Valuing Soil's Economic Worth

Matthew Oliver Ralp Loria Dimal

VALUING SOIL'S ECONOMIC WORTH

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"The soil is the great connector of lives, the source and destination of all. It is the healer and restorer and resurrector, by which disease passes into health, age into youth, death into life. Without proper care for it we can have no community, because without proper care for it we can have no life."

Wendell Berry (1977)

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Chapter 1. **Introduction to Soil Value**

1.1. **Introduction**

Soil is an essential resource with diverse ecological functions and socio-economic contributions. But due to abuse and mismanagement, coupled with the increasing demands from conflicting usage, it has been under threat from being substantially degraded (Pimentel 1993, Lal 2014). Much research has been conducted on modeling soil degradation and improving conservation technologies, but it remains to be an enormous global problem (Pimentel 2006). Soil degradation can be a naturally occurring phenomenon caused by biotic and abiotic agents, but the increasing rates of degradation have been associated mainly to anthropogenic land-cover changes. Intensification of agricultural activities has extensively augmented food production but concomitantly exacerbated environmental problems (Albizua, Williams et al. 2015). Uncontrolled and unmitigated, soil degradation becomes a calamitous concern, which threatens global food security and results in various economic costs including increased flood frequency, greater risk for landslides, and sedimentation of the rivers and reservoirs (Bandara, Chisholm et al. 2001, Kabir, Dutta et al. 2011, Pimentel and Burgess 2013, Lal 2014).

The problem becomes even more complex for developing countries (Thapa and Weber 1991, Middlebrook and Goode 1992). Underdeveloped economies often characterized by heavy dependence on the agricultural industry are compelled to favor profitability and production over sustainability and environmentalism. Due to incapacity to finance conservation measures, marginalized communities are most vulnerable to the economic impact of soil degradation. Unsuitable government policies, detrimental consequences of technological change, and weak institutions are also largely to blame for the prevalence of soil degradation in the developing world (Ananda and Herath 2003). Thus, the deterioration of soil resources should be contextualized not just in its environmental and political features, but also with regards to economic, technological and social aspects (Bandara, Chisholm et al. 2001). Responding to the challenges of long-term sustainable soil management requires collaborative action from both government and local communities. Policy distortions, market failures and the paucity of stakeholder participation could cripple efforts toward sustainably managing soil resources (Tomich, Chomitz et al. 2004). Understanding the numerous contributions of soil to society is essential to complement them with appropriate policies and sufficient investment expenditure.

There has been rising interest to integrate economics with environmental and policy science in resource management. This has been buoyed by the urgency for stronger policies in support of broader ecological protection, especially those that highlight human dependence on well-functioning ecosystems (Nestle 2008, Salles 2011). But without an agreed-upon measure to evaluate the economic aspect of conservation and ecology, people have been less-accepting of regulating sustainability, especially when brought against maximizing profit. It is therefore imperative that a credible and comprehensive soil valuation process would be constructed that would provide realistic and normative value estimates of soil contributions.

But the process of determining the economic worth of soil is complex and multi-faceted (Adhikari and Hartemink 2016). Soils are equivocally one of the most complex Earth systems that are intrinsically connected with biodiversity, climate change, and the health of the broader environment (Haygarth and Ritz 2009). Its ecological functions and environmental services are often unrecognized and not well understood (Dominati, Patterson et al. 2010). As an economic resource, soil performs a variety of roles and functions. Aside from the multiple soil amenities directly benefiting private individuals, soil provides a broad range of public service to the broader community. Soil found in private property is characterized as a natural constituent of land; the use and management of soil are left to the discretion of private individuals. However, the loss of soil's indirect utilities and the impact of degradation impact the whole community who bear much of the social cost (Ananda and Herath 2003). In this sense, soil as a resource cannot be treated simply as a private good but must be assessed in the context of being a public good.

In resource accounting, soil occupies a unique typology of environmental products. Although erosion is a naturally occurring process, soil is organically renewed and regenerated back into the system. An ecological equilibrium between lost and created soil material is formed, which essentially makes soil as a renewable resource. But when large-scale degradation occurs, soil is transformed into a non-renewable resource, often with irreversible effects on land fertility and economic productivity. Accelerated erosion is defined by an unnatural increase in soil loss caused mainly by anthropogenic disturbance (Alexander 1988). Various human activities have increased the incidence of soil erosion, such as deforestation, intensified farming operations, over-grazing, and construction activities (Terranova, Antronico et al. 2009 & Iaquinta, 2009). The harmful consequences of accelerated erosion transpire not only on-site where detachment occurs, but also on the lowlands and water systems where sedimentation takes place (Bandara, Chisholm et al. 2001). Erosion adversely impacts soil quality and fertility, by decreasing available nutrients and organic matter in the soil (Pimentel 1993). The loss in organic matter causes deterioration in soil structure and infiltration rate, decreases water retention, and leads to the reduction of plant-needed nutrients such as N, P, K, Ca and Mg (Lal 1993). Downstream areas are also adversely affected by soil degradation. Reservoir sedimentation, disruption of ecosystems, and water contamination are just some of the offsite consequences induced by soil erosion. The complexity of the nature of soil and its dynamic economic roles make the valuation process perplexing and abstruse.

Even with the general acceptance of the correlation between soil health and anthropogenic benefits, success in the sustainable use of soil has oftentimes been elusive. A major contributory factor has been traced to the lack of understanding and appreciation of the economic contributions of soil in the various aspects of human wellness. Not knowing the soil's true worth has resulted in lower priority being given to soil in the decision-making table, and poorer stakeholder participation in the conservation measures. The purpose of this chapter is to review and discuss the complexities of valuing soil, and lay the foundations in the development of standard frameworks for soil valuation process. The relation of soil as a natural capital with its economic value is introduced, based on how ecosystem services and environmental goods have been defined in the developing literature. A critical assessment of how these different valuation frameworks can be used in soil value estimation is offered, including how they measure up to their intended applications. An integration scheme to enhance the valuation framework for soils is also presented using participatory modelling and possible applications for the approach are discussed.

1.1.1. Soil as an Environmental Public Good

To understand why soil's economic value is not intuitively intimated, we first need to understand the concept of environmental public goods (EPG). Environmental public goods, which include soil resources, are naturally occurring products that provide a number of direct and indirect benefits. These economic goods are characterized as being non-subtractable and non-excludable. Excludability refers to the ability to restrict access or right of use, while subtractability indicates the rivalry of consumption wherein the use one diminishes the ability of another. Private goods, which are both subtractable and excludable, are inherently valued by intervening market forces as dictated by utility, and supply and demand curve.

For soil and other EPG, due to their non-excludability and non-subtractability attributes, estimating their economic worth is more complicated. Its non-excludability results in a 'free-rider syndrome,' reflecting the people's unwillingness to pay for their portion of costs when the good is communally enjoyed (Boadway, Song et al. 2007). Most EPG have no developed markets to determine the benefits derived by each household, which results in the underestimation of the EPG's actual worth (Engel, Pagiola et al. 2008). 'Market failure' is the economic concept referring to the inadequacy in regulating and optimizing the transaction of goods, leading to underproduction or the exploitation of the market (Willis and Garrod 2012).

The primary sources of market failure for soils come from its relative abundance, imperfect and weak property rights, and the insufficiency of complete information. Soils have been viewed not as a distinct natural capital (global stock of natural assets) but merely as a component of an ecosystem or a quality indicator of land. Many soil amenities, especially those that benefit the general public, were often precluded which have led to misconceptions about soil worth. Prices have been used by the market to communicate scarcity to assess utility trade-offs and optimize resource allocation (Schlapfer, Roschewitz et al. 2004). But because of their abundance in relation to the population demand, soil and other EPG have no automatic mechanism to assign value for the benefits derived (Boadway, Song et al. 2007, Ulgiati, Zucaro et al. 2011). This perception of having zero-value lays the foundation of the seeming disconnect between economics and the environment, and reflective of the people's unwillingness to pay for their portion of costs. The problem is confounded further by the presence of price externalities, which are costs and benefits generated as by-products of economic activity but not reflected in transacted prices. A comprehensive and well-defined process of soil valuation exposes most externalities, providing a much clearer on the importance of soil to personal welfare. Furthermore, it allows an avenue for communicating the economic impact of soil use and conservation, which could be useful both in promoting participation amongst stakeholders and advocating for more sustainable policies.

1.1.2. **Environmental Public Good**

In classical economics, environmental goods often were categorized in distinct and separate categories because they benefit the general public, and oftentimes at no cost. Its fundamental nature of non-excludability often causes a free-rider syndrome, wherein people consume more than their fair share because of a lack of mechanism to control their appetite. Prices have been used by the market to communicate scarcity to assess utility trade-offs and optimize resource allocation (Schlapfer, Roschewitz et al. 2004). But because of their abundance in relation to the population demand, most public resources have yet to be assigned value (Ulgiati, Zucaro et al. 2011). Most of these public goods have no developed markets, and thus no automatic mechanism to determine the benefits derived by each household (Boadway, Song et al. 2007, Engel, Pagiola et al. 2008). This perception of having zero-value lays the foundation of the seeming disconnect between economics and the environment, and reflective of the people's unwillingness to pay for their portion of costs. The problem is confounded further by the presence of market failures, or the inadequacy of the market to regulate the transaction of goods, which often leads to underproduction or exploitation. In the context of soil, market failure can be due to price externalities (costs and benefits generated as by-products of an economic activity but are not reflected on transacted prices), collective utilization of land, imperfect or weak property rights, absence of perfect competition, or the inadequacy of perfect information among stakeholders.

Accurate assessment and recognition of the economic contributions of soil and other public goods are important in promoting more sustainable use of environmental goods. EPG's significance to human existence has often been overlooked, which have led to their exclusion from the decision-making process (Plantier-Santos, Carollo et al. 2012). Soil's value has often been entwined with the price of land, which emphasize private benefits to landowners but fails to consider the numerous public benefits and possible social costs of degradation. Without an agreed upon measure of value for evaluating economic, normative, and conservation actions, governments have been passive in correcting these market failures, and people have often been less-accepting of restrained use especially when faced against maximizing profit. This knowledge-gap has resulted in inefficient land-use policies, creating a distorted picture of their economic value, and ultimately to the mismanagement and exploitation of natural resources.

1.2. Developing Soil Valuation Frameworks

In order to grasp soil's actual worth, a credible framework is required to explicitly link various ecological functions into soil amenities that directly or indirectly contribute to human well-being. To have a comprehensive approach, the framework would require the integration of a variety of disciplines such as ecology, economics, soil science, spatial statistics and physical modeling. The central goal would be providing a pecuniary estimate for the value of soil amenities, that could serve as economic assessment for alternative soil usage or policy initiatives.

Similar to the valuation frameworks of other ecosystems and environmental goods, the framework for soil valuation requires that the methodology and approaches within the system are well-grounded on theoretical principles of environmental economics. This would guarantee that the process remains credible and would result in realistic estimates. Chee (2004) lists four vital economic concepts relevant in the formation of valuation frameworks (definition of terms found in **Table 1.1**). These include the following: (a) market essentialism; (b) substitutability, fungibility, and technological optimism; (c) rational actor and consumer choice theory; and (d) utilitarian, anthropogenic and ethical framework. These essential elements are necessary to ensure that the valuation approaches yield reliable and unbiased results and that any new framework will remain to be objective-centric and systematic. The credibility of the valuation technique has to be ascertained as to merit public and institutional acceptance, and not merely become a biased poster-child on the issue of conservation versus profit.

To understand how soil value is estimated, we begin with the basic sequential diagram for assessing soil resources (see **Figure 1-1**). As earlier stated, the valuation framework espouses an anthropocentric and utilitarian argumentation. This means that only the soil functions that provide direct and indirect benefits to humans would be assessed. These ecological functions that that contribute to the different aspects of human wellness are referred to as ecosystem services (ES). Different market-based and non-market based approaches would then be used to translate these amenities into economic value, depending on the service-type. For many soil services, proxy indicators would have to be used to determine soil value since they are not transacted in the current marketplace. Some proxy indicators that could be used include the following: implicit expenditure for the use, consumption or access of a similar good; the amount people are willing to pay for the continued use or access of the soil service; and, the cost needed to rehabilitate or avoid the adverse impact of loss of a soil amenity.

External variables affecting the use and value of soil should also be considered in the valuation process. A number of factors could significantly influence soil value, including: the stakeholders (population that are directly and indirectly affected by the utilization of soil), policies (laws and programs associated with soil use and management), market forces (supply, demand and the changing utilities of soil), and the environment (the exogenous environment and ecosystem). This highlights the importance of contextualizing soil value not just through an environmental periscope but also understanding the underlying socio-demographic, political and economic aspects. The results of the valuation can be utilized in a number of useful applications. Valuation can be used to modify stakeholder cognition and behavior to promote soil conservation and sustainability, especially in farm operations. Policies (e.g., subsidies, new taxes) can be modified to correct for externalities and market failures that are often linked with overlooked soil services. New forms of social contract between social winners and losers (service beneficiaries and producers) have been constructed based on environmental valuation. Payments for ecosystem services (PES) actively incentivize the protection of soil resources to promote more environmental benefits and discouragement of environmentally detrimental activities (Chen, Lupi et al. 2012). Financial resources or in-kind payments are usually awarded to beneficiaries to guarantee the continued provision of specific ecosystem services, such as biodiversity, carbon sequestration, landscape beauty, and watershed protection (Muñoz Escobar, Hollaender et al. 2013). Other applications for soil valuation include the creation of new property rights, inputs for environmental accounting, and the long-term strategic planning in zoning and land use allocation.

Table 1-1. Defining the Essential Elements in Valuation

Concept	Definiton
Market essentialism	Contextualizing environmental services in the marketplace
Substitutability	Availability of suitable surrogates to associate nature-derived benefits some value using comparable benefits
Fungibility	Adequacy and sufficiency in supply of substitutes
Technological optimism	Belief that foreseeable growth in demand would be answered by advancement in technology
Rational actor	Economic behavior described as wanting to have more rather than less of a certain good
Consumer choice	Consumer preferences and expenditures are driven by motivation to maximize utility, based on limitations of budget
Utilitarian and anthropogenic	Man-centric valuation which focuses on estimating value based on the various utilities that satisfy man's needs
Ethical Framework	Environmental goods have intrinsic value outside the conventional utilitarian definition

(adapted from Chee, 2004)

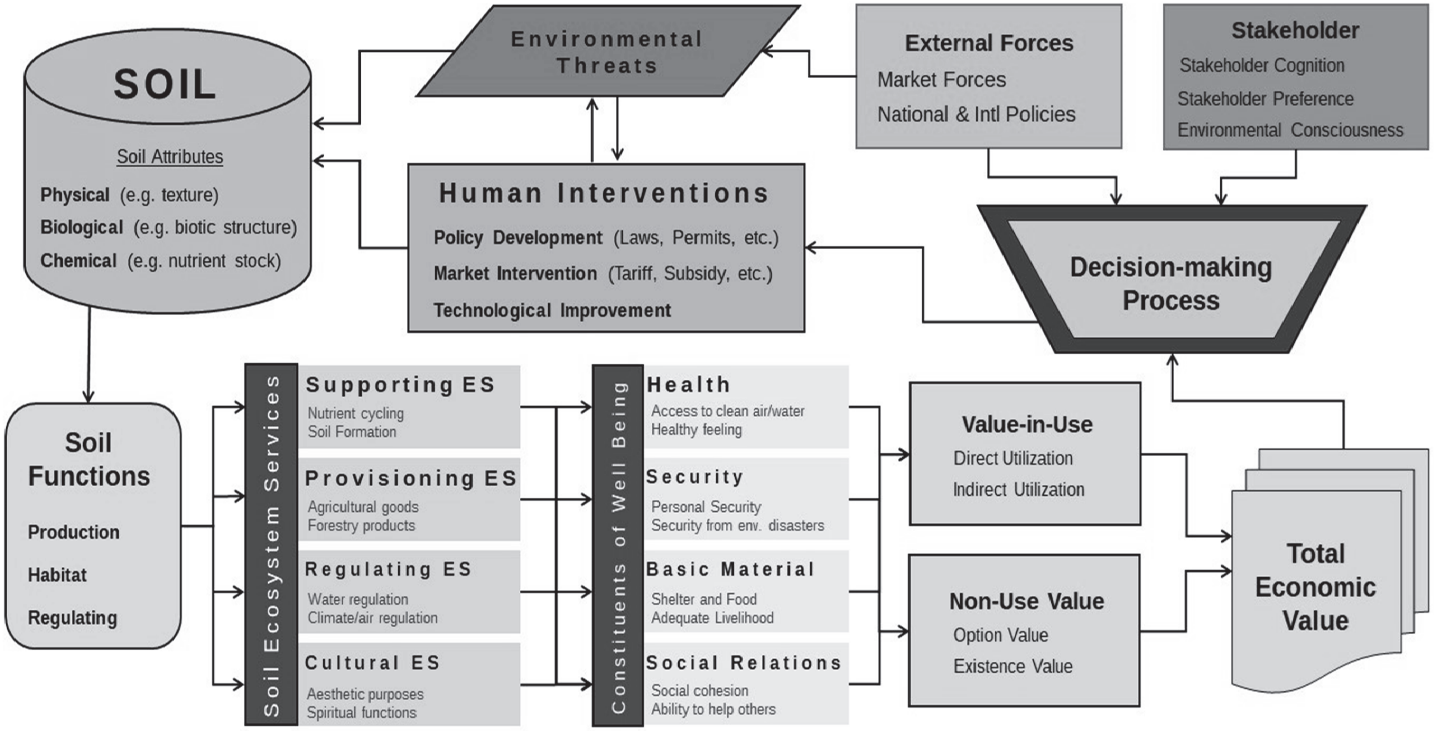

Figure 1-1. Fundamental interaction of economic value of environmental resources with environmental and anthropogenic systems

1.3. Different Frameworks for Soil Valuation

There have been a number of conceptual frameworks proposed to provide a comprehensive assessment of economic value to the environment. Four of the most commonly used valuation frameworks, which have also been applied to assess soil value, are discussed in this section. These are the ecosystem services approach, the flow-fund approach, the cost-based assessment approach, and the total economic value approach. Ecosystem Services Approach

1.3.1. Ecosystem Services Approach

One of the most prominent valuation approaches is the MEA framework (presented in **Figure 1-2**). Named after the Millennium Ecosystem Assessment (2005), this approach focuses on the processes and conditions in nature that directly or indirectly fulfil human satisfaction called ecosystem services (ES) (Fisher and Turner 2008). ES include the production of goods, delivery, and transport, regulating and regeneration, protection and maintenance, and other life-supporting services to humans and other living creatures (Chee 2004). The MEA framework was one of the first to extend the idea of environmental services into a heuristic classification system for value assessment.

