for the production function, pollution function and allowable emission level respectively. The multiplier λ is interpreted as the shadow price of the allowable pollution constraint given in equation (3).

The first order conditions for interior solutions with respect to y_i^j, d^j are:

(5) $\partial L / \partial y_i^j = p_i - \psi_j f_i^j - \eta_j g_i^j = 0$ for all i and j

(6) $\partial L / \partial d^j = \eta_j - \lambda = 0$ for al j

(7) $$\partial L / \partial \lambda = \bar{d} - \sum_j d^j = 0$$

(8) $\partial L / \partial \eta_j = d^j - g^j(\mathbf{y^j}) = 0$ for all j

(9) $\partial L / \partial \psi_j = f(\mathbf{y^j}) = 0$ for all j

The system of equations (1) through (9) define the optimal solution to the problem. After rearranging, (5) states that the marginal value product of using input i is equal to price of input *i* (the value of its marginal product) plus the value of marginal pollution associated with its use. Thus, by comparison with the case of no restriction on allowable pollution, the model implies lower use of input *i* if pollution increases with its use. With no environmental policy in place (i.e. when the allowable pollution level is not limiting), the producers will maximize profit ignoring the condition indicated in equation (3). However, when the social planner wants to limit total pollution to a level below the aggregate of privately optimal pollution levels, the condition given in equation (3) becomes part of the maximization problem. The optimal solution to the planner's problem then requires higher abatement efforts from high-pollution agents. With spatially distinct pollution characteristics, then, the solution to the social planner's problem will be a unique tax rate for each land area unit–that is, spatially differentiated targeting of abatement policies. To induce socially optimal production decisions by each agent, the per-unit tax or subsidy, τ , set by the social planner is found by choosing a tax per unit of pollution reaching the receptor site equal to λ. This policy, of course, requires the measurement of pollution reaching the receptor site from each agent.

Another way to pursue the optimal policy is to base the taxes on observable choices that producers make and on the biophysical environment in which they produce. When there is a known functional relationship between emissions and the set of inputs and output choices, it is possible to design incentives based on these choices rather than on an emissions charge. Because of variations in biophysical characteristics among plots in a watershed, the marginal land use incentive is different from the marginal incentive for each farmer operating at a unique site. Without loss of generality, assume that the *J* unique areas are ordered (from low to high) in terms of their erosion potential- such as field slope and distance to a stream. Denote the erosion potential by *a(j),* with *a(j)* increasing as *j* increases. For simplicity, assume that pollution is a linear function of area planted for any given *j*. Redefining the pollution function, we have

(10) $$d^j = a(j)A^j$$

It is straightforward to show that tax per unit of area planted, ϖ , will increase as *j* increases since $\varpi = a(j).\tau$ and *a(j)* increases as *j* increases.

We have assumed that the household's decision is static even though the model could be enriched by inclusion of a time dimension. However, the general implication on pollution control responsibility does not change by adding dynamics. The use of a static model is justified on several grounds. 1) A dynamic model is appropriate when the agents have an incentive to be forward-looking. With weak property rights, the incentive to invest in land improvements is very small. 2) The dynamic model will increase the dimensions of the empirical model quite significantly. The gains from including dynamics thus come at a high computational cost. Thus we believe that, for our purposes, a static model is sufficient to highlight the major elements of the environmental policy we are considering. However, this is not to say that a dynamic model is not desirable, and thus we will comment on how to extend the static model towards the end of this paper.

4. Empirical Application to Manupali Watershed

4.1. Environmental Degradation in Lantapan

The Municipality of Lantapan is found in the upper part of Manupali River watershed. The boundary of Lantapan runs from 15 km south of Malaybalay City along the southern boundary of the Mount Kitanglad Range Nature Park. The landscape of Lantapan climbs from river flats (400-600m) through a rolling middle section (600-1100m) to a high altitude, steeply sloped mountainside (1100-2200m). The municipality of Lantapan consists of five major sub-watersheds all draining from Mount Kitanglad to the Manupali River. The Manupali River runs into a dam that diverts flows into Manupali River Irrigation System (MANRIS), a 3350-hectare system constructed by the National Irrigation System. The River ultimately drains into Pulangi River, one of the major waterways of Mindanao Island, a few kilometers upstream from the Pulangi IV hydroelectric power generation facility, one of the six largest hydropower-generating plants in the country (Coxhead, 2004).

For the past several decades, rapid agricultural growth in the Philippines has been fueled by combination of three factors: a) population growth and migration; b) unrestricted access to forest margin areas for conversion; and c) agricultural development policies that provide incentives for upland farmers to intensify land use by planting vegetables and corn rather than perennials (Coxhead, 2004). The main cause of population growth in the study area is migration caused by opportunities created in uplands through infrastructural development and promotion of commercial crops. Population growth in the uplands is also driven by limited growth of economic opportunities in non-agricultural sectors and lowland agriculture (Coxhead, 2004). Agricultural expansion into forest margins is the result of ill-defined property rights and weak institutions to enforce environmental policies. In pursuit of food self-sufficiency, the Philippines government has awarded considerable support to the production of corn and vegetables through high tariffs on these goods and their substitutes (Coxhead, 2000). Thus, increased import protection for these crops is associated with rapid land degradation in the uplands. Markets and policies for agricultural products have clear consequences for agricultural expansion and intensification. Empirical studies support this; in Lantapan, relative prices play significant roles in the expansion of corn and vegetable areas (Coxhead and Demeke, 2004; Coxhead, Shively and Shuai, 2002).