The MEA framework classifies services into four categories: (a) provisioning, (b) regulating, (c) cultural and (d) supporting (Millennium Ecosystem Assessment 2005). Provisioning ES are the tangible and the most readily perceived ES. These are mainly private commodities derived from the environment which have their own markets and pricing mechanism. Agricultural and timber goods are examples of the soil provisioning ES. Regulating ES include the processes that provide ecological maintenance such as climate regulation, water purification, waste treatment and protection from natural disasters. Cultural ES are the non-physical amenities that relate to the fulfilment of man's spiritual and cognitive needs. Examples of soil cultural ES are aesthetic values, cultural heritage and diversity, and leisure needs. Supporting ES are the processes that provide assistance to the other services. These amenities often impact man indirectly and are measured over long periods of time. Nutrient cycling and soil formation are examples of soil supporting ES.

While the MEA approach has become the most dominant valuation framework, it has been criticized to perpetuate double counting of environmental benefits (Boyd and Banzhaf 2007, Fisher, Turner et al. 2009). Double counting results from overlapping services being valued twice which creates an overestimation of economic value. Some have recommended the exclusion of the supporting services in the assessment of services and instead focus only on the other three categories (Maynard, James et al. 2010, Ojea, Nunes et al. 2010, Chiabai, Travisi et al. 2011). But this could also lead to gross undervaluation specifically for supporting services that do not have associated regulating or provisioning ES. Some have suggested that while the MEA framework provides an adequate approach to understand the types of services environments provide, its direct application towards valuation can be counter-productive (Ojea, Martin-Ortega et al. 2012). They note that output-based approaches (e.g., fund and flow, TEV) provide better disambiguation of economic value and averts the risks of double counting.

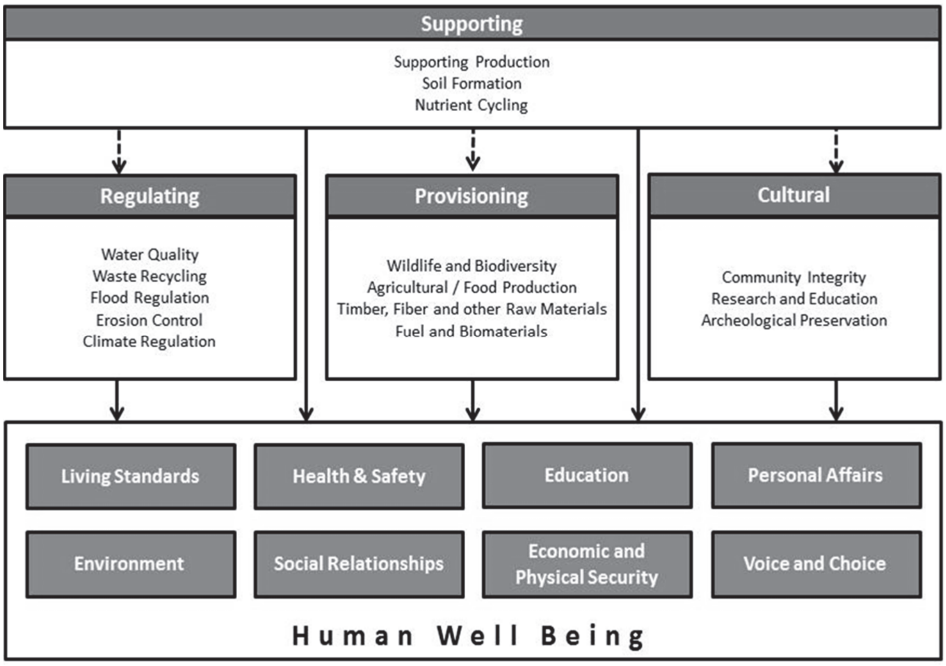

Figure 1-1. Soil services and their relations to components of human well-being

1.3.2. **Fund and Flow Approach**

Natural capital is defined as "the stock of materials or information contained within an ecosystem" (Costanza, dArge et al. 1997) that enables the production of goods and services that are converted into wealth and well-being (Hinterberger, Luks et al. 1997). The features of natural capital correspond to the functions of a transformative fund or a source of material flows. This approach is called the stock flow and fund service (fund/flow) framework (presented in **Figure 1-3**). It focuses on the earth-system management of resources, differentiating between the tangible and intangible goods, and recognizes that the final classification is based on utility (Robinson, Hockley et al. 2013).

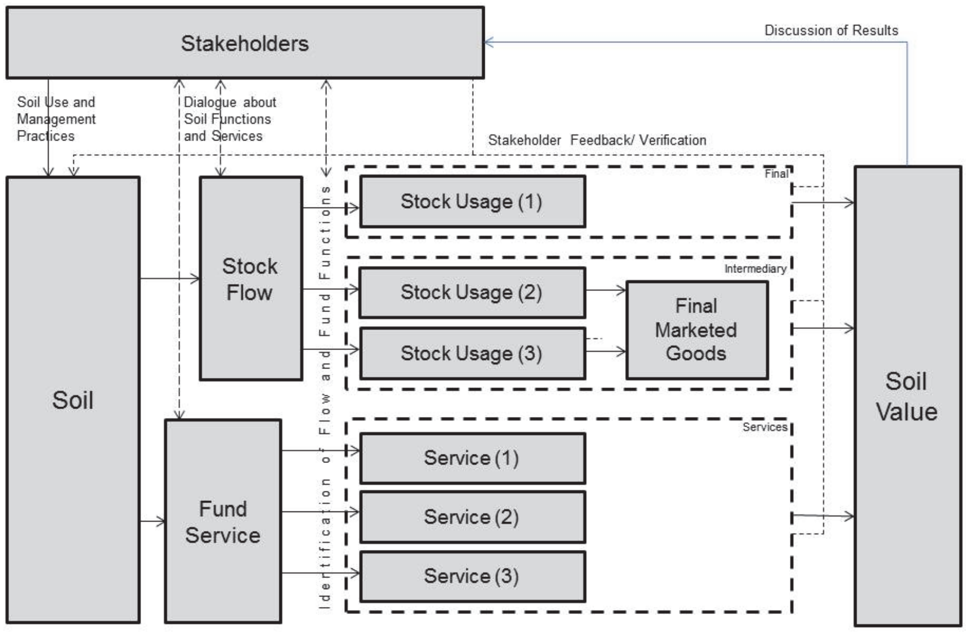

Figure 1-2. Framework using flow/fund approach for valuing soil

There is value in identifying and separating the fund/flow roles of environmental goods. Whereas the fund service is entirely utilized at any given moment but does not depreciate from usage, stocks are discretely utilized and depleted based on consumption needs (Kraev 2002). Soil and other environmental goods play both roles as a stock source and a fund service, the tasks are distinct and are treated differently. Soil can be viewed as a natural stock from which goods can be obtained or produced such as agricultural and timber products, as well as a fund of ecological services including climate regulation and water purification. Some contend that the use of the natural capital stock provides a better elucidation of economic value than the MEA's (2005) concept of ecosystem services (Boyd and Banzhaf 2007, Wallace 2007, Robinson, Hockley et al. 2013). By focusing only on final services which provide direct benefits, ambiguity in the benefits being valued is significantly reduced which in turn diminishes the risks of the double counting. While the critics of the MEA framework concur that intermediate products are themselves valuable, their value should only be embodied in assessing final ecosystem services (Boyd and Banzhaf 2007).

The valuation of the environment would therefore either be as a function of the service-providing fund's value or in terms of the rate of a change of the stock (Kraev 2002). For soils, if we would view it as a stock-source, it would be considered as a provider of nutrients and platform from which agricultural products grow. The valuation would then be roughly based on the amount of agricultural yield, and in relation to the change in the nutritional content of the soil. If we would look at soil's fund service, for example, its capacity, this soil's contribution would be based on the value of the total water purified.

The core principles and fundamentals of the fund/flow approach are deeply rooted in mainstream economics, and often provides a more conservative estimation of economic value. This substantive inkling towards conventional economics is where many of its critics base their objections. The primary argument is that by only focusing on final products, the estimates would be a significant underestimation of environment's real value, which could be counterintuitive for environmentalism. Behavior towards environmental use may be skewed in favor of production and profitability when intermediary ecological functions are excluded in assessment. It also excludes much of the socio-cultural benefits arising from the environment. These criticisms create areas of further research for those supporting the fund/flow approach, especially in the context of valuing soil resources.

1.3.3. **Cost-Based Assessment Approach**

The cost-based assessment provides a valuation framework that focuses on the capacity of a healthy environment to prevent natural disasters, minimize environmental risks, and avoid the disruption of services. The basic framework for the cost-based assessment is shown in **Figure 1-4**. This approach is a pragmatic way of overcoming perception bias that continued use and Similar

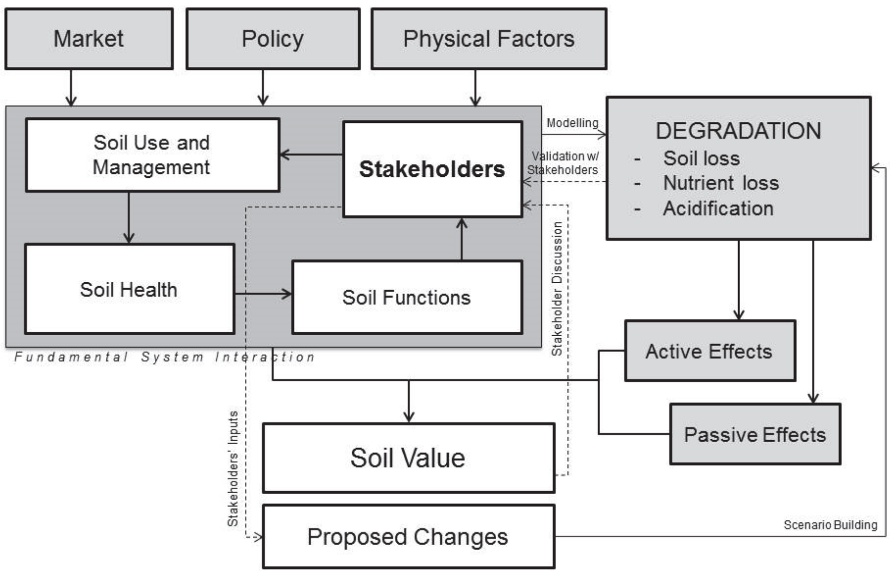

Figure 1-3. Cost-Based Soil Valuation

consumption of environmental services will always be available, consequence-free, and without financial costs. Economic value is estimated based on the costs of services that could potentially be lost or reduced. This approach examines the passive and active effects of soil degradation. Passive effects include latent economic consequences of soil degradation that tend to manifest gradually, such as reduced agricultural productivity from nutrient deficiency. Active consequences are mainly the soil-related natural hazards that can potentially disrupt human safety or security, such as flooding, landslides and river pollution.

Similar to the idea of evaluating risks in financial transactions, soil and other environmental goods can be valued using a variety of pricing mechanisms. Two common cost-pricing schemes are the replacement cost (RC) and damage-cost avoided (DCA). These valuation techniques very much related, such that instead of pricing how much people are willing to pay for specific services, the value is based on either the cost of replacement or prevention.

The DCA value is based on the costs needed to prevent the loss or reduction of supply due to soil degradation. These defensive expenditures are considered to provide much lower value estimates since preventive measures are generally inexpensive and most economical. For example, the loss of topsoil from exacerbated erosion can be prevented through conservation measures such as reforestation of upland areas, creating drainage infrastructure to minimize overland flow or household-level implementation of sustainable farming practices. The DCA value can change depending on the projected risk of soil degradation, which is highly dependent on land use, physical factors and various anthropogenic factors.

Another pricing scheme that can be used in the cost-based assessment is the replacement cost (RC), which estimates the value of environmental damage according to the price that would be needed to restore the environment from its previous undamaged state. The costs of rehabilitating the upstream farmlands, restoring the downstream ecosystem, dredging sediment-filled reservoirs and decontaminating polluted water supplies are some examples of replacement cost for soil amenities. A modification of the RC value uses the expenditure of shadow projects that can provide a commensurate alternative to the services that would be lost due to degradation. This has been suggested for areas that have reached high levels of degradation that rehabilitation is not feasible or financially impractical. The use of RC to establish value has been criticized whether it is truly reflective of the environmental damage that it aims to assess. Arguments against the use of the RC say that it provides a myopic understanding of environmental degradation and that it does not consider many of the different ancillary ecological services. Again, this could be counter-intuitive with environmentalism, and can even create a misperception regarding complete substitutability of environmental value.

1.3.4. Total Economic Value Approach

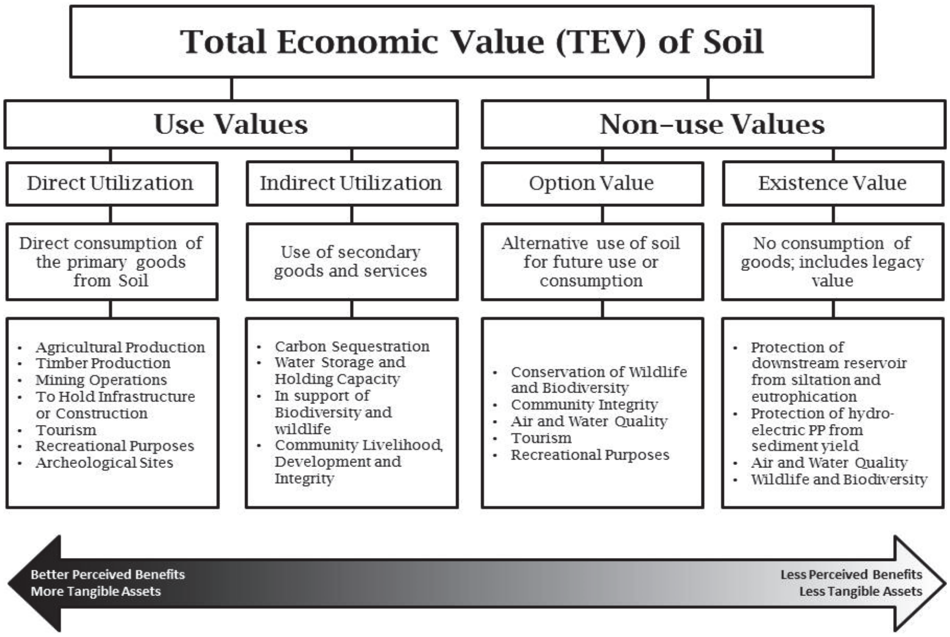

Figure 1-4. Components of total economic value for soils

The concept of the total economic value (TEV) has been widely used to provide the utilitarian estimates of ecosystems (Sarukhan and Alcamo 2003). It utilizes a functional approach that aggregates the use (active) and non-use (passive) values derived from soil services that directly and indirectly benefit human well-being (Gomez-Baggethun, de Groot et al. 2010). The components of the soil's TEV and the relevant services associated with these components are presented in **Figure 1-5**. The main advantage of the TEV framework is that it identifies and distinguishes both the tangible, direct amenities and the less apparent ecological services. It features the patrimonial significance and irreversibility concerns of environmental protectionism by integrating the non-use values alongside the traditional use value (Plottu and Plottu 2007). It reinforces the ecological argumentation of nature's intrinsic worth but still using an anthropogenic argumentation in establishing value.