Trends in watershed function capture the combined effects of deforestation and upland land degradation (Coxhead, 2004). The removal of biomass in the form of forest cover reduces water storage capacity in upper watershed areas and exposes soils to rain and wind. Data from the Manupali watershed show that land clearing and conversion of cleared land to agriculture are strongly associated with increased fluctuations of seasonal stream flows, decreased overall stream flows, and increased loadings of sediments as well as pollutants introduced by cropping activities (Deutsch et al. 2001). The effects of deforestation and upland land degradation are not confined to upper-watershed areas; water pollution and soil transport contribute to sedimentation in dams and canals, accelerated wear on turbines and other hydro-power generation infrastructure, increased health costs for downstream human and animal populations (Doolette and MacGrath 1990). The downstream effects of agricultural intensification in the municipality of Lantapan have been severe. Water pollution in the lower parts of the municipality, and in downstream the municipality, show high loadings of bacteria, such as E. coli, that are detrimental to human health (Deustch et al., 2001). Data collected on water quality from sub–watersheds of the Manupali, monitored by locally based citizen volunteers, indicate significantly excessive soil loadings in the streams and high stream flow variability. Also perception of pesticide residue has made some residents avoid bathing their animals in the streams during or after rainfall events (Coxhead, 2004). Another detrimental consequence of soil erosion in the watershed is the deterioration of the Pulangi IV hydropower generation plant. A high level of siltation in the reservoir has prevented the hydropower inform achieving its full capacity. The MANRIS irrigation system is facing similar problems, only irrigating 1000 hectares of land during wet season and about 790 hectares during dry season, much less than its intended 3350 hectares capacity (Coxhead, 2004).

Modeling efficient least-cost watershed pollution control policy requires a large scale economic model capable of estimating the costs of alternative land uses on spatially heterogeneous land, combined with the capacity to estimate the environmental effects of alternative land uses at the watershed scale (Tanaka and Wu, 2004). With increasing availability of GIS data and development of physical models that can predict environmental outcome, site-specific models are becoming popular. Some recently used biophysical models are i-EPIC (Kurkalova, Kling and Zao, 2003), SWAT (Tanaka and Wu, 2004, Ancev, Stoecker and Storm, 2003) and AGNPS (Khanna, et al. 2003). EPIC is a continuous simulation model that can be used to determine the effect of management strategies on agricultural production and soil and water resources. i-EPIC is a user-friendly interactive computer model developed based on EPIC. SWAT (Soil and Water Assessment Tool) is a USDA-ARS hydrologic model developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time. These biophysical models differ in complexity, data requirement and prediction accuracy. Even though these models are useful in accurately measuring soil erosion at a basin level, however, the database of information they require is relatively complex. In addition, most of these process-based models are not designed for optimization, and it is infeasible to run these models over the possible alternative land uses from each zone in a watershed.

In this study we will make use of the much simpler and widely used soil erosion model, the Universal Soil Loss Equation (USLE). Even though our results could benefit from use of better soil erosion models such as SWAT, we have chosen USLE for the following three reasons. 1) It is very easy to apply 2) USLE requires less data: for example, unlike the SWAT model which requires continuous-time weather data, USLE requires only average yearly rainfall. 3) Given the simple form of the USLE, it is possible to include the equation in an optimization model. Following this we will briefly discuss the USLE and the data inputs needed for calculation of soil erosion.

The Universal Soil Loss Equation is an empirical model developed by Wischmeier and Smith (1978) to estimate soil erosion from fields. While its limitations have been extensively documented, the equation remains the basis for soil and sediment measurement and prediction applications in many countries. The USLE parameters have also been adapted to fit local conditions in many tropical countries.

The USLE equation is given as:

$$(11) \qquad d = R \bullet K \bullet L \bullet S \bullet C \bullet P$$

where, d is soil loss in tons per hectare per year, R is a rainfall-erosivity index, K is a soil erodibility index, L is slope length, S is the slope steepness factor, C is a land cover management factor, and P is a supporting practices factor. Data input, calculation of each factor and data pre-processing are discussed in greater detail below.

The USLE does not account for the deposition of detached soil along hillslopes. We augment the USLE by assuming that plots farther away from streams have lower sediment delivery ratios (see Amore et. al. 2004 for an excellent discussion of how slope length and the size of the watershed considered for analysis affects predicted soil erosion). We instead take a simple linear approach to the effect of distance on the sediment delivery ratio (SDR). We assume that the furthest point in the watershed from a stream will have an SDR value of 0, while the closest will have a value of 1. The in-between cases are linearly interpolated. Most of the USLE parameters we used for this study are given in David (1988).

4.2. Zoning the watershed for spatial analysis

To account for spatial variability within the watershed we take the following steps to arrive at the final data set used for modeling.

Zoning: Using data generated from a 30X30 meter resolution Digital Elevation Model (DEM), we create sub-watersheds and locate the streams. The DEM is also used to calculate slope, using the Arcview GIS Spatial Analyst extension. We then subdivide the fields by slope, distance to the nearest stream, and elevation. We combine those fields that have similar attribute classes into zones representing fairly uniform areas in the watershed. Even though the municipality is composed of 5 major sub-watersheds, our analysis focuses only on the three for which we have household data that are geo-referenced. The three sub-watersheds are the *Alanib, Maagnaw and Tugasan* watersheds. We have divided distance to stream into 6 discrete distances at 200 meters intervals. Any distances above 1000 meters from the streams are assumed to be uniform in terms of contributions to sediment delivered. We have 5 slope classes at 10% intervals, with a maximum slope of 50%. We exclude fields which lie in the forest buffer zone, in forest areas outside the buffer zone, and those that lie in areas that have steepness of 50% or more. The 50% cutoff is based on our observation from the geo-referenced survey farm-plots for which the maximum observed slope was 47%. With these assumptions, we have 149 distinct zones with a total area of 7421 hectares of cultivable land. Table 1 through table 4 present the physical characteristics of the watershed.