The TEV framework is comprised of two main components: the use value and the non-use value. The use value determines economic worth from the utilization or consumption of the environmental good, which can be partitioned into being of direct use and indirect use. Direct use pertains to the direct utilization of the resource, which is often associated with commodities or marketable products. The direct use-value may either be consumptive of goods) or non-consumptive (does not affect quantity). Soil fertility which contributes to agricultural production is an example of direct use value. For most private goods, their total value is almost equal to the aggregated direct use values (Birol, Koundouri et al. 2008). But for soil and other environmental goods, they oftentimes perform other roles that do not necessarily produce marketable outputs but provide vital service towards the common good. The value arising from these benefits is called indirect use-value. Indirect use-values pertain to goods and services that are used as intermediary inputs for production and are associated with the ecological aspect of analysis. Some of the soil's indirect benefits include climate regulation, water quality regulation, and water storage. **Table 2-1** lists some of the soil's use values and summarizes some methodology and pricing mechanism that can be used in assessing value.

Table 1-2. Suitable methodology, pricing mechanism and data requirements for main soil ecosystem services

Soil ES	Methodology for Value Assessment	Pricing Mechanism
Production of agricultural and forest products	Market pricing and production function models could be used to estimate price and cost; environmental risks and price distortions must be included	Based on market price of raw materials and crops; soil is treated as an agricultural input and its value to production is computed alongside other inputs of production
Water storage	Damage cost avoided or infrastructure value (cost of new catchment facility) coupled with relevant risk assessment	Estimated cost from experts or infrastructure value from actual expenditure of related projects
Archeological Preservation	Conservation value or stated preference approaches is most suited.	WTP for preservation serves as the baseline.
Support Structure	Substitution cost or infrastructure cost (converting soil to be suitable to support specific structures)	Conversion value from current soil structure to required soil strength for alternative usage
Biodiversity	Conservation value measured using stated preference approach, coupled with ecological	WTP for soil health / biodiversity is most appropriate. BT values can be applied, with caution.
Erosion Control	Damage cost avoided or damage value with projected added risks from risk assessment	Estimated cost from experts or actual expenditure from related projects; WTP for erosion control could also be used
Flood mitigation	Damage cost avoided or damage value with projected added risks from risk assessment	Estimated cost from experts or actual expenditure from related projects; WTP for flood mitigation could also be used
Carbon Sequestration / carbon storage	Estimate the change in soil carbon storage using direct measurement or indirect means from current status to the alternative soil use.	Gains or loss of SOC will be valued using market price of soil carbon credits
Pollution Mitigation	Cost-based approaches are recommended to measure value change from attenuation capacity of soil or substitution cost for soil's bioremediation	Potential costs from degradation of attenuation capacity of soil; BT values can also be applied, with caution

Non-use value, or value-not-in-use, refers to the value that the general public places on the existence of resources regardless whether they directly use or experience the resources now or in the future (Evans, Banzhaf et al. 2008). It is not dependent on the resource's existing usage but relies on the quality and quantity of goods that are not consumed (O'Garra 2009). Because of its connection to the collective good, it is usually connected with the social aspect of analysis. Non-use values are divided into option value and intrinsic value. Option value arises from keeping alternative usage of the environmental good in other capacities in the future. This component of value is especially important for resources that are not currently being used at its optimum levels. Soil used in agricultural production may be used in other capacities, such as for timber production, for grazing or supporting structures. Soil's option value becomes particularly essential when its ecological functions have been greatly diminished due to degradation that it is sub-optimal for its current use.

Existence value pertains to the amount people value a specific resource solely for the sake of its existence. It encompasses altruistic and bequest values. Altruistic value refers to the worth that individuals allocate for specific resources so that others may be able to enjoy them, while bequest value arises when the concern is towards the enjoyment or use of future generations. Arguably, the existence value is the most understudied and undervalued among the different aspects of economic value. Since it is related to socio-cultural aspects, existence value could be substantial for indigenous peoples whose livelihoods and heritage are heavily tied to the environment (Oleson, Barnes et al. 2015). In the Philippines for example, the Ifugaos consider rice planting as a religious and cultural duty, and conserving the ancient terraces as their social responsibility.

1.4. Thesis Structure

This study is aimed in estimating the economic value of soil and analyze particular nuances that are often overlooked or misunderstood when valuing soil resources. Similar to valuing a house, a painting or a piece of jewellery, the valuation methodology is almost as important (and at times even more important) as the final estimated value. Contradictory value estimates are resolved by examining the valuation report, which serves as the 'black box' used to investigate how the valuation was undertaken. Similarly, this book will serve as a surrogate valuation report to help understand how the estimates of soil value were derived.

1.4.1. Research Questions

While there has been growing literature proposing and developing conceptual frameworks for soil value estimation, these have remained largely hypothetical, with sparse real soil valuation studies other than those that valued soil services being part of a larger ecosystem. It is therefore critical to understand how actual valuation of soil can be implemented, which would entail the use of non-market based approaches and how these approaches relate to soil valuation frameworks. Understanding the dynamics of soil valuation frameworks, value typology and valuation approaches will provide not just an acceptable estimate of soil value, but also answer the following questions:

- Should there be a unified all-encompassing valuation framework to estimate soil value that would be suitable for all context?
- Among those proposed valuation frameworks, which is the most suitable for the study area and which approaches are most recommended?
- Is the role of stakeholders essential in soil valuation? And why?
- Does the addition of spatial environmental variables provide important inputs to the valuation process?

1.4.2. Book Format

This book is divided into eight chapters. **Chapter 1** introduces the concepts in soil valuation and discusses the different frameworks in environmental valuation. It also serves as an overview for the entire thesis. **Chapter 2** provides a comprehensive summary of non-market based valuation techniques that would be used in subsequent chapters in estimating soil value. It discusses some technical matters and current research trends in soil valuation. A description of the study area is also provided at the end of the chapter. **Chapter 3** analyses the economic contributions of soil in agricultural production by using Production Function (PF), a revealed preference approach that estimates soil value based on its apparent contribution to productivity and profit. **Chapter 4** explores the use of contingent valuation (CVM) approach in determining the willingness to pay (WTP) for soil conservation. It examines the various socio-demographic and soil degradation determinants to stated value. **Chapter 5** continues in exploring stakeholders' willingness to pay (WTP) for conservation and examines the use of other contingent valuation formats to limit some of the constraints of the open-ended structure. **Chapter 6** analyses stakeholder WTP heterogeneity for soil's indirect use-value by assessing the various socio-demographic and spatial determinants influencing preference variation with the use of discrete choice experiment (DCE). **Chapter 7** estimates soil value using the replacement cost method, which analyses the financial strain brought about by medium- and long-term degradation. Lastly, **Chapter 8** provides the synthesis of the research, providing a concise summary of the experiences learned from the research and answers to the research questions stated previously.

1.5. Conclusion

The steep rise of human population in the past century has significantly strained the amount and quality of environmental resources. Soil, in particular, has been under threat from degradation due to poor resource management and the lack of understanding of its contribution to human well-being. Many soil products and services generally have no developed markets to gauge their economic worth. Without a pricing mechanism to communicates their utility and scarcity, soil resources have been substantially degraded due to exhaustive usage and gross mismanagement. Decision makers have often chosen short-term profitability over the long-term sustainability of soil use because much of the soil's ancillary economic contributions and indirect benefits have not been well-recognized.

Economic valuation of soil provides an explicit connection between the principles of welfare economics and the need for environmental protection and sustainable resource management. The valuation of the environment has dramatically altered the public discourse on sustainable resource management and shifted the paradigm decoupling economics with the environment.

The current debates in environmental valuation have centered on concepts, valuation coverage, suitability of techniques and the usability of results. It wasn't a question whether frameworks and methods can be developed, but whether these economic instruments will result in credible and usable value estimates. For change to occur, the methods should not only be scientifically grounded for replicability but also be logically justified for acceptability. And given the considerable diversity in economic and environmental thoughts, it would almost be unfathomable to unify all intended applications into a singular classification. Arguably, it will not be feasible nor will it be beneficial to dictate on a single valuation framework. The expansion in valuation usage has necessitated that these frameworks are allowed to mature to provide better elucidation of various aspects of economic value.

The recent entry of soil science in environmental economics has significantly enriched the discourse in soil valuation. It does so by promoting the need for further differentiation of ecological services based on soil quality indicators and highlighting the implicit linkage of soil amenities with the different aspects of soil degradation. While many soil valuation frameworks still have their shortcomings to consider, the progress has definitely been a positive leap forward towards a more comprehensive picture of soil contributions.

Chapter 2. **Approaches to Economic Value of Soil**

Since most environmental amenities do not have their own developed markets to provide price estimates, non-market based techniques have been developed to assess their values. This chapter investigates the various valuation techniques that could be used in soil valuation and provides a review discussing their main strengths, shortcomings and potential applications. A discussion on technical matters and current research trends in the field of soil valuation is also presented in the second half of this chapter. Lastly, a brief description of the research area is reported at the end of this chapter. The different methods and value types that are discussed in the first two chapters will be employed in the study site, which will be discussed in the succeeding chapters.

2.1. Valuation Techniques

Numerous valuation methods have been advanced to estimate the economic value of soil and other environmental goods. The primary topological system used to categorize these methods rely on how values are exposed, whether through explicit consumer choices (stated preference), implicit consumer behavior (revealed preference), or through the costs associated with the resource's use or degradation (cost-based).

2.1.1. Stated Preference Approach

Stated preference (SP) approach has been the most dominant technique in monetary valuation of the environment (Bateman and Mawby 2004). In this approach, people are directly asked to indicate their stated preference for a given scenario. Depending on the study, the respondents are asked either for their willingness to pay for the use or access of a particular environmental service or for their value they would be willing to accept for the loss of access for a particular amenity. WTP is usually asked from respondents benefitting from the use of the amenity, while WTA is solicited from those to be adversely affected by a scenario change (Edwards-Jones, Davies et al. 2000). Two farmers with similar economic profile and land holdings will commonly exhibit comparable WTP values but may have entirely different WTA given slight changes in personal background and character behavior. Although the higher variability of WTA makes it harder to accept as a standard measure of value, both WTP and WTA play essential roles in estimating the real worth of an environmental good in the valuation process. It is essential to identify which of these values one is measuring, which should then be related to the objectives and with the analysis of the report. The method of directly soliciting stakeholders' views is considered to be the strongest and the weakest argument for stated preference. It is said to be strong because it directly involves relevant stakeholders in estimating the value of a particular good or service that incorporates taste, perception and demographics; but is also considered weak because stakeholder responses can be extremely unstable and erratic that can result to high degrees of variation, fluctuations, and uncertainties. It gets around multi-collinearity and double counting issues, and also mitigates the effect of omitted variables (Guignet 2012). Unlike most other valuation technique, SP can be used to measure the non-use values and cultural ES (Ruijgrok 2006, Baez and Herrero 2012).

For soil amenities, stated preference has been used a number of times in analyzing the benefits of soil conservation and in estimating a variety of indirect use values. To enhance the reliability of SP estimates, actual costs of alternative improvements are calculated and are reflected in the questionnaire. The respondents should find the payment vehicles to be credible, comprehensible and realistic; otherwise, the results would be speculative and would lack credibility (Evans, Banzhaf et al. 2008). Pre-testing and statistical analysis of probable price determinants are recommended to be undertaken as supplemental safety measures.

Although SP remains as the most widely used approach, there have been serious doubts on its usability, given intrinsic methodological limitations. Critics have argued that SP studies are highly vulnerable to inconsistencies of human perception, resulting in irregular and unpredictable responses (Arrow, Solow et al. 1993, Carson 2000). A range of psychological triggers can be activated inadvertently during the interview process that can influence the respondents and skew the results (Moore 2002, Cai, Cameron et al. 2011). Minimizing the hypothetical bias from WTP responses is the primary consideration in structuring the experimental design. The credibility of the design depends very much on the plausibility of the given scenarios and whether the results have been formed consistently (Flores and Strong 2007). Proper population stratification is essential in informing respondent selection, which can lead to more efficient statistical estimation and minimize unintended inconsistencies. A rigorous focus group discussion and well-crafted debriefing questions usually accompany reliable stated preference studies (Evans, Banzhaf et al. 2008). There are two primary techniques using the stated preference approach: the **contingent valuation method**, and the **discrete choice experiments**. Both these SP techniques are applicable in estimating WTP or WTA, which can then be used as a proxy to the stated value.

Contingent Valuation Method

Contingent valuation method (CVM) has been the dominant valuation technique for environmental goods, using the philosophy of direct participation in decision-making. Typical CVM setting would have the respondents informed of hypothetical settings presented with specific information on the nature and extent of damages and the cost needed to support such environmental program (Arrow, Solow et al. 1993). The respondents' WTP would then be solicited through: an open-ended question format, a multiple choice format with specified price bids; or a referendum format to reject or accept the proposal.

From its initial conception, CVM has undergone improvements providing stronger theoretical foundation and statistical efficiency (see Adamowicz, Boxall et al. 1998, Carson 2000, Bateman, Carson et al. 2002, Cuccia 2003). However, various empirical and methodological issues still remain. In CVM surveys, a proportion of respondents would indicate a refusal to pay any amount for the use of a public good due to some mitigating circumstance or procedural dissension. Protest bids are often associated with the free-rider syndrome, which can unduly skew WTP averages (Green, Jacowitz et al. 1998). Protest bids can also come from those inherently against additional taxation, those who naturally distrust the government on principle, or from those unaware of environmental benefits. While protest responses are routinely removed from the samples and are assumed to be non-indicative of the true values, this method of censoring could also be problematic (Jorgensen, Syme et al. 1999). Some argue that zero value responses should not be suppressed entirely and that rules on how to contextualize and adjudicate the censoring procedure are critical. While a number of studies have tried to understand protest bids better and, there is still no collective agreement on how to deal with them in CV analyses. Another crucial issue with CVM is its limitations to value marginal changes in quality, quantity or attribute of environmental goods (Kahneman and Knetsch 1992). This arises when respondents evaluate economic value independent of the characteristics of the actual good. Some consider CV to include passive value not motivated by the utility but by altruism or "warm glow" (Carson, Flores et al. 2001). This not only limits the comparability of WTP values but also creates skepticism on the usability of CVM in a variety of applications. Nevertheless, while other methodological and empirical concerns exist, CVM has remained the preferred technique applied to many environmental and resource valuation studies.

Discrete Choice Experiment

The other primary SP technique is the discrete choice experiment (DCE) method, otherwise referred to as choice modeling or attribute-based surveys. This method is founded on Lancaster's model of consumer choice and McFadden's random utility theory (Lancaster 1966, McFadden 1974). A list of attributes describing two to four scenarios is given to respondents, who are then asked which of the scenarios they prefer. The standard technique presupposes the respondents' utility to be defined over a defined array of attributes including cost (Colombo, Christie et al. 2013). The individual's indirect utility equation can be estimated using statistical analysis (e.g., multinomial logit models, random parameter logit models). And with the inclusion of cost attribute, the marginal utility can be converted into estimates of the willingness to pay (or accept) attributed to the change in an environmental attribute.

Unlike other non-market valuation approaches, DCE provides the estimation of value change in some attributes, as well as the compensating surplus measures of multiple changes in attribute levels (Viteri Mejía and Brandt 2015). The expected changes from the perspective of natural science are transmitted through which the respondents would be able to relate directly. DCE allows the analysis of the effects of attributes in a resource's value, of the endowment and choice on the functional form of the welfare measure. Considered as a more sophisticated SP, the choice experiment analysis has been used to quantify the marginal value of environmental goods and services including biodiversity enhancement (Bartczak and Meyerhoff 2013, Zander, Signorello et al. 2013, Yao, Scarpa et al. 2014), health-risk aversion (Veronesi, Chawla et al. 2014, Vidogbena, Adegbidi et al. 2015), climate change adaptation (Nguyen, Robinson et al. 2013, Andreopoulos, Damigos et al. 2015), and soil conservation measures (Colombo, Calatrava-Requena et al. 2006).

2.1.2. **Revealed Preference**

Revealed preference (RP) techniques use the indirect approach to valuation, employing a complementary and substitutive relationship between non-marketed goods and those priced from market transactions (Ferreira and Moro 2010). Unlike explicit estimates from SP, the values are implicit computed and assigned by the researcher through the use of utility functions and rationalized from observations of consumer choices and budget constraints (Beshears, Choi et al. 2008). The RP approach quantifies the use value, the functional aspect of the total benefits and exposed by variations in compensated demands for comparable private goods (Eom and Larson 2006). The demand is inferred from the demand function of related private goods which is aimed is to minimize misrepresentation of consumers when stating their preferences (Bradford and Hildebrandt 1977). Economists describe these revealed preferences to be normative preferences or those that represent the actor’s real interests (Beshears, Choi et al. 2008). Revealed values may provide more accurate results because the estimates are based on consumer behavior, making use of actual purchases of marketed goods as substitutes (Eshet, Ayalon et al. 2006). Errors and biases associated with surveys and interviews are substantially minimized.