4.3. Household Characteristics and production function

After subdividing the watershed into uniform sections, the next task is to associate household characteristics with those subdivisions of the watershed. The problem here is that the household survey was based on a stratified random sample of villages, whereas our GIS data are based on watersheds. Using census population data, with the GIS data base, however, we are able to compute the population in each area of the watershed. We then use the household survey information linked to geo-referenced farm plots to populate the watershed with the desired numbers of households and sets of household characteristics. Table 5 presents the household characteristics in the watershed.

The household survey began in 1994 with 190 randomly selected households and runs through 2002. We geo-referenced the plots of 101 households- those that have remained in the survey until 2002. In populating the watershed with these households, we use the iterative linear imputation procedure provided in STATA[1]. Populating the watershed with these household characteristics contributes additional heterogeneity to the already heterogeneous landscape. Populating was also useful in predicting the spatial production technology for each zone in the watershed. Information such as production technology, household-size, farm-size, percentage of coffee production, and other information are all imputed to the whole watershed.

According to this procedure, in each zone we have *m* identical households (*m* is obtained by dividing surface area of a zone by the corresponding imputed farm size). We also estimated

Cobb-Douglas production functions from the household survey panel data for the three main crops grown crops: corn, vegetables and coffee (see table 6). In addition to inputs used (land, labor, fertilizer), the production function also includes information on field slope, distance to national road, elevation, and distance to stream as additional explanatory variables.

5. The Economy-Environment non-linear Programming Model

At its most basic level, an economy-environment model is an abstract representation of the biophysical, social, and economic features of a well-specified geographical landscape (Shively and Coxhead, 2004). At its most detailed representation, an economy-environment model links people with the spatial landscape they inhabit. In principle it is possible to link the resource constraints and choice sets of those people to the land type they operate. The environmental outcomes of economic decision-making can also be linked directly to distinct geographic regions. Our approach is to take as much as detailed information as we can use to link people to the landscape in the watershed. We then build an economic model that characterizes those distinct zones and the associated household characteristics. As detailed above we link people to each zones based on the sample information.

The integrated economy-environment model developed here is a spatially explicit non-linear programming model consisting of two sub-models – the economy and the environmental sub model.

5.1. The economy sub-model

In this sub-model, households choose land allocation, labor allocation, level of fertilizer application, and levels of soil conservation in order to maximize current period returns given an

[1] We are assuming our sample is representative of the three watersheds we have modeled.

exogenously imposed environmental policy . Following the conceptual model we developed earlier, the per period returns are maximized as follows:

(12) $$\max_{v,A} \sum_{i}^{N} [p^i f^i (A^i, v^i) - wv^i - t \sum_{i}^{N} d_i]$$

where A^i is area of crops allocated to crop *i*, *v* is variable inputs such as fertilizer and labor, *p* and *w* are the prices of outputs and inputs respectively, *t* is tax rate for sediment, and d_i is the soil loss reaching the downstream in the form of sediments.

We assume that farmers have access to limited land resources and describe the constraint as

(13) $$\sum_{i}^{N} A^i \leq \bar{A}$$

Additional constraints are imposed by family labor and capital endowments, which vary depending on the assumptions we adopt on the functioning of labor markets and capital markets. Since we have distributed the households to unique zones, these constraints are unique to each zone. We have developed scenarios for various labor market situations.

5.2. The environment sub-model

The environment sub-model describes the relationship between alternative land use choices and sediment delivery (soil loss), for a given spatial location. Thus choices in the economic sub-model are inputs to the environment sub-model. The impact of production activities of households on the environment is then quantified based on where they are located and what land use choices they make. GIS information is a very powerful tool in locating and mapping the degradation potentials of the watershed. Thus the environmental sub-model links the households' economic decisions to environmental degradation, i.e. in this module for every production decision households make, there is a mathematical formula that links the decisions to sediment delivered. The formula uses topographic information to calculate soil loss from each zone as a function of the following variables based on equation (11).

(14) $h_i = f(\text{Slope, Distance to Stream, Crops grown, Conservation measures})$

The explicit mathematical equation for soil loss is as provided by the USLE in equation (11) with distance to streams as an added dimension. Then we aggregate the soil loss from all the zones to arrive at watershed-scale pollution.[2]

The Social Planner's approach to modeling

In order to solve for the optimal tax, the planner solves the problem given in (12) through (14) without the taxes imposed, but with a target abatement level. Similar to the conceptual model in equations (1) though (10), the planner maximizes net profits, taking account of resource constraints faced by each household. The equilibrium for all households in the watershed will match the planner's problem when the planner imposes a tax rate that is equivalent, for each household, to the shadow price of the abatement target- a target that limits the sediment reaching a receptor site. The model predicts that for a given crop choice, technology, and distance to the stream, taxes increase as slope increases. In addition, we expect taxes to decrease as distance increases, holding the other variables constant. Consider the following grid of a rectangular region that represent watershed.

[2] The validity these estimates of soil loss depend on how well the USLE model predicts sediment delivery. In any case the estimates from such models are expected to have large errors.

In Table 11, the cells depicted by ***R*** represent streams. The first element in any cell represents distance to stream, while the second element represents slope; in each case, higher numbers imply higher values. Thus our economic model predicts that the abatement responsibility of **(1,4)** is greater than that of (**1,1**), and so cell (**1,4**) pays more tax than cell (**1,1**) for a given land use. Notice that these two cells are equidistant from the stream. Going up through the columns, the model also predicts that cell (4,1) pays less taxes than, for example, cell (**2,1**). In addition, within a given cell, if a household adopts soil conservation methods, then the tax payment is less. Similarly a farmer will pay tax at a higher rate on erosive crops than on less erosive crops.

Table 11. Visual representation of hypothetical cells in a watershed.

	Slope →			
Distance To streams	4,1	4, 2	4, 3	4, 4
	3,1	3, 2	3, 3	3, 4
↑	2,1	2, 2	2, 3	2, 4
	1,1	1, 2	1,.3	1,.4
Stream	**R**	**R**	**R**	**R**

The policy simulation is undertaken by successively restricting the total allowable level of sediment delivered to the receptor site located at the lower end of the watershed, where it is assumed to cause externality costs.

As described in the theoretical section, the model is run by setting taxes exactly equal to the marginal value of sediment delivered, as obtained from the constrained optimization problem without including restrictions on the allowable sediment reaching downstream sites. This exercise is done to check theoretical equivalence and also to check for consistency of the GAMS runs. In summary, our model predicts differentiated land use taxes to attain a desired soil loss target from the watershed. Using this methodology, the policy maker could in principle charge a differentiated tax for each zone, each land use and each conservation measure adopted.

6. Model Results and Policy Simulations

The model is run using GAMS-Minos5 non-linear programming solver. Scenarios are developed for differentiated tax policy and for uniform standard policies. We have also included various levels of labor market activities. However, the results happen to be close to the limited labor market problem. This surprising result is obtained due to capital constraint that is necessary for hiring labor. Our model predicts that at the existing wage rate, it is profitable to hire labor in rather than out, and had it not been for the capital constraints, more labor could have been employed and thereby increasing the profits to a level more than to the constrained case. It is also possible to solve the model by relaxing both capital and labor constraints to see the role played by these two markets in alleviating poverty. In addition relaxing such restrictions also imply an expansion in more labor-intensive crops. The required income transfer to reduce poverty could also decrease depending on how the impact of labor market and environmental policies interact.

Personal observations at the study site reveals that non-farm or of-farm wages tend to be much higher than the on-farm wages; however landing those off-farm jobs require additional skills. For example, the daily wages for laborers in the Banana Companies, located in the watershed, are about 160-185 Pesos, while our survey data reveals the average daily on-farm wages are about 60-85 Pesos. Thus hiring out at the current wage rate we imposed in our model (65 Peso per day) is not very attractive to the households. Thus the model could better represent reality if we also differentiated the wage offer function based on household characteristics such as education. This exercise is left for future research. Following this we will present the results from the various scenarios.

6.1. The cost of environmental protection:

In table 7, we present a sample of optimal tax rates for different land uses, slope classes and distance from the streams. Taxes increase with slope and decrease with distance from the streams. In addition, adopting soil conservation is associated with a lower tax rate. However, the private cost of environmental policies is measured in the reallocation of resources from the most profitable activities absent the policies. Table 8 presents the estimates of economic costs of the tax policies designed to attain various targeted abatement levels. The table shows the costs of policies that reduce the sediment delivered by 10% to 50%. The results reveal that when tax revenues are returned to households as a lump-sum rebate, environmental targets can be met at a modest cost to society. For example, a 10% reduction in sediment delivery requires less than a 1 percent loss of over all income, i.e. income including tax revenues. Even the 50% reduction is attained at only a 5.5% loss in overall income. This is due to the inexpensive conservation measures that could easily be adopted by households with an associated 50% reduction in soil loss. The P factor in the universal loss equation is equal to1 for no conservation, but 0.5 with the relatively inexpensive conservation measure. This implies, using the USLE, that without altering other choices including crop choices, it is possible to attain a 50% reduction in sediment deposited at the outlet of the watershed. Thus the cost of the policies is quite modest if the tax revenues are transferred to the households in the form of lump sum rebate.

However, the loss to households could be much higher, reaching up to 20 percent of base income, when the tax revenues are not returned to the households. The losses come directly from payment to the environmental authorities and indirectly through reallocation of resources away from more privately profitable activities. The losses reported in the table are only averages. Given the heterogeneity of farms, losses are expected to be much higher than the 20% for households operating on the most fragile sections of the watershed. Figure 2 and Figure 3 show upward sloping abatement cost and marginal abatement cost curves, respectively. As can be seen, the marginal abatement cost curve increases at an increasing rate. This implies that it gets more and more expensive to prevent additional units of sediment delivered to downstream areas as abatement effort increases.

The ratio of land area under hedgerows, by slope class and distance to the stream is presented in table 9. The results reveal for the most part that most highly erosive areas receive conservation measures, even though the results show some results contrary to this argument. However the shown in table 9 represent only those areas under hedgerow conservation measures excluding those areas under coffee and fallowing. This is also aggregated across elevation units, thus hiding the technological differences.