Although revealed preference may be considered a reflection of human behavior, several factors may influence choices and actions to deviate from the individual’s normative preference. Real preferences of consumers are not always revealed through their choices due to decision-making errors (Beshears, Choi et al. 2008). Passivity, complexity, limited personal experience, third-party marketing, and the inter-temporal choice may cause individuals to act contrary or atypical to their real economic interests. The theory of revealed preference belongs to the general standard model of consumption or demand theory. The results of revealed preference studies can be compromised by several factors such as analytic errors, myopic impulses, inattention, passivity and misinformation (Beshears, Choi et al. 2008). Researchers usually make use of untested assumptions regarding the public’s awareness and perception (Guignet 2012). The use of self-reported preferences has been developed to fill the gaps in insufficient market data. The use of self-reporting may beget unintended noises usually associated with SP; some researchers have found the use of self-reports to be useful in revealing behavioral aspects (Beshears, Choi et al. 2008). Self- reports can be used in forecasting future behavior, and in revealing consumer confidence on the optimality of his behavioral choices.

The major drawback, however, in using revealed preference is that the values derived are implicit prices for the attributes, which may be reflective of private gains but not necessarily of their social benefits. The quantity of data of the different variables necessary in the analysis, which are needed to be used in statistical analysis, is enormous. Much of these data are not usually available from statistical records or are not measured on a standardized scale. There are numerous techniques which utilize the principles of revealed preferences. Three of the most commonly used RP techniques are **market pricing**, **production function**, and **hedonic pricing**.

Market Pricing

Market pricing (MP), also known as market analysis, is a valuation technique which estimates the resource's value, based on the comparable market price of the products and services. It combines the observed utilization of goods with the concepts of consumer theory and incorporates econometric methods to derive the demand curves (Turner, Georgiou et al. 2008). It measures value from amenities derived from goods and services, based on the quantity of demand and supply at varying prices. The total economic change estimate is computed by combining the change in consumer surplus and the change in producer surplus. The change in consumer surplus is the difference between the demand functions before and after the change in provision, which is estimated using real and projected market demand function and consumer surplus of the environmental good. Similarly, the change in producer surplus is the difference between the estimated supply function before and after the change in provision. In the case of soil resources, the estimated change in value would be based on the economic impact resulting from soil degradation or conservation. Depending on the good and its specifications, the operational associations of the supply, demand, and determinants could be anything from explicitly simple or overwhelmingly complicated.

The method is theoretically sound since it estimates value based on actual market transactions and relationships, but it is only applicable to cases where there are available market data. In the case of soil value, market pricing is almost exclusively used for provisioning ES (e.g., fertility) and is typically limited to small-scale valuation. Agronomic data are available but are often limited to mesoscale and macroscale levels. Prices of soil products are affected by a variety of seasonal variations and other factors, which must be considered in the assessment. Since it assesses only the direct and indirect values-in-use, the estimate usually forms the lower bound of the good's total economic value.

Production Function Method

The production function (PF) method, or net factor income, is mainly used in valuing the indirect use-values of environmental goods, estimating value based on the good's contribution to production (Birol, Koundouri et al. 2008). The environmental good is treated as a factor of production, alongside labor, land, and capital (Edwards-Jones, Davies et al. 2000). The general approach consists of determining the physical impact affecting the resource or an associated ecological function and valuing the impact of these environmental changes based on the effects on their marketed outputs (Barbier 2000). The changes in the quality or quantity of an environmental good alter production cost, output, prices, and the returns from other factor inputs. The changes to production provide the estimates of environmental good's economic value. This straightforward approach makes the method relatively easy to understand, which may partly be the reason for the method's widespread use (Edwards-Jones, Davies et al. 2000).

However, assessing how the changes in environmental inputs affect the actual response in production is quite challenging given the complexities of the market structure and the nature of production. Since the value would be dependent on the apportionment of the prices of marketed commodities, the critical assumption is that all the other inputs of production must also be assessed (Turner, Georgiou et al. 2008). These prices from other production inputs must be reviewed carefully so to reveal externalities that distort the exchange value, such as subsidies, protective tariffs, and taxation. The effects of soil degradation on the downstream may not also be reflective on the estimates from PF, resulting in an undervaluation of total value. Also, when the effect of the environmental service is small compared to the other inputs in production, assessing the value is almost impossible.

Hedonic Pricing

The fundamental principle in hedonic pricing is that the price of a marketed good is related to its utilities and its attributes (Rosen 1974). Value is estimated by how much people are prepared to pay for complementary products that will allow them to consume particular goods or services (Nerlove 1995). Individual preferences on quality may be inferred from the differential prices of purchased goods, which are then used to reveal the implicit value of environmental characteristics through econometric techniques. The HP framework would first require the approximation of the hedonic function by relating the price with essential attributes. It involves data collection and sampling, model estimation and estimation of the welfare measures (Riera, Signorello et al. 2012). The demand function for each characteristic is then estimated through regression of the hedonic function against physical and socio-economic variables (Nerlove 1995). The monetized benefits of an environmental good are established through regression of the prices of other marketed goods against their attributes. The inclusion of spatial attributes has become a staple in more recent HP studies, together with climatic, environmental and urban amenities (Moro, Brereton et al. 2008). HP models have been commonly applied to determining housing prices, and to assess the marginal benefits arising from proximity to specific locations (Tapsuwan and Polyakov 2016). Previous environmental applications of HP include estimating externalities from ecological degradation, influence of conservation, aesthetic benefits from the natural environment, and impact of environmental risks.

While the use of HP has been growing in the past few decades, its application for certain environmental goods particularly in soil resources have been limited. A necessary assumption in HP is the existence of a freely functioning and efficient market, where buyers are well-informed of the products and their purchases. But in reality, the market is non-homogenous which can be segmented into submarkets, and that consumers may not be fully aware of environmental attributes and relevant amenities (Xiao 2017). People are primarily affected by their own environmental perception, which can be highly qualitative. Perception-related ideas of acceptability, tolerance, and satisfaction, may be considered to be highly subjective per individual and influenced by space, location, and style. Also, HP models require large amounts of data to gather, implement and interpret, which would include property values, environmental quality attributes and property characteristics.

2.1.3. Cost-Based Approaches

Cost-based approaches consist of methods that estimate value based on the cost associated with avoiding damages from lost services, incurred cost due to damages or loss of ES, or the cost of substitute services for degraded ES (Kumar 2012). They are considered as less informative as benefit-based measures, as they contextualize value based on potential losses from changes in soil physical and chemical characteristics, quality, or use. They are highly relevant in measuring the extent of value change caused by policy intervention or soil-use change. The three main valuation techniques under cost-based approaches, are **damage-cost avoided**, **replacement cost method**, and **substitution cost method**.

Damage Cost Avoided

The Damage-Cost Avoided (DCA) estimates based on the cost to society to prevent the loss or reduction of supply or quality of environmental goods. Also referred to as defensive expenditure method and preventive cost approach, this method is based on the household production function theory of consumer behavior (Birol, Koundouri et al. 2008). DCA measures the lower bounds of value estimates since it is assumed that individuals would choose economical ways to secure environmental goods or services (Swinton, Lupi et al.). In DCA, it is not the actual cost of damages that are evaluated, but the cost of damages prevented subtracted by the cost of intervention measures.

Empirical aversion to risks varies broadly on the context (Nestle 2008). From the soil resource perspective, stakeholders may counter the risks of soil erosion by adopting aversive expenditures or change their land-use practices to minimize the rate of soil degradation. For example, farmers may adopt more sustainable farming practices (e.g., no-till farming), or implement conservation technologies that will reduce exposure of soil from erosion agents (e.g., mulching).

Replacement Cost

Replacement cost method (RCM) estimates the value of environmental damage according to the price that would be needed to restore the environment from its previous undamaged state. In soil valuation, replacement cost method may be used to estimate to value-based on the degradation rates occurring from its current use. The erosion and degradation of farmlands affect not only agricultural production upstream, but may also degrade reservoirs, contaminate water supplies, cause sedimentation in dams, or disrupt ecosystems downstream. The costs of rehabilitating the upstream farmlands, restoring the downstream ecosystem, dredging sediment-filled reservoirs and decontaminating polluted water supplies would be tallied, which would then be used as an estimate.

A modified RCM is using shadow projects which provide an equal alternative to the environmental good or service that would be lost due to degradation. The different costs of the shadow project would then be calculated and then used as the estimate for the value of the environmental good. RCM has often been criticized whether the estimated value is reflective of the real cost of damage. Some argue that once the environment has been damaged, it would be unlikely that any amount would be able to restore it from its pristine state. Others fear that by using the replacement cost method, the assessment would only be reflective of the short- and medium-term consequences of environmental degradation while sacrificing the long-term impacts.

Substitution Cost

The substitution cost method (SCM) makes use of the cost of supplying the substitutes for the environmental good or services being estimated. It is closely related to the RCM, but instead of finding the cost of rehabilitation of the environmental good or service, the cost of accessing a viable alternative is used as a proxy value. Some doubt whether the substitute good or service would deliver the same level or quality of benefits as the natural resource.

2.1.4. Benefit Transfer Analysis

Benefit transfer (BT) is technically not a valuation approach, but rather a meta-analysis that can calculate economic value by using previous valuation studies as comparable estimates. When direct valuation is too expensive or time-consuming, this alternative approach takes preexisting values from related case studies to develop a customized benefit estimate (Kaul, Boyle et al. 2013). A number of online catalogs of valuation studies and reports have been generated expressly developed for BT use. Some examples of databases relevant to soil resources are listed in **Table 2-1**.

While BT analysis has created its niche in the realm of environmental valuation, the approach remains to have some theoretical and methodological weaknesses. Because it is still relatively new, there is still no widely used standards or guidelines for the use of BT. Best practice for BT requires that the nature of the services and the characteristics of the local population be substantially similar (Huguenin, Leggett et al. 2006). When previous studies used in BT are not sufficiently comparable, then the accuracy of results becomes highly questionable.

2.2. Spatial Models and ES Value Mapping

In recent decades, the development of geospatial technologies such as geographic information systems (GIS) and remote sensing (RS) have provided valuable advantages in environmental assessment and monitoring (Anselin 2001, Cristofori, Facello et al. 2017). Their capability of incorporating spatial data from a variety of sources with different formats and structures has been a powerful tool in spatial analysis (Payn, Hill et al. 1999). For environmental valuation, the use of spatial data is crucial in analyzing ecosystem services, environmental risks and degradation, and spatial determinants to value. Environmental attributes and economic value commonly exhibit spatial dependency (Bateman, Day et al. 2006). Previous studies have demonstrated the advantage of integrating spatial data and physical models in econometric studies.

Table 2-1. Existing databases of valuation studies relevant to soil resources

Database Name	Description
Environmental Valuation Reference Inventory (EVRI) https://www.evri.ca/	Provides compendium of summaries of environmental and health valuation studies which can filter by publication date, document type, environmental asset, economic measure and value type
Review of Externality Data (RED) www.isis-it.net/red	An extensive review of literature and documentation of external cost analysis, mainly from Europe; database filters include the type of environment, sector, valuation type, receptors and environmental burdens
Conservation Gateway https://www.conservationgateway.org	A wide array of conservation topics including planning, practices and filtered by region; Valuation studies are found under Library > Ecosystem Services
The Economics of Ecosystems and Biodiversity (TEEB) – Valuation Database http://www.teebweb.org/publication/tthe-economics-of-ecosystems-and-biodiversity-valuation-database-manual/	Collection of more than 1300 valuation studies, the studies are subcategorized by biome, ecosystem, service types, country income groups, valuation technique and value type

In recent years, spatial variability in valuation has been elaborated more explicitly with developments in methodological frameworks (Willemen, Verburg et al. 2008). Proximal and distal analyses have been used to interpret preference heterogeneity and to contextualize value determinants (Borchers and Duke 2012, Choi 2013, Kousky and Walls 2014). For examples, proximity to environmental amenities has been shown to increase property values in specific instances by enhancing aesthetic value, providing recreational opportunities, and improving ecological health (Bowman, Tyndall et al. 2012, Abildtrup, Garcia et al. 2013, Tapsuwan, Polyakov et al. 2015, Nicholls and Crompton 2017). But other studies have shown the opposite, identifying environmental disservices from contiguous natural environment that negatively impact the housing market. Spatial information related to land use changes, population migration patterns, and climate change scenarios have been used to examine changes in amenities and economic value. They have also been shown to be very useful in conducting BT analysis, evaluating the relationships between variables, and analyzing hypothetical scenarios and predictive models.

In the succeeding chapters, the incorporation of geospatial data in soil valuation will be further investigated. Geographic attributes will be used to explain stakeholder preference heterogeneity and investigate the spatial determinants affecting economic value. Soil degradation models will be generated to contextualize how environmental risks can affect stakeholder consciousness and choice preferences, which can consequently influence the valuation process. And with the evolution of new spatial models with greater flexibility, the expanding roles and research potentials of spatial data and geo-information in environmental economics will also be further discussed.

2.3. Soil Quality Indicators

The need to better analyze marginal benefits from provisional changes in soil services has intensified interest to include soil attributes in the valuation process. Early frameworks assessing soil economic value did not explore the direct relation of soil properties with ecosystem services (Adhikari and Hartemink 2016). Soil had been considered merely as a constituent of larger ecosystems which homogenized natural soil features and suppressed some important geomorphological and pedologic properties and processes. Not incorporating soil attributes led to weak methodological flexibility in assessing land policies and soil use alternatives. Recently, soil has been characterized in some valuation frameworks as natural capital, expressed through its various biological, chemical and physical properties (see Dominati, Patterson et al. 2010, Robinson, Hockley et al. 2012). This addition of soil quality indicators presents new perspective that can further highlight sustainability in land use and management practices in agroecosystems (Shukla, Lal et al. 2006). These provide baseline measurements needed to contextualize the assessment of pedologic services such as carbon reserves and nutrient supply (Robinson, Hockley et al. 2013). The primary constraint limiting the use of soil quality parameters in valuation has been the logistical requirements to obtain field measurements. Traditional in-situ surveys to acquire soil quality data are oftentimes laborious, exhaustive and time-consuming. With the already extensive economic and environmental data required the valuation, adding soil quality surveys could easily turn the valuation process into a herculean undertaking. Thus, advances in soil surveying techniques (e.g., remote sensing applications) must be explored on how they could supplement the theoretical and practical aspects of soil valuation.

Existing soil resource audits would need to be updated to include the requirements of soil valuation. Since assessment of soil value is undertaken at varying scales, soil data repositories would need to modernize at the local, regional and national scales. Upgrading the soil data inventories would be crucial in conducting a comprehensive valuation of soil resources and in assessing land use management and policy alternatives.

2.4. Soil Degradation Processes

An important aspect of assessing soil’s economic value, particularly when viewed as a natural capital with heterogeneous attributes, is understanding the processes that diminish soil quality (Dominati, Patterson et al. 2010). Soil degradation, defined as the detrimental changes in soil attributes or the removal of soil altogether, is intimately linked with various ecological functions and soil amenities (Yakovlev, Molchanov et al. 2015). The decline in the soil's capacity to support ecological functions affects the provision of essential soil amenities, which in turn lead to diminished economic value (Dominati, Patterson et al. 2010, Lal 2012).

Soil degradation results from the long-term interactions of various factors (i.e., anthropogenic, biotic, abiotic) affecting soil characteristics (Yakovlev, Molchanov et al. 2015). The susceptibility for each type of degradation is contingent mainly on the synergy among the biophysical and human structures, and the processes that are functioning across the different scales (Orchard, Stringer et al. 2017). Similar to the valuation of other environmental goods, the assessment of soil degradation is based mainly on how it impacts services and amenities. Determining whether the changes in the soil quality would be considered as degradation depends on the system, which suggests that degradation cannot be assessed without considering the spatiotemporal, socio-economic, and environmental contexts (Orchard, Stringer et al. 2017).

Some of the most common types of soil degradation include erosion, salinization, acidification, compaction, loss of organic matter, contamination, biodiversity reduction and structure decline (see Lal 2012, Stoessel, Sonderegger et al. 2018). Similar to the categories of soil properties, the principal types of degradation are physical, chemical, and biological. **Table 2-2** lists some of these primary threats to soil quality and how they impact ecosystem services.

The role of soil degradation processes in the context of economic valuation will be further investigated in the succeeding chapters. Water erosion, in particular, will be a central issue in analyzing stakeholder cognition and behavior, and how it impacts the estimated implicit and explicit values.

2.5. **Study Area**

To implement different soil valuation approaches and test some hypotheses, the agricultural town of Norzagaray, Philippines was chosen as the study site. It was selected from a small group of potential candidate sites based on four criteria: (a) the willingness of stakeholders to participate, (b) support from local officials, (c) the accessibility and security of the site, and (d) the availability of required data inputs.

Norzagaray (14°55°N 121°3'E) is located on the southeastern side of the Province of Bulacan in the island of Luzon, as shown in **Figure 2-1**. Norzagaray's eastern half is covered by steeply sloped forestlands, which is part of the Angat Watershed, a critically important reservation that provides water and electricity for the country's capital region. The whole Angat Watershed is 62,309 hectares located at the southern tail of the Sierra Madre Mountain Range. Due to its critical nature, the Angat Watershed has been declared as a protected forest reserve for watershed purposes and cannot be subject to sale nor settlement. The western side is characterized by gently to rolling terrain, where much of the economic activities take place.