Uniform versus differentiated policies

We also compare the efficiency difference between uniform standards and the differentiated standards we discussed in our model. The uniform standard we analyze restricts the allowable soil loss per hectare to be the same across all zones. For a 10% abatement target for example, with uniform standards the cost saving in terms of efficiency amounts to 2110 Pesos per hectare for the case of a non-existent labor market, and slightly more (2150 Pesos) for the scenario where labor market functions. Thus the results show that even for a moderate reduction in target of sediment loss, targeting will have a significant impact on economic efficiency. The economic loss from a 10% abatement target using uniform standards is much larger than the economic loss from implementing a differentiated policy for a 50% abatement target. Even though the transaction costs could be high, there is clear potential for spatial targeting of tax or subsidy policies to manage the Manupali watershed. See table 8 for comparison of the effects of the scenarios we analyzed.

Table 7 presents the tax rate per hectare, for various geographical characteristics of the watershed. As can be seen from the table, for a given distance to the stream, the tax rate increases with slope of the land. By the same token, tax rates are lower for plots more distant from the stream. In addition, farms with conservation measures are required to pay less tax per hectare. Vegetables, the more highly erosive crop, are associated with the highest tax rate, other things equal.

7. Poverty versus the environment

Would it be possible to use environmental tax revenues to reduce poverty? Would it be possible to use efficient environmental taxes as a source of funding to alleviate poverty? The answers depend on a number of factors, including the severity of poverty in the watershed, the proportion of people below the poverty line and the incomes of people above the poverty line. In addition, the answers also depend on who pollutes the most in the watershed and on the structure of the tax instrument.

If poor households are also those that farm on the most fragile part of the watershed, ceteris paribus, it is unlikely that revenues from environmental taxes could cover their losses incurred as a result of the policy. If, however some of the most polluting households are also households with higher per capita income, then there exists a range of abatement targets that could lead to both environmental improvements and poverty reduction. In addition, larger percentage of people with income above poverty line increases the probability of attaining poverty reduction strategy using the proceeds from taxes.

In order to answer the questions empirically, we need several clarifying concepts. The first is how poverty itself is measured. We measure poverty as the aggregate income required to enable the poor to get to the poverty line. The second is the definition of poverty line itself. The UN determines $1 a day as a "conservative" poverty line; however, we have used the maximum per capita income of the lowest deciles of the survey households as the poverty line. That income is about 12,650 Pesos[3] per capita, much less than the recommended $1 a day, resulting in a highly conservative poverty measure. So our analysis looks at the possibility of lifting people who are in the tail of the income distribution. Our analysis compares the environmental tax revenue and the total income needed to bring every individual in the watershed to a level of income equal or greater than the poverty line. The results, presented in table 10, reveal such a possibility. The tax revenues from the 10% and the 20% abatement targets are sufficient to cover the costs of lifting the poor to the poverty line. With a 30% target though, the tax revenue is not enough to cover the loss of income, since the taxes both cause more people to fall below the poverty line, and exacerbate the severity of existing poverty. Thus, the results demonstrate that moderate levels of abatement targets can be met without increasing poverty; however, for higher abatement targets, poverty will increase when environmental taxes are used as the sole instruments. At high target levels of abatement, therefore, a tradeoff between poverty reduction and environmental protection is apparent. For more modest abatement targets, there is less conflict between the two targets—but then the environmental gains are also very modest. This tradeoff is an illustration of the “two targets, two instruments” dilemma; a policy designed to reduce pollution could also optimally reduce poverty only under exceptional circumstances.

Further research is needed to establish whether alternative environmental policies might more efficiently serve both policy objectives. Marketable permits, for example, are alternatives to taxes that lead to the same efficient environmental outcome. Marketable permits however do not require the knowledge of abatement costs of each individual. Because the available permits could be distributed arbitrarily without affecting the least-cost outcome, poverty considerations could be included as criteria to allocate the permits in such a way that more permits are given to the poor.

To summarize, when policy makers are concerned with both environmental quality and poverty alleviation, modest levels of environmental taxes could help reduce poverty. More ambitious environmental targets, however, mean that the income loss by the poor cannot be compensated. Therefore, for higher levels of abatement targets, when funds can be generated, environmental subsidies are the preferred instruments to address both poverty and environmental concerns.

8. Extensions

Dynamics

Throughout the paper so far, we have assumed that the decision makers (farmers) do not take into account the impact of their actions on future productivity onsite. The problem will be more realistic if we add the inter-temporal aspects of decision-making. However, if we include this

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[3] 1USD= 56 Peso

time dimension, the empirical model will become computationally cumbersome as the number of state variables to deal with will be very large. However, a sketch of a simple theoretical model could be presented. Theoretically, the model that incorporates both spatial variation and time could be solved in two stages. In the first stage, for a given time period, the spatial equilibrium will be solved. In the second stage, taking the optimal solution for the spatial dimension as given, optimization is done in the time dimension. There are only very few studies that have attempted to include both time and space together; one exception is a theoretical analysis for ground water pollution over time and space by Goetz and Zilberman (2000). Thus future research should consider both spatial and dynamic aspects of the problem.

Parameter uncertainty, asymmetric information, and transaction costs

We also believe that there is a need to use process-based watershed models to truly represent the contribution of each field to the downstream sediment delivery. Though simple, the USLE usually is said to overestimate the amount of soil delivered to downstream sites. In addition process-based models take account of the impact of runoff from adjacent upslope fields to soil erosion from that field. That means soil erosion depends on runoff coming from upper fields. Thus taking into account the interconnectedness of fields will make the model dynamic and of course add complexity in modeling policy. The problems with process-based computer simulation models are that they are not designed to be economic optimization models and that they are information intensive. The economy-environment model could be made more realistic with better representation of the pollution process.