In terms of population, Norzagaray has more than tripled its residents in the last 25 years. The rise in population is due primarily to increased urbanization in Poblacion and the expansion of cement manufacturing and quarry operations. Trade and Agriculture led all industries in total employment comprising 28% and 23% of the total workforce, while mining and quarrying lead all industries based on tax revenue. In 2013, the total household population stood at 22,401 (105,470 inhabitants), with 59% employed in the agricultural sector. Norzagaray has an estimated 9250 hectares of agricultural lands representing 28% of the town’s total land area. Roughly 44% of the agricultural land is used for production, planted mainly with rice, mango, banana, corn, vegetables, and root crops. The rest consists of idle lands covered mainly with grass.

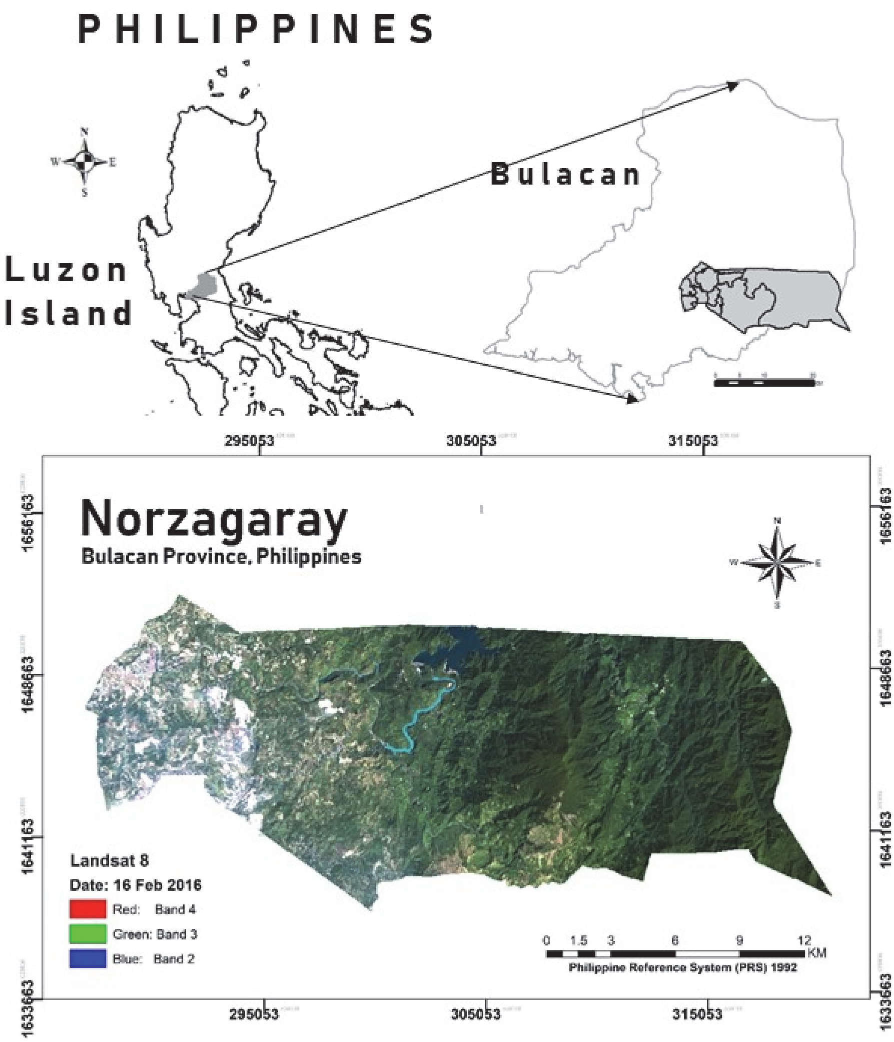

Figure 2-1. Map of the Study Area – Norzagaray, Bulacan

The main soil types found in the study area are Buenavista silt loam (0.43%), Novaliches loam (10.61%), Novaliches clay loam (40.45%), Presna clay loam (0.90%), Sibul clay (6.42%) and Novaliches soil undifferentiated (41.19%). The underlying geological formations are abundant in shale and limestone, which are used in the cement manufacturing industry. Previous assessment studies have revealed the vulnerability of the area from a variety of natural hazards. The mountainous regions have high landslide risks in the form of differential settlement with lateral downslope movement and minor slump. Deforestation due to illegal logging operations have intensified land instability, and have caused massive landslides in the past. Erosion and its associated sediment yield pose significant concerns due to its potential economic impact on power generation and water supply.

In the late 1970s to the early 2000s, population increase coupled with the intensification of agricultural activities was blamed for the high rates of erosion. In response, stricter zoning policies and dedicated reforestation efforts were implemented to counter soil loss. In recent years, the expansion of industries, such as quarrying and mining, combined with the stagnation in agricultural profit, has led to the dormancy of farmlands. The abandonment of agricultural lands has exacerbated soil degradation and landslide susceptibility problems, which have already been challenging given Norzagaray's topography, climate and soil characteristics. Increased river siltation and sedimentation in the main reservoir are significant concerns in the management and maintenance of the Angat dam. It minimizes the dam's storage capacity, which adversely affects the dam's power generation, flood mitigation, and water supply.

These physical and anthropogenic characteristics of Norzagaray will be among the primary considerations in implementing and analyzing the economic value of soil. The diverse assortment in soil quality, land usage, degradation vulnerability, and soil ecosystem services found in the study area provides a suitable platform to test the different hypotheses, which will be further discussed in the succeeding chapters.

Chapter 3. **Understanding Use Value from Production Function Method**

This chapter integrates econometric modeling with spatial statistics to quantitatively estimate soil conservation value in agricultural production. It provides blueprints for the use of production function in soil valuation, which is a type of revealed preference approach well suited to quantify the contributory value of soil in production. Production Function (PF) is an economic tool that generally associates physical products with various inputs or factors of production, which has been a key concept in neoclassical theories in economics. This chapter aims to provide some clarity on the complex relations between agricultural yield, environmental degradation, soil conservation, and stakeholder perception. This information can provide a critical step needed in developing strategies that can promote a framework of sustainable participation in conservation measures that bring together public interests with private needs.

3.1. Introduction

Farm management's primary goals include efficiently utilizing land and capital resources, increasing agricultural production and ensuring long-term profitability (Guerra and Pinto-Correia 2016). Farmers have to balance the short-term issues of profitability and efficiency with medium and long-term goals of sustainability and environmental protection. Limited capital, poor soil quality, technological inexperience, land tenure issues, and high levels of environmental uncertainties are some of the major constraints a farmer encounters in farm operations (Carter and Barrett 2006, Barrett and Bevis 2015). Aside from ensuring the long-term economic viability of farmlands, effective farm management is essential in mitigating environmental impact caused by intensified agricultural production (Felix-Henningsen, Morgan et al. 1997). Previous studies have explicitly linked exhaustive agricultural activities with increased soil erosion rates (Pimentel 2006, Guerra and Pinto-Correia 2016). The pressure to increase agricultural yield coupled with ineffective land management has brought enormous stress to the environment and constraints to various ecosystem services (Mutoko, Hein et al. 2014). Short term maximization of agricultural production has led to exploitative farm practices and unregulated land conversion costing long-lasting damage to soil productivity and economic efficiency (Nyssen, Poesen et al. 2009). Soil degradation threatens not only the farmers' livelihood and regional food security, but its *off-site* effects can result in more disruptive complications (Foley, Defries et al. 2005). A variety of indirect soil services including water regulation and filtration, carbon sequestration, and pollution control, could be jeopardized by increased rates of soil degradation (Morgan 2005, Lal 2014). Sedimentation reduces the storage capacity of reservoirs downstream resulting in costly disruptions to power generation, irrigation and flood control. Previous studies on soil degradation analyzing the annual cost of soil degradation in different countries have shown staggering costs due to soil loss and the resulting sedimentation (Moller and Ranke 2006, Hein 2007, Telles, Dechen et al. 2013).

There has been general agreement that prevention of soil degradation is far more cost-effective than the rehabilitation of degraded land (Naidoo and Iwamura 2007, Park and Sawyer 2016, Wang, Yang et al. 2016). Soil conservation methods have become integral in land and farm management for both economic and environmental reasons. Promoting greater community involvement and understanding the factors influencing the decision-making process are crucial to the success of conservation schemes (Assefa and Hans-Rudolf 2016). Policy and market interventions are oftentimes necessary given the pervasive and interwoven economic disruptions resulting from soil erosion. For policy-makers, identifying conservation priorities and developing tactical schemes are complex undertakings, given contested stakeholder values, and uncertainties on the costs and benefits brought by conservation measures

(Pert, Lieske et al. 2013). Whereas policy interventions such as economic incentives and taxes can be used to advocate for environmentalism and sustainable farming practices, much of the decision-making is left to the farmer. It is important to understand and contextualize soil conservation management from the farmer's perspective, given that much of its success depends largely on stakeholder participation (Nackoney, Rybock et al. 2013).

In this chapter, the primary inputs of agricultural production and soil-related factors affecting agricultural operations are evaluated on how they impact productivity and profitability. As in many intensively cultivated regions in developing countries, soil degradation is considered one of the most serious economic and social threats to the sustainability of agricultural systems in the Philippines (Olabisi 2012). Deforestation, unrestrained land-use changes, poor farm-management practices have aggravated the problem of soil degradation. Past studies have noted the importance of analyzing the motivations behind the farmer's decision-making process, and have highlighted their critical role in the success of implementing soil management (Cramb, Garcia et al. 1999, Lapar and Pandey 1999). Since the farmer is an economic agent and active stakeholder in soil management, understanding his thought process may provide useful insights on the factors affecting his decisions, specifically in his or her expenditure choices, investments on soil conservation, and perception on soil risks. An additional parameter, environmental awareness, which is aimed at representing

The economic value of soil is assumed to be the value of its role in agricultural production. But like other environmental goods, the 'real' contribution (and value) of soil is immeasurable given its indispensable function in agriculture. Agricultural production will be modelled using traditional inputs of production, which were gathered through rigorous household expenditure surveys. To determine the effects of stakeholder heterogeneity and soil degradation to productivity and profitability, two additional models would be analyzed. This would reveal which parameters would have significant effect on production and on profit, while correlation analyses would reveal significant associations with conservation value. To overcome this issue of soil value immeasurability, the farmer's expenditure on soil conservation measures is used instead of directly valuing the contribution of soil in production. This argumentation is similarly used in stated preference approaches of using the willingness-to-pay for conservation as a proxy for soil value; but in the production function, actual farm expenditure is used for implicit value approximation. The elasticities for each of the production inputs including the conservation value would also be calculated to determine significant effects on crop yield.

3.2. Econometric Model

Production function (PF) is a revealed preference approach used mainly in valuing the indirect use-value of the environment based on its contributions to production (Birol, Koundouri et al. 2008). It is a straightforward approach of estimating the economic value that treats the environmental good as a factor of production, alongside traditional elements of labor, land, and capital (Edwards-Jones, Davies et al. 2000). In microeconomic theory, PF provides the estimated maximum output that can be produced from the set of inputs using the existing technology available (Battese 1992). The basic PF equation accounts only for the material variable inputs to production and is given by:

$$\boldsymbol{Q} = \boldsymbol{f}(\boldsymbol{L}, \boldsymbol{Z}) \qquad \textbf{Eq. 3-1}$$

where Q is the total agricultural yield produced, Z stands for agricultural material inputs; and L is agricultural labor. To understand the relationship of the material inputs and labor with agricultural production, one of the earliest estimation equation used was the Cobb-Douglas production function, originally designed to estimate the comparative productivity of capital investments against labor costs. The original (sometimes referred to as true Cobb-Douglas) function, which contains only two input variables and assumes homogeneous of degree 1 with respect to the input bundle, is given by the equation:

$$\boldsymbol{Q} = \boldsymbol{A}\, \boldsymbol{L}^{\boldsymbol{\alpha}}\, \boldsymbol{Z}^{\boldsymbol{1-\alpha}} \qquad \textbf{Eq. 3.2}$$

where A is the parameter that represents the level of societal technology upon which the parameters of the function were to be estimated; and α is a constant between zero and one, used to measure the elasticities of labor and capital. The function presented marginal returns to either capital or labor, when the other is treated as fixed input so that the law of variable proportions hold. Using Log transformation on both sides, the equation reads:

$$\boldsymbol{log\, Q} = \boldsymbol{log\, A} + \boldsymbol{\alpha\, log\, L} + (\boldsymbol{1} - \boldsymbol{\alpha})\, \boldsymbol{log\, Z} \qquad \textbf{Eq. 3.3}$$

Newer generalization of the Cobb-Douglas production function allowed the input parameters to have a total sum greater than 1 (allowing for returns to scale other than 1) such that:

$$\boldsymbol{Q} = \boldsymbol{A}\, \boldsymbol{L}^{\boldsymbol{\beta_1}}\, \boldsymbol{Z}^{\boldsymbol{\beta_2}} \qquad \textbf{Eq. 3.4}$$

This equation, like the true Cobb-Douglas, is readily log-transformable, and the parameters could be estimated using least squares regression. The input variable Z can be further disaggregated into conventional farm inputs of seedlings (S), fertilizer (F), and pesticide (P). If the general Cobb-Douglas function was then expanded to include these three input parameters, the equation would be:

$$\boldsymbol{Q = A \cdot L^{\beta_1} \cdot S^{\beta_2} \cdot F^{\beta_3} \cdot P^{\beta_4}}$$ **Eq. 3.5**

A modified translog function can then be used to estimate the values of the constants which would transform the previous equation into:

$$\boldsymbol{ln\,Q = \alpha_0 + \left(\sum_i \beta_i \cdot ln(X_i) + \frac{1}{2}\sum_i \sum_j \beta_{ij} \cdot ln(X_i) \cdot ln(X_j)\right)}$$ **Eq. 3.6**

For this model, regularity conditions of translog production would require parameter constraints, such that $\sum_i \beta_i = 1$, $\sum_i \beta_{ij} = 0$, and $\beta_{ij} = \beta_{ji}$. The basic econometric model can then be expanded to incorporate socio-demographic attributes (I), and the farmer's environmental consciousness rating (E_C). The socio-demographic attributes (I) would include: farmer's educational attainment, farming experience, type of land ownership, type of farm ecosystem[1], farm size, topography, and whether the farmer receives some level of government assistance. In estimating the environmental consciousness (E_C) three variables were used namely: farmer's expenditure on soil conservation measures (E_X), the farm's erosion vulnerability rating (E_V), and the farmer's environmental awareness score (E_A). The additional variables would expand the production function as:

$$\boldsymbol{Q = f(Z, L, I, E_C)}$$ **Eq. 3.7**

A number of functional forms can be used, but since the parameter of the actual technology is unknown, the choice of the appropriate function form becomes essentially an empirical issue. Similar to the modified translog function in Eq. 6., incorporating the additional parameters would transform the equation into:

$$\boldsymbol{ln\,Q = \alpha_0 + \left(\sum_i \beta_i \cdot ln(X_i) + \frac{1}{2}\sum_i \sum_j \beta_{ij} \cdot ln(X_i) \cdot ln(X_j)\right) + \left(\sum_k \gamma_k I_k\right) + \left(\sum_l \delta_l E_{C_l}\right) + u}$$ **Eq. 3.8**

Using the calculated constants generated from the econometric model, the output elasticity of each specific input can be calculated. In economics, output elasticity refers to the percentage change of output in response to a change in the level of input. The general formula for calculating average output elasticities is:

[1] *Farm ecosystem denotes where the farmland mainly gets its water needs. In this study, the farms were grouped either as irrigated farms, or non-irrigated farms.*

$$\varepsilon_{QX_n} = \frac{\partial \ ln\, Q}{\partial \ ln\, X_n} = \beta_n + \sum_n \beta_{1n} \cdot ln\, x_n \qquad \textbf{Eq. 3.9}$$

3.2.1. Impact of Erosion on Productivity

Accelerated soil degradation jeopardizes food security by damaging the physical and biochemical functions of arable cultivated lands and affecting crop production (Lewis, Rowan et al. 2013). Organic matter and essential nutrients, such as nitrogen, phosphorus, potassium, and calcium, are detached during the erosion process. This impedes vegetative growth and reduces the health of soil biota. Fertile soil typically contains about 100 tons of organic matter per hectare, equivalent to about 4 to 5% of the total weight of the topsoil (Pimentel and Burgess 2013). When erosion occurs, the soil organic carbon is greatly depleted which causes the productivity of soil and the whole ecosystem to decline significantly. Poorly performing farmlands with diminished soil nutrients are either abandoned or are compensated by the addition of nitrogen and phosphate fertilizers. Apart from impacting fertility and productivity of agricultural yields, soil erosion disrupts vital soil services and causes a number of environmental problems (Liu, Yao et al. 2018). The decrease of soil structure due to crusting, compaction, and a decrease of microbiology decreases infiltration and causes runoff, which in turn increases erosion. Therefore, erosion is an indicator of soil degradation processes that affect soil structure and often implies a wider environmental and land-use problems.

The good news is the process is reversible: soil resources can be renewed over time, and fertility can be regenerated. In farm operations, conservation tillage can be adapted to restore soil structure and decrease erosion. Such measures include the use of riparian vegetation, terracing, hedging, check dams, mulching, and no-till farming (Wang, Yang et al. 2016, Pezzuolo, Dumont et al. 2017, Liu, Yao et al. 2018). But due to the associated costs and labor requirements, farmers and local communities are not always keen on investing resources that will implement these measures. To complicate the matter, the symptoms and effects of soil erosion are not always clearly perceptible, and therefore may not adequately be dealt with. Long term declines in productivity are not always immediately conveyed through yield decline (depending on the severity and form of erosion).

Thus it is essential that the impact of soil erosion is examined when contextualizing the economics of farm management. While there is little agreement as to exact nature of how erosion quantitatively impacts agricultural yield, there is a general acceptance that erosion lowers agricultural productivity (Moller and Ranke 2006). It would be advantageous that the production function that analyses the economics of farm management include the cost component of soil conservation and the impact of erosion risks in agricultural expenditures. One popular approach to estimate and analyze soil related risks is through empirically modeling of soil erosion. Through the use of soil assessment, the effect of erosion on farm expenditures can be analyzed implicitly, specifically the cost of investments towards soil conservation.

3.2.2. Survey Design

The survey was conducted in February to April 2015 using door-to-door questionnaire surveys, to describe the physical, socio-demographic, and agricultural economics of participating households. Using random sampling technique, 250 Norzagaray residents were selected using an additional list of criteria: (1) respondent is the head of household, and that he/she either owns a farm or rents/leases the farm; (2) has farmed (planted and harvested) for at least the last three years; and (3) willing to participate in the survey and was open to the idea of revisits for future diagnostic analysis of results. A number of the respondents had previously participated in previous valuation surveys related to this study. The survey questionnaire was partially based on the rice production survey questionnaire from the Philippine Statistics Authority and was modified in coordination with Norzagaray's Agriculture Office. In determining farmer's environmental awareness (E_A) score, a five-level Likert scale test[2] measuring the individual's understanding and willingness to implement environmentally sound farm management. With the help of the Provincial Government of Bulacan and some Municipal Agriculture Officers from the province, a pre-test was conducted with 10 individuals, which provided key insights on how to further improve and truncate the survey questionnaire. The modified version was pilot-tested with the staff from Norzagaray's Agriculture Office and some selected farmers. The estimated completion time for each respondent was about 45-50 minutes, and six personnel were hired and trained to assist in conducting the survey.

3.3. Results

3.3.1. Demographic statistics

In total there were 181 completed questionnaires equal to a response rate of 72%. This is considered to be reasonably high for this type of survey (extensiveness and length), which was achieved through deliberate acts of reducing non-response (e.g., concise and clear-worded questionnaire, well-trained staff, etc.). The sample population was overwhelmingly made up of men (88%), reflecting the gender trend in the heads of households in agricultural communities. The average age of respondents was 56 years old,

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[2] *The Likert Scale is a rating mechanism used to quantify people's perception or attitude towards a specific topic. In this study, a five-level scale was used ranging from strongly agree to strongly disagree, with a neutral option also provided.*

and the average farm size was 1.45 hectares. Although there were some small deviations between the sample demographics and the census data, the respondent population can be considered well representative of Norzagaray farmers.

3.3.2. Agricultural Inputs

The sample population for the survey was mainly rice-producing farmers. Aside from rice, the respondents were also harvesting a variety of other crops including root crops, coconut, vegetables, and fruits. The average rice production was calculated to be 3.95 tons[3] per hectare, equivalent to about P71,093.52 (using farm gate price of P18/kilo).

In terms of farm ecosystem, there was a small but significant difference in the rice produced per hectare between irrigated farms and non-irrigated farms. Irrigated farm production averaged 4.29T/ha while rainfed farm production has 3.82T/ha ($F=6.293$, $p<0.05$). In terms of seedling use, the average Norzagaray household used about 69.15kg/ha of seedlings, with homogenous supply between irrigated and non-irrigated farms. The use of fertilizer was common in Norzagaray averaging about 212.99 kg/ha, comprising mainly of inorganic type. The main fertilizer of choice was 'complete' (nitrogen (N) 14%, phosphorous (P) 14%, potassium (K) 14%), followed by 'Urea' ((N) 46%, (P) 0%, (K) 0%), and 'Ammonium Phosphate' ((N) 16%, (P) 20%, (K) 0%). Irrigated farms significantly were using more fertilizer averaging 235.62 kg/ha, as opposed to non-irrigated farms with mean fertilizer use of 204.49 kg/ha. Bivariate analysis showed farming experience to have a positive correlation with fertilizer use ($F=3.056$, $p<0.05$). This means that those who have worked for more years as farmers tend to use more fertilizer in farm production. On average, pesticides (e.g., insecticide, molluscicide, and herbicide) were used about 2.02 L/ha, which were mainly in liquid form. Irrigated farms have greater pesticide usage (2.81 L/ha), significantly higher compared to rainfed rice production (1.72 L/ha). For agricultural labor measured in man-days, the estimated mean was about 56.19 man-days/ha. Farm ecosystem was not a significant factor affecting the amount of labor required as there was minimal variation in labor inputs between non-irrigated (59.01 md/ha) and irrigated farms (55.14 d/ha). The summary statistics for the agricultural inputs are provided in **Table 3-1**.

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[3] *1 metric ton is equal to 1,000 kilograms*

Table 3-1. Summary of agricultural inputs of production and differentiation of values between irrigated vs non-irrigated farms

		Mean	Std Error	Min	Max	F	Sig
Seedlings (kg/ha)	Total	69.15	1.27	33.02	133.35	0.033	0.857
	Irrigated	69.53	2.49	33.02	118.04		
	Non-irrigated	69.01	1.49	36.02	133.35		
Fertilizer (kg/ha)	Total	212.99	4.89	60.58	449.89	8.266	0.005
	Irrigated	235.62	9.28	74.89	449.89		
	Non-irrigated	204.59	5.60	60.58	434.85		
Pesticide (L+kg /ha)	Total	2.02	0.06	0.38	5.00	87.982	0.000
	Irrigated	2.81	0.10	1.25	5.00		
	Non-irrigated	1.72	0.06	0.38	3.86		
Labor (md/ha)	Total	56.19	0.95	13.00	92.48	3.309	0.071
	Irrigated	59.01	1.54	33.35	92.48		
	Non-irrigated	55.14	1.16	13.00	90.02		

3.3.3. Total cost, revenue, and profit margin

Aside from seedling, fertilizer, pesticide and agricultural labor costs, other production-related costs were included in estimating the total cost (see **Appendix A** for more details). Other principal expenditure included irrigation fees, equipment cost (tractor and animal-labor), transportation, utilities, estimated agricultural rent, and the installation and maintenance of soil conservation strategies. The summary of the average agricultural expenditure calculated per hectare per planting cycle is shown in **Table 3-2**. Irrigated farms had higher production costs per hectare than non-irrigated farms, ₱42909.12 vs. ₱41165.38, although the difference was not statistically significant. The Pearson product-moment correlation coefficient was calculated in assessing the relationship between total farm expenditure and the socio-demographic make-up of the respondents. There was a positive correlation found on farming experience, r (181) = 0.193, p<0.01, which meant those with longer farming experience having greater farm expenditure. Also, there was a moderate correlation between costs and farm size, r (181) = -0.217, p<0.01, which meant that higher farm spending per hectare was increased over smaller farms.

Table 3-2. Summary of agricultural inputs differentiated between farm ecosystem type

	All Farms		Irrigated		Non-Irrigated	
	Mean (Std Error)	Percent	Mean (Std Error)	Percent	Mean (Std Error)	Percent
Seedlings	2074.52 (38.20)	4.98%	2085.83 (74.59)	4.86%	2070.32 (44.63)	5.03%
Fertilizer	5324.96 (122.32)	12.79%	5890.98 (232.16)	13.73%	5114.84 (140.06)	12.43%
Pesticide	1567.61 (32.74)	3.76%	2014.89 (47.70)	4.70%	1401.58 (30.53)	3.40%
Labor	21352.30 (361.19)	51.28%	22423.83 (586.81)	52.26%	20954.53 (441.02)	50.90%
Conservation	2337.22 (111.18)	5.61%	1544.61 (122.98)	3.60%	2631.45 (137.08)	6.39%
Miscellaneous	3980.83 (180.11)	9.56%	3948.97 (260.22)	9.20%	3992.66 (227.83)	9.70%
Land (est)	5000.00	12.01%	5000.00	11.65%	5000.00	12.15%
Total Cost	41637.44		42909.12		41165.38	

Almost three-fourths of the total farm expenditure can be attributed to agricultural labor (51%) and material inputs (22%). The visual representation is presented in **Figure 3-1**. Significant marginal difference between farm ecosystem types was observed in the allocation for soil conservation management ($p<0.05$). Non-irrigated farms allocated about 6.39% of their total expenses to soil conservation measures, while irrigated farms only spent about 3.60%. The average revenue per hectare per planting cycle was ₱71073.67 (~$1592.51) and is summarized in **Table 3-3**. Using Analysis of variance (ANOVA) to assess the revenue difference among socio-demographic groups, farm size, farming experience, ownership type, and farm ecosystem were found to have a significant effect on farm revenue. Farm size and farming experience were both positively correlated with farm revenue, with r (181) = 0.319 ($p<0.001$) and r (181) = 0.334 ($p<0.001$) respectively. Also, the type of farm ecosystem was found to have marginal influence on revenue ($F=6.293$, $p<0.05$), with irrigated farms having higher farm revenue than non-irrigated farms.

In terms of farm profitability, the average income was estimated to be ₱29,436.22 (~$659.56). Irrigated farms had statistically significant higher net profit (₱34,306.23) than non-irrigated farms (₱27,628.42) with $F=4.971$ ($p<0.05$). The profit margin, which is the ratio of income to revenue, was calculated to be 0.38 and was homogenous in farm-ecosystem type. Net income was found to have significant correlation with farm size (r (181) = 0.319, $p<0.01$), while profit margin had significant correlation with farming experience (r (181) = 0.294, $p<0.001$) and farm size (r (181) = 0.347, $p<0.001$).

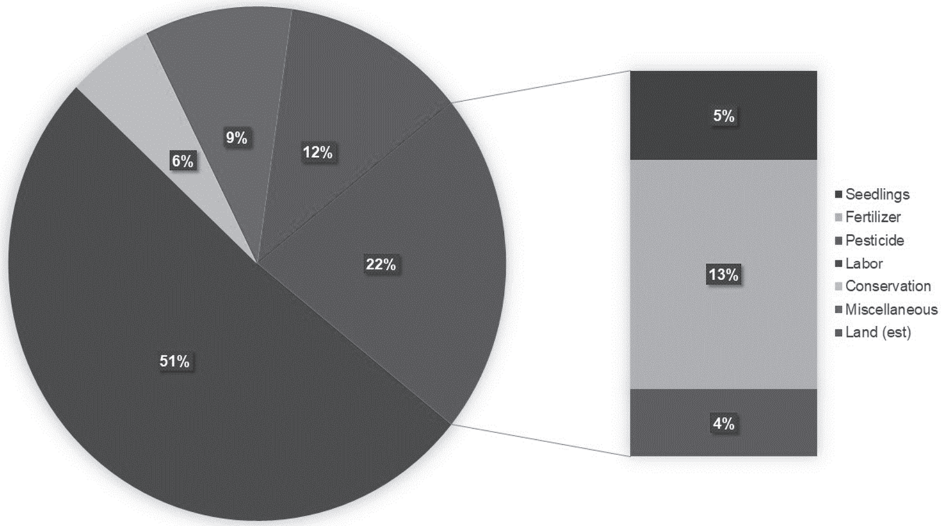


Figure 3-1. Chart showing the average expenditure allocations for all farm type

3.4. Environmental Consciousness Index

The main conservation measure used by the respondents was mulching, which was practiced by 41%, followed by strip cropping (25%), no-till farming (22%), terracing (14%), and intercropping (10%). The farmers also had high familiarity and working knowledge with mulching (62%), followed by strip-cropping (50%), terracing (49%), and no-till-farming (49%). Only 21% of the respondents said they had a working knowledge of intercropping as a means of soil conservation. The breakdown in percentages of users and those who have a working understanding of each conservation method is shown in **Figure** 3-2**2**.

Table 3-3. Gross Income as Response Variable and Socio-economic demographics as explanatory variables

	Gross Output (PhP/ha)	N	Std Dev	F	p-value
Size of Farm (interval)				8.223	0.005
Education (categorical)				0.233	0.873
w/o High School Diploma	70662.07	95	20733.88		
Graduated HS	70619.16	44	22077.15		
Technical School	71152.59	30	16402.30		
College Degree	75801.32	12	21667.14		
Farming Experience (ordinal)				11.655	0.000
<10 years	62711.63	20	15499.77		
11-20 years	59944.54	32	13422.55		
21-30 years	64256.89	31	15504.38		
>30 years	78570.53	98	21528.30		
Land Ownership Type (categorical)				3.421	0.019
Owned through Patent/AR	76578.55	45	24894.60		
Owned through Purchase	65996.28	48	15979.85		
Owned through Inheritance	66417.61	39	16532.52		
Rent / Lease	74697.76	49	20857.25		
Farm Ecosystem (categorical)				6.293	0.013
Irrigated	77215.35	49	22196.63		
Non-irrigated	68793.80	132	19230.31		
Terrain (ordinal)				1.190	0.307
Gentle (<8% slope)	68554.97	79	18052.44		
Moderate (8-30% slope)	73618.85	74	23149.06		
Rolling/ Hilly (>30% slope)	71453.44	28	18363.41		
Receiving Govt Assistance (dummy)				0.962	0.328
Yes	71940.46	135	20477.39		
No	68529.80	46	20021.96		

Farmgate Price for Rice used in calculation: PhP 18/kilogram
$1 (in April 2015) = PhP 44.63

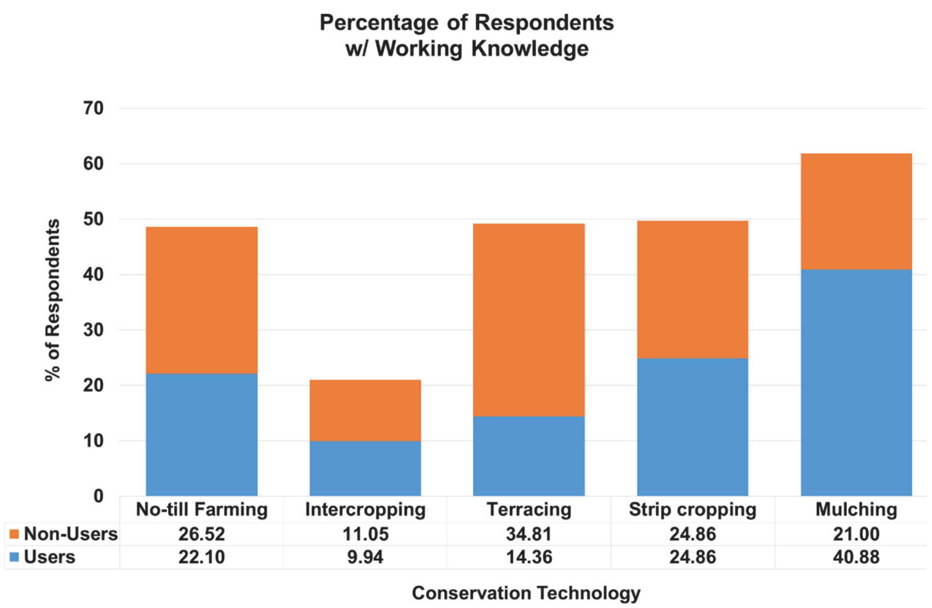


Figure 3-2. Summary of respondent population with working knowledge of conservation measure type

Main reasons hindering or minimizing the use and investment in soil conservation practices included labor requirements for establishing and maintaining conservation technology (44%) and the perception that soil erosion is not a major problem in their farms (23%). Other reasons included land-tenure issues (16%), and lack of technical know-how or support (12%), and concerns on profitability or economic returns (9%), as shown in **Figure 3-3**. In terms of soil conservation expenditure, the mean value for conservation among respondents was P2,337.22 per hectare (std. Error = 111.18) or about 5.61% of the total farm expenditure. Non-irrigated farms spent higher for soil conservation measures than irrigated farmland ($F=20.960$, $p<0.001$) with average values for conservation measures amounting to P2631.45 and P1544.61 respectively. The farming experience and farm size were found to have a significant influence on investments in conservation measures. Respondents who have had longer farming experience spent more in erosion mitigation practices ($F=5.863$, $p<0.05$). Those owning larger farms spent less on conservation measures per hectare compared those with smaller farms ($F=14.269$, $p<0.001$), with farm size having r (181) = -0.274 with conservation expenditure.[4]

[4] *For more details on correlating socio-economic demographics with gross income, see **Appendix B**.*

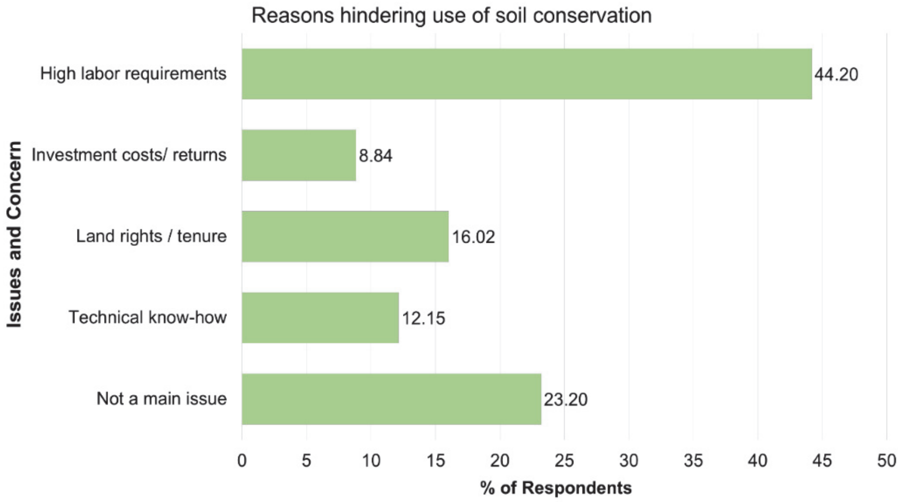


Figure 3-3. Percent of respondents citing their issues that hinder their use or investment in soil conservation measures

The farmers' environmental awareness score (E_A) was calculated by averaging the responses in the self-evaluation test. The mean E_A was calculated at 4.07, ranging from 2.00 to 5.00. The effect of the different socio-demographic factors in the respondents E_A score was analyzed. Farming experience was found to have significant correlation with the E_A score (Coefficient = 0.204, p<0.01). This suggests deeper appreciation for conservation among farmers, which could indicate some level of success on government initiatives for conservation measures and information drives. A one-way analysis of variance (ANOVA) was conducted and revealed significant relationship (F=3.749, p<0.05 level), specifically between farmers with experience of 20 years or less, and those who have been working in the farm for longer than 20 years.