Even when it is possible to use state of the art erosion models, there still remain other sources of uncertainty that could only be obtained through costly information collection. Enforcement costs will be high when policies are disaggregated according to spatial location. When production technology and other farm-level technologies are not known with certainty, policies based on such information will not be efficient. The literature on non-point source pollution has long acknowledged the significance of information asymmetry and transaction costs in determining the efficiency of alternative policies (e.g. Cabe and Herriges 1992; Helfand and House, 1995; Wu and Babcock 1996). Despite such acknowledgments, the empirical application of watershed management in the presence of information asymmetry and transaction costs is very limited. In practice, information administration and enforcement costs essentially require that the tax/subsidy base be truncated to a subset of choices that are both relatively easy to observe and correlated with ambient impacts, and tax rates that are relatively uniform within jurisdictions (Shortle and Horan 2001). Information problems and enforcement costs will even be magnified in developing countries due to weak institutions and the limited budget faced by regulators. The level of desegregation will thus depend on the costs of enforcement and information acquisition. Hence an optimal design of environmental policy takes into consideration the trade-offs between transaction costs and the benefits of disaggregating policy. In general when transaction costs are present, the deterministic efficient policy we have modeled will not be efficient any more. In general, the optimal level of desegregation will decrease as transaction costs of increase. Thus the need to extend this analysis to include transaction costs is very important in designing efficient watershed management in developing countries. We leave this exercise for future research.

Poverty

Finally, poverty analysis should be formalized by using explicit forms of welfare functions. The dilemma of environmental protection versus poverty alleviation could also be formalized within this context. Moreover, specifying the environment-economy model in general equilibrium would also a useful extension to this study.

9. Conclusions and Policy Implications

The Philippines faces the twin challenges of low levels of economic development and high levels of environmental degradation. It is important for the wellbeing of society that both challenges be addressed. In this study we use an environmental policy instrument and pursue its potential as a tool for simultaneous poverty alleviation. We find instances of complementarity between poverty reduction and environmental degradation, but typically the two objectives conflict. The real issue is to balance both. At the local scale, an economy-environment model of the kind developed here seems to present the most efficient way of attaining any environmental quality goal at the minimum cost to economic development. How to set the environmental quality target is the job of policy maker. However, the economy-environment model provides alternative efficient solutions given the policy maker's targets. If the objective is to protect the environment without considering implications for poverty or distribution, the model predicts that those farmers operating on highly erodible lands will have to pay higher taxes regardless of their income losses or initial poverty. In our empirical research, we have also shown that for moderate environmental abatement targets, revenues from differentiated taxes could cover the costs of alleviating the most severe poverty. When such possibilities exist, the policy has "double" benefits of protecting the environment with reduced poverty. For the most part significant reduction in soil erosion rate could be attained at a reasonably low cost to society. Given that households respond to economic incentives, efficient policies that result in least overall loss are crucial. In addition this study also has demonstrated the efficiency difference between uniform targeting and versus differentiated policies. When environmental outcomes could be estimable from observable choices, highly targeted policies are useful to minimize the costs of protecting the environment.

An alternative strategy to protect the environment, while at the same time shielding the poor from severe income losses, could be to allocate more erosion permits to the poor. The environmental objective will still be met at the minimum cost when transaction costs are not excessive. For example, in the Manupali watershed there is a mix of extremely diverse farm types, including small farms operated by very poor farmers producing for subsistence, middle sized farms mainly producing for market (big corn farms, sugarcane farms, vegetable farmers) and big commercial farms (banana plantations, pig farms, poultry farms). Thus, policies could be designed to make rich farmers pay the poor so that the poor adopt soil conservation measures. However, such kinds of policies could raise issue of fairness and thus could be difficult to implement, especially when law makers represent the well-to-do section of society. An alternative may be to create a market for environmental services and use the proceeds from their sale to pay the upland poor. Such initiatives are underway in many developing countries. One example, mentioned above, is the RUPES in South East Asia. Thus our model is capable of generating the most efficient subsidy/tax incentive to protect watersheds in developing countries when appropriate transfers can be designed to shield the poor from the impacts of environmental taxes. Of course, our results depend on the assumptions built into the model. Soil erosion estimates based on USLE overstate downstream impacts and thus the results put high taxes per unit of area planted. For this and other reasons the results of the analysis have to be qualified and caution must be taken before such policies are adopted for implementation. However, our use of household survey panels and GIS data sets means that many of the behavioral and technical parameters of the model are drawn from data rather than from assumptions, and in this respect our model is more robust than many of its predecessors.

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Table 1 Land area(in ha) by watershed and distance from national highway

	Alanib	Maagnaw	Tugasan
0-1km	1966.528	476.341	865.569
1-2km	1406.781	394.788	667.248
2-3km	676.515	357.719	237.244
3-4km	140.863	148.277	79.699
4-5km	0.000	3.707	0.000

Table 2 Glossary of Codes

Distance to Stream		Elevation		Slope	
Distance (m)	**Code**	**Elevation (m)**	**Code**	**Slope (%)**	**Code**
0-200	DS1	500-900	Lowland	0-10	SL1
200-400	DS2	900-1300	Middle	10-20	SL2
400-600	DS3	>1300	Highland	20-30	SL3
600-800	DS4			30-40	SL4
800-1000	DS5			40-50	SL5
>>1000	DS6				