The estimated average annual erosion is shown in **Figure 3-4 (see Chapter 4 for methodology)**. The eastern region has low erosion values since most of the area is covered with dense forests and has been delegated as a protected region. The western half has a more varied vulnerability values given its complex topography and varying land-cover types. Soil erosion is categorized into five classes: very low (<5 t/ha/yr), low (5-15 t/ha/yr), moderate (15-30 t/ha/yr), high (30-50 t/ha/yr), and very high (>50 t/ha/yr). In assigning the erosion vulnerability rating (E_V), the median classification value of the neighborhood 3x3 kernel was used. Since the respondent operates in a spatial environment, his perception regarding soil-related problems is similarly affected by the surrounding cells. The generated median value became the erosion vulnerability rating (E_V), which was then used in computing the farmer's consciousness (E_C) score. The erosion vulnerability was found to have significant moderate correlation with the farm income (*r* (181) = -0.154, p<0.05) and with the profit margin (*r* (181) = -0.161, p<0.05).

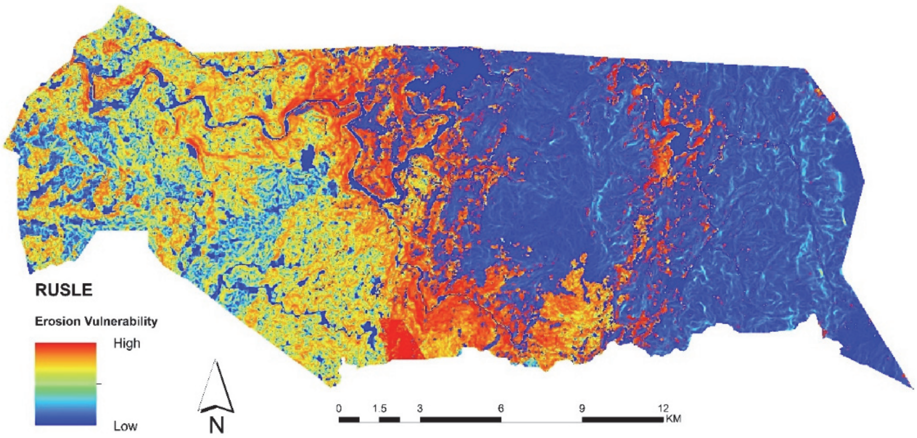


Figure 3-4. Resulting soil vulnerability map for Norzagaray

3.5. Econometric Models

The parameter estimates for econometric Models A[5], B[6], and C[7] are shown in **Table 3-4**. Model A demonstrates the classical thought in agricultural production, where the regressors are mainly the inputs of production and their interaction. Model B expands the initial model and considers the heterogeneity of farmer-stakeholders, as distinguished by socio-demographic and farm attributes. Model C includes all the variables of Model B and incorporates soil degradation parameters in the analysis. To estimate the goodness of fit of the different non-linear regression models, the standard error of the regression (SE) and pseudo-R2 values were computed. Based on the results Model 3 had the best fit, having the lowest S-value and highest pseudo-R^2 score. The visual interpretation of the predictive capability of Model 3 is shown in **Figure 3-5**[8]. This suggests that the inclusion of socio-demographic and environmental consciousness parameters as explanatory variables have significant positive influence in the regression model.

[5] *Model A views only classical inputs of production – seedling costs, fertilizer costs, pesticide costs, and labor costs*

[6] *Model B includes classical inputs plus stakeholders' demographic and farm attributes – education, farming experience, ownership type, farm ecosystem, government support, and topography*

[7] *Model C includes all inputs from Model B plus degradation parameters – conservation expenditure, level of erosion, and environmental awareness*

[8] *For further explanation on Model 1, 2 and 3, see **Appendix C, D** and **E** respectively.*

Table 3-4. Summary of parameter estimates for the different econometric models

	Parameter	Model 1		Model 2		Model 3	
		Esti-mate	Std Error	Esti-mate	Std Error	Esti-mate	Std Error
α	Intercept	9.002	11.100	7.827	10.718	7.801	10.258
β_1	Ln Seedlings Cost	1.237	2.471	1.051	2.402	0.984	2.306
β_2	Ln Fertilizer Cost	-1.839	1.890	-1.481	1.855	-1.446	1.761
β_3	Ln Pesticide Cost	0.761	2.194	0.243	2.176	0.750	2.087
β_4	Ln Labor Cost	0.842	2.371	1.186	2.319	0.713	2.215
β_{11}	Ln Seedlings x Ln Seedlings	0.379	0.434	0.428	0.418	0.548	0.398
β_{12}	Ln Seedlings x Ln Fertilizer	-0.247	0.507	-0.155	0.480	-0.352	0.457
β_{13}	Ln Seedlings x Ln Pesticide	0.575	0.597	0.516	0.570	0.404	0.547
β_{14}	Ln Seedlings x Ln Labor	-1.086	0.647	-1.218	0.629	-1.149	0.607
β_{22}	Ln Fertilizer x Ln Fertilizer	0.585	0.216	0.517	0.216	0.525	0.205
β_{23}	Ln Fertilizer x Ln Pesticide	-0.974	0.483	-0.778	0.477	-0.773	0.453
β_{24}	Ln Fertilizer x Ln Labor	0.051	0.577	-0.101	0.550	0.075	0.524
β_{33}	Ln Pesticde x Ln Pesticide	0.068	0.394	-0.061	0.379	0.105	0.361
β_{34}	Ln Pesticide x Ln Labor	0.264	0.556	0.383	0.541	0.158	0.525
β_{44}	Ln Labor x Ln Labor	0.386	0.253	0.468	0.245	0.458	0.234
γ_1	Educational Attainment			7.827	10.718	0.010	0.019
γ_2	Farming Experience			1.051	2.402	0.039	0.018
γ_3	Ownership Type			-1.481	1.855	-0.003	0.016
γ_4	Farm Ecosystem			0.243	2.176	-0.066	0.054
γ_5	Government Assistance			1.186	2.319	0.067	0.021
γ_6	Terrain			0.428	0.418	-0.024	0.026
γ_7	Government Support			-0.155	0.480	-0.029	0.042
δ_1	Ln Conservation Cost					0.113	0.031
δ_2	Erosion					-0.033	0.013
δ_3	Ln EAS					0.092	0.106
Standard Error (S_E)		0.183		0.173		0.160	
Pseudo R^2		0.356		0.444		0.501	

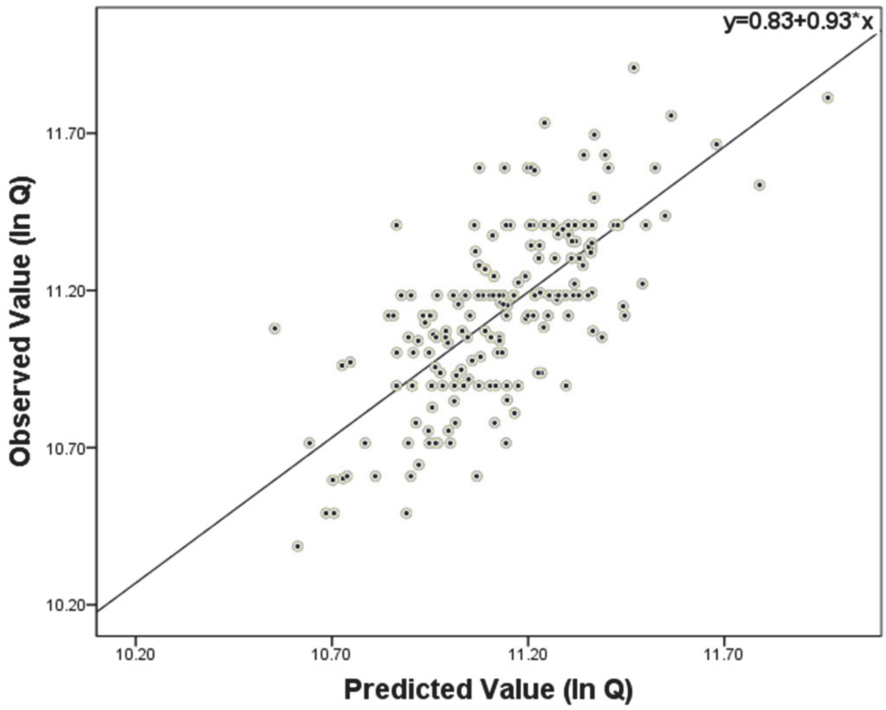


Figure 3-5. Predicted and Actual Output Values using Econometric Model 3

The parameter estimates in the regression models were used in calculating the mean output elasticities for the different explanatory variables and are presented in **Table 3-5**. In all three models, labor was estimated to highest output elasticity, positively correlated with productivity. This suggests that rice production in Norzagaray continues to be heavily reliant on agricultural labor. Since both the labor and seedling paramaters show significant positive elasticities, this suggests that seedling use (density of seedling) in many of the farms in Norzagaray are not optimally utilized resulting in lower crop production. Assuming the level of technology remains constant, increasing seedling use, which in itself is very labor intensive, would result in higher crop yield. The output elasticities of fertilizer and conservation were also found to have a significant positive association with crop yield. For pesticides, the output elasticity in all regression models was almost zero indicating that incremental change in pesticide use would not significantly affect productivity.

Table 3-5. Output elasticity of the explanatory variables in the different models

	Model 1	Model 2	Model 3	Model 3A (Irrigated)	Model 3B (Non-irrigated)
Seedling Elasticity	0.310	0.279	0.274	0.057	0.303
Fertilizer Elasticity	0.333	0.324	0.342	0.039	0.379
Pesticide Elasticity	-0.017	-0.038	-0.025	0.312	-0.078
Labor Elasticity	0.377	0.431	0.410	0.593	0.396
Conservation Elasticity			0.113	0.111	0.149

The output elasticities for each farm ecosystem type were computed to differentiate variable effects on productivity between irrigated and non-irrigated farms (shown as Model 3A and 3B in **Table 3-5**). Increasing more seedling and fertilizer use was found to have no significant impact on agricultural productivity for irrigated farms but had a significant effect on non-irrigated farms. The opposite was found true for pesticide use showing significant influence in crop production for irrigated farms but not for non-irrigated farms. Output elasticity for labor was significant in both farm ecosystem but was notably higher for irrigated farmlands. Conservation investment was positively correlated with productivity, but without significant difference between irrigated and non-irrigated farms.[9]

3.6. Discussion

The results reveal the role of soil conservation spending in the production function and on the total farm expenditure. Farms, on the average, allocated 5.6% of their total expenditure to soil conservation, with 82% of the respondents spending at least ₱1500. Non-irrigated farms spent significantly higher for protection mechanism by 70.4% more. In terms of its effect on productivity, there was no statistically significant relationship found between conservation expenditure and agricultural yield. However, the results revealed significant negative correlations between average annual erosion and farm income ($r=-0.155$), and between erosion and profit margin ($r=-0.161$). This suggests the adverse effects of soil erosion on profitability, which underscores the relevance of understanding soil-related threats in agronomics.

Another important finding is Norzagaray's dependency on high amounts of manual labor and low per hectare yield, reflective of the lack of modernization in the agricultural industry. High demand for labor is reflected by the large proportion of labor costs (51%) despite the relatively low cost of farm work

[9] *For more details on conservation expenditure correlated with demographic and farm attributes, see* ***Appendix F****.*

salaries (₱380/md ~ \$8.51/md). This makes the agricultural industry distinctively exposed from various market forces such as the growing mining and cement industries which could potentially reduce labor supply and increase labor costs. Also, while many of the upland land farms (mainly rainfed) produced a comparable yield to the low-lying farms (mainly irrigated), the inadequacy of infrastructure and access has elevated transportation expenses making them less profitable. This low economic profitability has become the underlying cause of poverty in much of world's rural communities (Gardner 2000), which further leads to exhaustive farming practices, intensified soil degradation, and other ecological problems (Blackie 2017). Reducing the high cost of material inputs could address some of the profitability issues, such as government support for agricultural modernization and infrastructure development, and collective action to provide financial, technical and production support for farmers.

The results of this study also have considerable implications for household-level farm management. Given the underlying assumption that the farmer manages his/her fields as a for-profit business, decisions are influenced by output demands, crop prices, cost of agricultural inputs, and potential value of alternative land use. Understanding the dynamics and associations of agricultural inputs with crop yield and profit is crucial in making decisions affecting the financial efficiency and economic viability of farms. For many agricultural communities, the availability of water and irrigation infrastructure are essential considerations in farm operations. In this study, the results indicated a small but significant difference in agricultural output between irrigated and non-irrigated systems, translating to about 12.3% greater rice production for irrigated against non-irrigated farms. The difference was also evident in the resulting output elasticities, which showed how marginal variations in material inputs affected total production. Increasing labor was shown to boost productivity for both farm types, but would have a greater effect on irrigated farms (ε=0.59) than non-irrigated farms (ε=0.40).

It was also important to connect the agricultural production function not just with yield but also with profit. Economic profitability, instead of yield, primarily affects the farmer's decision-making (Hatt, Boeraeve et al. 2018). This would include whether the farmer would in conservation measures and sustainable farming techniques. In the analysis of the profitability of farms, while there was statistically significant difference in agricultural yield between irrigated and non-irrigated farms, this did not translate into a substantial difference in net profit. This was because the cost of material inputs was much higher for irrigated farms, offsetting the edge in rice production over non-irrigated farms. Using bivariate correlation, farm size was shown to be a better determinant of farm profitability, with larger farms being more cost-effective and financially advantageous. This highlights the continuing challenges imposed by the further land fragmentation of farmlands in Norzagaray and the rest of the country.

Fragmentation of landholdings is considered as a major obstacle in agricultural development and is a typical feature in less developed agricultural systems (Tan, Heerink et al. 2006). As observed in previous empirical studies, land fragmentation alters the marginal outputs of agricultural inputs, which often leads to higher farm expenditures and lower productivity (Sklenicka, Janovska et al. 2014, Lu, Xie et al. 2018). Policies that promote the reconsolidation of small farm-holdings may be implemented in the future, but these are long-term goals that would require the support of the various stakeholders, and land reform and funding strategy by the government, which could pose logistical challenges (Jurgenson 2016, Janus and Markuszewska 2017). Concurrently, landowners can take action to mitigate the adverse consequences of highly fractured landholdings. One such measure is through the creation of farmer's cooperatives that can cost-effectively finance the upgrading and mechanization of smaller farm holdings.

3.7. Implications for Agricultural Policy

Various government programs and initiatives have been put in place by the local and regional governments, through the agriculture and the environmental planning offices to promote soil conservation. Developing free seminars that teaches sustainable farm management, providing financial aid to farmers that encourages soil protection, and supporting the reestablishment of the farmer's cooperative union are just some of the programs that the local government has implemented in the past ten years. However, soil erosion in farmlands remains to be a major problem in the community. Although the majority of the respondents allocated funds for conservation, there was no explicit correlation found between conservation spending and erosion vulnerability (modelled). High erosion risk areas did not necessarily translate into farmers' allocating greater expenditure spending for soil protection.