Table 3 land area (in ha) by elevation and slope

Elevation (masl) ↓	Slope (%) →	0-10%	10-20%	20-30%	30-40%	40-50%
500-700		231.683	25.949	5.560	1.853	5.560
700-900		1060.183	444.832	48.190	7.414	1.853
900-1100		1017.553	620.911	205.735	57.457	18.535
1100-1300		1288.159	861.862	205.735	46.337	14.828
>1300		322.503	531.945	202.028	124.182	37.069

Table 4 Land Area (in ha) by slope and distance to streams

Distance to stream (m) ↓	Slope (%) →	0-10%	10-20%	20-30%	30-40%	40-50%
0-200		1207.013	760.794	264.996	95.740	54.709
200-400		1208.722	641.119	152.159	37.612	13.677
400-600		736.859	463.315	95.740	13.677	6.839
600-800		441.090	266.705	76.934	10.258	6.839
800-1000		193.190	162.417	49.580	10.258	8.548
>1000		129.933	164.126	70.096	11.968	3.419

Table 5 Household characteristics by watershed attributes

Distance to stream	Farm size	Share of Coffee area	Share of Vegetables area	Share of Corn area	Education	Family Size
0-200	2.97	0.12	0.17	0.41	7.38	6.27
200-400	2.65	0.14	0.19	0.36	7.37	6.23
400-600	2.63	0.18	0.20	0.33	7.64	6.15
600-800	2.64	0.21	0.21	0.32	7.68	6.12
800-000	2.79	0.25	0.20	0.29	7.98	6.06
>>1000	2.62	0.36	0.22	0.23	7.95	6.16
Elevation	**Farm size**	**Share of Coffee area**	**Share of Vegetables area**	**Share of Corn area**	**Education**	**Family Size**
Low land	5.57	0.27	0.00	0.44	9.35	5.65
Middle	3.43	0.19	0.11	0.40	7.58	5.99
Upland	1.74	0.19	0.28	0.27	7.35	6.40
Slope (%)	**Farm size**	**Coffee area share**	**Vegetables area share**	**Share of Corn area**	**Education**	**Family Size**
0-10	2.99	0.21	0.17	0.34	7.80	5.80
10-20	2.96	0.21	0.17	0.34	7.75	6.00
20-30	2.66	0.19	0.20	0.33	7.55	6.29
30-40	2.32	0.16	0.23	0.33	7.25	6.59
40-50	2.14	0.18	0.27	0.29	7.55	6.91

Table 6 Production function parameters

	Corn		Vegetables		Coffee	
	Coefficient	P>t	Coefficient	P>t	Coefficient	P>t
Log(land)	0.590	0.000	0.539	0.000	0.625	0.000
Log(labor)	0.202	0.000	0.439	0.000	0.298	0.000
Log(fertilizer)	0.034	0.000	0.071	0.000	0.032	0.053
Log(slope)	0.095	0.183	0.088	0.611	-0.073	0.677
Log(D_Hwy), km	-0.022	0.577	-0.227	0.009	0.123	0.151
Log(D_stream), km	0.063	0.070	-0.144	0.151	0.108	0.075
Log(elevation), km	-0.240	0.256	2.087	0.054	-1.082	0.045
Variety (Dummy)	0.180	0.048				
Log(Education), Yr	0.055	0.123	0.033	0.676	0.024	0.705
Year98 (Dummy)	-0.493	0.000	0.115	0.666	-0.247	0.223
Intercept	-0.930	0.001	-0.512	0.459	-2.710	0.000

Table 7 Tax rates by distance to stream and slope for a 10% abatement target

Land Use ↓ | Slope(%) →	10%	20%	30%	40%	50%
	Tax/Ha ('000 peso) for 200 m distance to stream				
Corn	1.16	2.89	4.94	7.24	9.74
Corn, Hedge Rows	0.58	1.44	2.47	3.62	4.87
Vegetables	1.74	4.33	7.41	10.86	14.61
Vegetables, Hedge rows	0.87	2.17	3.71	5.43	7.31
Coffee	0.23	0.58	0.99	1.45	1.95
	Tax/Ha ('000 peso) for 1000 m distance to stream				
Corn	0.65	1.61	2.75	4.02	5.41
Corn with Hedge Rows	0.32	0.80	1.37	2.01	2.71
Vegetables	0.97	2.41	4.12	6.04	8.12
Vegetables, Hedge rows	0.48	1.20	2.06	3.02	4.06
Coffee	0.13	0.32	0.55	0.80	1.08

Table 8 Costs of environmental policy without transfers

Abatement Target	Income (000 Peso)	Total tax Revenue	Total HH Income	Total cost (% income)	Cost on HH (% income)
0%	198312	0	198312	0.00	0.00
10%	197000	13400	183600	-0.66	-7.42
20%	195400	14258	181142	-1.47	-8.66
30%	193300	15281	178019	-2.53	-10.23
40%	190900	16012	174888	-3.74	-11.81
50%	187500	29262	158238	-5.45	-20.21
Uniform, 10%	181600	23035	158565	-8.43	-20.04

Table 9 Ratio of land area with conservation measures for selected locations

Distance to Streams	Slope ► / Abatement target ▼	SL1	SL2	SL3	SL4	SL5
DS1	10%	0.38	0.19	0.33	0.41	0.51
DS1	20%	0.38	0.32	0.33	0.41	0.68
DS1	30%	0.48	0.49	0.71	0.73	0.73
DS1	40%	0.48	0.75	0.83	0.85	0.80
DS1	50%	0.84	0.86	0.82	0.67	0.46
DS3	10%	0.00	0.12	0.30	0.00	0.00
DS3	20%	0.51	0.33	0.44	0.00	0.00
DS3	30%	0.51	0.57	0.45	0.00	0.00
DS3	40%	0.73	0.65	0.68	0.66	0.70
DS3	50%	0.80	0.78	0.77	0.83	0.70
DS5	10%	0.00	0.31	0.42	0.00	0.00
DS5	20%	0.26	0.40	0.42	0.00	0.00
DS5	30%	0.61	0.46	0.42	0.00	0.06
DS5	40%	0.61	0.46	0.64	0.62	0.68
DS5	50%	0.69	0.69	0.66	0.72	0.68

Table 10 Poverty and environmental taxes with targeted transfer

Abatement Target ➔	**10%**	**20%**	**30%**
Tax revenue (‘000 Peso)	1379	1509	1632
Additional income required	803	969	5372
Possible to alleviate poverty	Yes	Yes	No

Figure 1 Manupali Sub-Watersheds

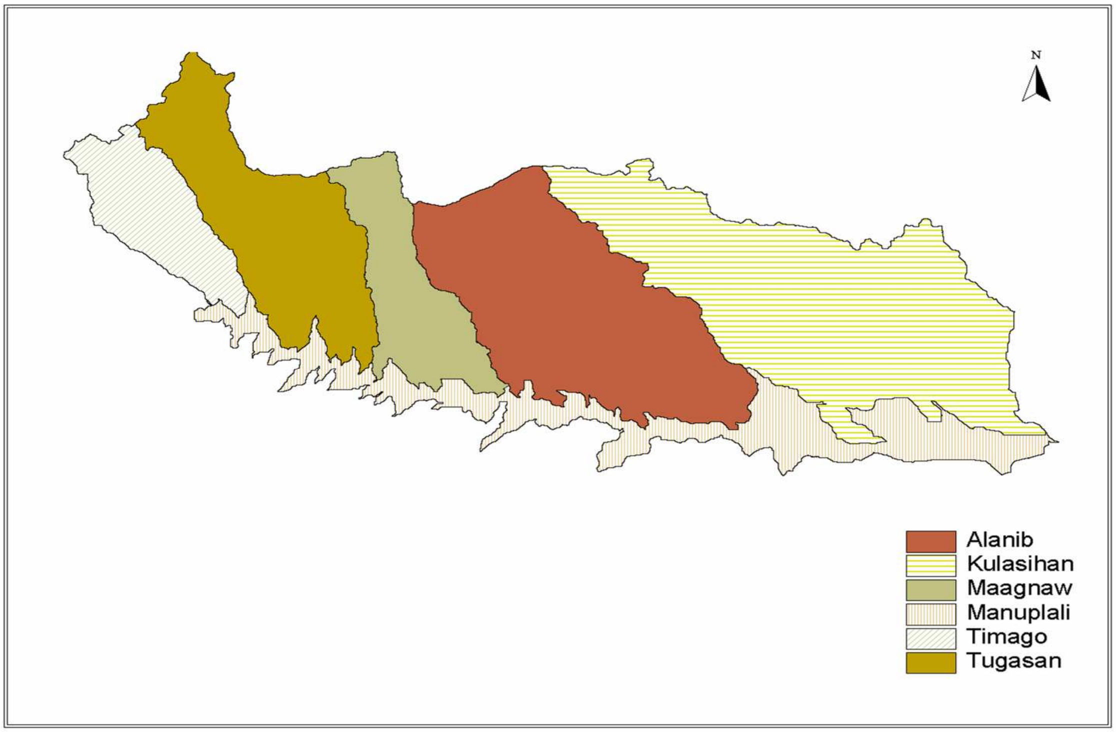

Figure 2 Total abatement cost ('000 Peso)

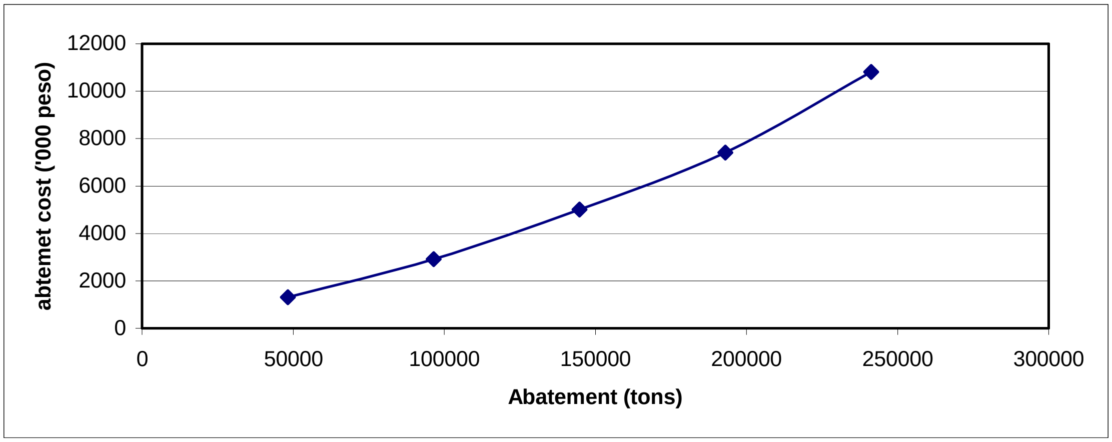

Figure 3 Marginal abatement cost ('000 Peso)