Several reasons can explain this disconnect. First, the cost of soil conservation measures are site-specific, and the selection is largely dependent on various biophysical and socio-economic parameters (Wang, Yang et al. 2016). And second, how the farmer views his exposure to erosion risks and how he chooses to respond may significantly vary. Prior experience and risk-appreciation can lead to very different appreciation of the need for soil conservation measures. Moreover, various impediments have been shown to hinder the conservation of soil resources. High labor requirement was found to be the primary reason impeding soil conservation measures, followed by the lack of recognition of erosion being a major concern, and land tenure issues. Providing financial subsidies and training support for farmers to implement conservation schemes in their farming practices would be short and medium-term policy responses to minimize erosion.

As earlier mentioned, the low economic viability in the agricultural industry could lead farmers to exhaustive forms of cultivation that are unsustainable and environmentally disastrous. In Norzagaray, low profitability coupled with the expansion of other industries providing alternative employment opportunities has contributed to the rising number of abandoned farmlands, especially in the highlands. Abandoned farmlands increase the exposure of soil from erosional agents. This can potentially reverse much of the gains brought about by the reforestation of upland forest reserves that have been undertaken to impede sediment discharge. Government action to address this growing concern is needed which could include changes in land-use rights, providing government subsidies, using or payment for ecosystem services schemes (see Narloch, Drucker et al. 2011, Schroeder, Isselstein et al. 2013, Wynne-Jones 2013). This highlights the delicate link between the advancement of farm profitability for the sake of the farmer and the promotion of environmental protection for the sake of the general public.

Aside from financial subsidies, land-use policies should be consistent, cost-efficient, ecocentric and should be promoting greater stakeholder engagement. One such land-use policy matter directly affecting soil conservation is farm sizes. As in many developing countries, the Philippines has aspired to democratize land-ownership through redistributiong of agricultural lands which would then elevate the farmers' living standards. Land tenure reforms also were seen to promote sustainable use, as landowners were more likely to invest in conservation measures than land-tenants. Its unintended environmental consequence, however, was that smaller farmlands have become less cost-efficient, leading to lower production yield which consequently led to unsustainable exhaustive farming practices. If environmental sustainability is to be achieved, policies should be catered towards minimizing the occurrence of uneconomical farm sizes while still protecting the rights and welfare of the farmers.

Other policy recommendations that came out as a result of this study that require further attention include: (1) resolving the high mismatch between farm gate prices and market prices of agricultural produce that greatly diminishes farm profits; (2) developing a payment-for-ecosystem services to partially subsidize soil conservation measures specifically for upland farms; (3) integrating environmental vulnerability assessment as prerequisite in land-use and economic planning; and (4) strengthening farmers' environmental awareness specifically those receiving government support.

3.8. **Conclusion**

This chapter quantitatively analyzed factors affecting agricultural household spending, and the effect of soil conservation in his management decisions. In developing the agricultural production function, material inputs, stakeholder attributes, and environmental consciousness factors were incorporated into a modified trans-log Cobb-Douglas equation.

This study showed the relevance of integrating economic theories and environmental valuation with pedologic and natural hazard studies. While micro-economics have long established the importance of social parameters in economic analysis, the inclusion of physical models (i.e., environmental vulnerability modeling) has only recently been applied in the analysis. The use of spatial techniques, specifically the use of GIS, in econometric analysis allows us a greater understanding of the effect of location not just in modeling environmental vulnerabilities but also in influencing economic decision-making. And with the growing trend of greater availability of maps and remote sense images that allow us to spatially detect more measurements indirectly, the fusion of spatial statistics and environmental science with agricultural and household economics will become more pronounced.

National and regional economic progress should be accompanied by rural development and improvements in the agricultural sector. Ensuring the profitability and efficiency of farms is a major socio-economic and public safety issue given its various implications for the environment and food security. The knowledge and understanding derived from this study can be used to support policies and economic decisions that can balance profitability options for the farmers with the need to protect the environment and preserve the various soil functions. It also allows local authorities to develop more cost-effective assistance that would promote production and be economically beneficial to the farmers. The results of this study can also help in constructing blueprints to operationalize and systematize production function to be further improved as a valuation approach for soil and other environmental goods.

Chapter 4. Estimating WTP using Open-Ended CVM

This chapter explores the use of contingent valuation (CVM) approach in determining the willingness to pay (WTP) for soil conservation. It examines the various socio-demographic and soil degradation determinants to stated value. By presenting a methodology that assesses human and environmental risk factors affecting soil economic value, it builds on the growing body of knowledge in soil valuation and environmental economics in general, and it provides for stronger argumentation for more stakeholder participation in the valuation process.

4.1. Introduction

Generating stakeholder participation is an essential component for long-term soil use planning and management (Enloe, Schulte et al. 2014, Knoot, Larsen et al. 2014). Public engagement promotes greater commitment and cooperation and provides a rich source of local insights useful in the decision-making process (Russell and Ward 2016). It reduces distrust among the different stakeholders and government agencies (Tsang, Burnett et al. 2009). In soil use, understanding stakeholders' preferences, cognition, and attitude is key to the effectivity and sustainability of soil conservation management. The economic valuation of the environment can be a powerful instrument in expressing the direct and indirect benefits of environmental goods and can be used to promote sustainability and environmentalism. Values are individual or collective norms that can subjectively discuss relative desirability and objectively establish a universal hierarchy among things (Salles 2011). Stakeholders may have low inclination for conservation that may be against their own self-interests due to classic market failures that result in many environmental benefits not to be perceived by the community (Schiappacasse, Nahuelhual et al. 2012). In environmental valuation, one approach to gauge stakeholders' economic perception of environmental utility is through stated preference. Stated preference measures economic value through direct solicitation of stakeholders using carefully worded questionnaires or interviews. This allows the assessment of stakeholders' awareness and inclination while promotes public participation in land-use management.

The most dominant of the stated preference techniques is the contingent valuation method (CVM). CVM is a direct flexible technique that has been widely used in a variety of environmental applications including cost-benefit analysis, environmental impact assessment, and infrastructure development (see Carson, Mitchell et al. 2003, Afroz and Masud 2011, Madureira, Nunes et al. 2011, Zhao, Liu et al. 2013, Khan, Brouwer et al. 2014, Tussupova, Berndtsson et al. 2015). A well-executed CVM study can provide a vast amount of welfare information fundamental in economic and environmental decision-making (Venkatachalam 2004). Since CVM is heavily reliant on stakeholder response, particular attention is required on questionnaire design, which would require the use of focus group discussions, expert consultations, and pre-testing (Birol, Koundouri et al. 2008). Unfamiliarity with the environmental good could result in methodological misspecification and information bias that can distort WTP estimates (Barkmann, Glenk et al. 2008). Proper orientation and suitable questionnaire design should be taken to ensure that the respondents understand the good being valued and the payment scheme being proposed (Evans, Banzhaf et al. 2008). Some recommended protocols in CVM questionnaire development and implementation scheme can be found in Arrow, Solow et al. (1993), Bennett, Mele et al. (2010), Venkatachalam (2004) and Whittington (1998).

In this chapter, an open-ended CVM study was conducted in Norzagaray, Philippines to better understand how the stakeholders perceived and recognized soil economic value. In this approach, a hypothetical scenario was presented to the respondents, followed by a direct solicitation of their WTP for soil conservation. The main objective of this study was to estimate the average WTP for soil conservation which would be used as proxy to the soil's stated value. The calculated WTP value would also be used to analyze how socio-demographic attributes and soil-related vulnerabilities influence stakeholders' stated soil value, which is also a primary objective.

4.2. Specifications

For this study, a focus group discussion (FGD) composed of six members representing the relevant stakeholder groups was undertaken prior to the implementation of the survey. The members included farmers, which were represented by leaders from the agricultural cooperative union, staff members from the municipal agriculture office, and a barangay (neighborhood) official. The primary objectives of the FGD were to check the comprehensibility of the questionnaire, to determine possible areas of confusion, and to elicit further suggestions and recommendations. A free discussion format was used to examine the draft questionnaire. Revisions were undertaken that produced a final version which was then pre-tested on a small group of 10 farmers. The final version was accomplished with very little clarifying questions from the pre-test group and was completed at an average rate of 27 minutes.

4.2.1. Survey questionnaire and implementation

The payment vehicle used in the study was a public fund that will be used to mitigate soil degradation in the community. The fund will be used to fund soil information outreach to teach farmers about identifying soil problems and solutions on how to control them. The fund will augment local initiatives on supporting sustainable agricultural production and environmental conservation, focusing primarily on soil protection. The payment vehicle was chosen deliberately based on the farmers' familiarity with this type of payment system. The respondents must have prior knowledge and experience on the payment vehicle, and find it to be credible, realistic, and well-defined to minimize hypothetical bias (Flores and Strong 2007, Evans, Banzhaf et al. 2008).

Different socio-economic and farming-related attributes were elicited in the questionnaire. A variety of factors have been shown to influence WTP in previous studies including income levels (Guo, Liu et al. 2014), gender (Ferreira and Marques 2015), age (Han, Yang et al. 2011), household size

(Ojeda, Mayer et al. 2008), and the perceived level of risk (Khan and Damalas 2015). Identifying significant WTP determinants can be very useful in decision-making especially in planning for soil conservation and land-use management. After the results of the questionnaires were encoded and analyzed, a post-survey discussion with some respondents was conducted to contextualize the results. The house-to-house survey was conducted between January to March 2015 in Norzagaray with the help of the Norzagaray Municipal Agricultural Office (MAO). For assistance, six MAO personnel were trained and commissioned for the survey. A total of 200 farmers composed of farmer-landowners and farmer-tenants, were randomly chosen from Norzagaray's 11 agricultural barangays. A short discussion introducing the objectives and extent of the research was conducted before each survey.

With an open-ended questionnaire format, protest responses and extreme values are to be expected. Protest bid or zero-value response is often associated with the free-rider syndrome, where respondents provide skewed representations of WTP when they expect to be burdened by costs (Green, Jacowitz et al. 1998). It is common practice in CV studies for protest bids to be excluded from WTP computations given that they are not necessarily reflective of the respondents' normative preferences (Grammatikopoulou and Olsen 2013). On the other end, extremely high values are not uncommon to WTP studies, especially when respondents do not expect the results to be implemented on them in the foreseeable future. Like protest bids, these values are often masked out in the final WTP analysis. But to completely remove protest bids and extreme values altogether, may not necessarily be judicious. Some have argued that these values provide essential information on actual stakeholder preferences. In this study, three datasets were used to cater for different scenarios. Aside from Dataset1 (N=174) which included all WTP from fully accomplished questionnaires, two additional datasets were created: Dataset2 (N=167) excluded all zero responses (= ₱0.00), and Dataset3 (N=159) excluded zero and extreme values (> ₱700.00).

4.2.2. **WTP Analysis**

To analyze the results, the utility variation model proposed by Hanemann (1984) was used as the econometric model to estimate individual WTP:

$$\boldsymbol{y_i}^* = \boldsymbol{\beta x_i} + \boldsymbol{\lambda_i} \qquad \textbf{Eq 4-1}$$

where y_i^* is the individual's WTP variable, x_i is a vector of the individual's attributes and suggested plan, β is the coefficient for the attributes, and the λ_i is the error term with the mean equal to zero (Alberini 1995).

However, in econometric analysis of CVM surveys, some WTP will have to be censored at times, when the measurement is known to exceed or fall below a certain threshold (Yang, Zou et al. 2014). For such analysis, the Tobit model could be used to account for variables that go beyond the upper or lower limits and which will take on the limiting value (Tobin 1958). Using the upper censoring threshold to 700, the latent variable regression can be determined using the equation:

$$\begin{cases} \boldsymbol{y = \beta x + \lambda} & \boldsymbol{if\ y < 700} \\ \boldsymbol{700} & \boldsymbol{if\ y \geq 700} \end{cases} \qquad \textbf{Eq 4-2}$$

Substituting y, x and β with y', x' and β' the equation becomes:

$$\boldsymbol{y' = 700 - y} \quad \boldsymbol{x' = \begin{bmatrix} -x \\ 700 \end{bmatrix}} \quad \boldsymbol{\beta' = \begin{bmatrix} \beta \\ 1 \end{bmatrix}} \qquad \textbf{Eq 4-3}$$

which transforms to the standard form of the Tobit model:

$$\begin{cases} \boldsymbol{y' = \beta' x' + \lambda} & \boldsymbol{if\ y' > 0} \\ \boldsymbol{0} & \boldsymbol{if\ y' \leq 0} \end{cases} \qquad \textbf{Eq 4-4}$$

where $\beta'$is the Tobit regression coefficients, x' is the vector of independent variables, λ is the random standard term, and where the dependent variables are all left-censored (all $y' \leq 0$ is encoded as 0).

4.3. Impact of Soil Related Risks

4.3.1. Soil Erosion

In implementing the erosion vulnerability model, an updated formulation of the of the Universal Soil Loss Equation (USLE) was used. The Revised Universal Soil Loss Equation (Renard, Foster et al. 1997) was used to estimate potential soil erosion for Norzagaray. The RUSLE is essentially a sediment 'production' function from fields and slope elements; it does not simulate transport and deposition of sediment, only sheet and rill erosion. Therefore, it is not suitable as an indicator of downstream siltation effects and river sediment loads. In this study, the simulated erosion rates were compared to the erosion experienced by farmers in their fields (average farm size 0.75-3 ha). In this context, the RUSLE was assumed to give a usable relative representation of annual erosion, comparable to the farmers' experience. The model for erosion vulnerability (V) used in this study is:

$$V = R\,K\,L\,S\,C\,P \qquad \textbf{Eq 4-5}$$

where V is the annual soil loss in tons per hectare; R is raa infall-runoff erosivity factor; K is the soil erodibility factor; L is slope length factor; S is the slope steepness factor; C is the cover management factor; and, P is the support practice factor. In this study, the RUSLE model was implemented using 30meter grid cells. The RUSLE offers a simple and clear methodology to estimate soil erosion potential that does not require the need for complex data (Renschler, Mannaerts et al. 1999).

Rainfall-runoff erosivity factor (MJ mm $ha^{-1}h^{-1}yr^{-1}$) is the parameter approximating the erosive impact of precipitation on soil (i.e., raindrop impact, surface run-off effects). Daily rainfall data of Angat Dam from 1964-2014 were used in calculating for R using the equation (Ganasri and Ramesh 2016)

$$\boldsymbol{R = \sum_{i=1}^{12} 1.735 \times 10^{\left(1.5\, log_{10}\left(\frac{P_i{}^2}{Pa}\right) - 0.08188\right)}} \qquad \textbf{Eq 4-6}$$

where P_i is the monthly rainfall (mm); and P_a is the annual rainfall (mm). The rainfall data was obtained from the Watershed Division of the National Power Corporation, which manages the operation of the hydroelectric power plant.

The Soil erodibility factor (K) gages the susceptibility of soil to erosion as influenced by texture, structure, organic matter content, and soil permeability. The erodibility factor was computed following the equation (Foster, Mccool et al. 1981):

$$\boldsymbol{100K = \left(2.1m^{1.14} \times 10^{-4}(12 - a)\right) + \left(3.25(b - 2)\right) + \left(2.5(c - 3)\right)} \qquad \textbf{Eq 4-7}$$

where m is (silt (%) + very fine sand (%)) (100-clay (%)); a is organic matter (%); b = soil structure; and, c = soil permeability class. In this study, the geologic/soil map from the Bureau of Soil and Water Management was used, with additional soil data provided by Municipal Agriculture Office.

The slope length and steepness factor (LS) estimates the effect of topography on the area's vulnerability to soil erosion. Slope length (L) is measured from the origin of the overland flow along its flow path to the location of the concentrated flow or deposition. The slope steepness (S) is the ratio of soil loss from the field gradient with 9% slope under identical conditions. In this study, four ASTER-GDEM images were processed, georeferenced, and mosaicked in generating the digital elevation model (DEM). The DEM was then used to estimate the combined effect of slope length and steepness with the formula (Moore and Wilson 1992):

$$\boldsymbol{LS = \left(FA\ x\ \frac{CS}{22.13}\right)^{0.4} + \left(\frac{Sin\ (S\%)}{0.0896}\right)^{1.3}} \qquad \textbf{Eq 4-8}$$

where FA is flow accumulation; CS = size of each raster cell; and, S% = topography's degree of slope.

The cover management (C) factor assesses the impact of varying cropping practices and management on erosion rates. It and is commonly used to compare the relative impact of various conservation plans. LandSat8 images captured on 08-February 2014 were used in calculating C. Utilizing Normalized Difference Vegetation Index ($NDVI$), C was estimated using the equation (Gutman and Ignatov 1998).

$$C = 1 - \frac{NDVI - NDVI_{min}}{NDVI_{max} - NDVI_{min}} \qquad \textbf{Eq 4-9}$$

Since the equation only applies to areas with vegetation, other regions were initially masked out in determining the value of C using this equation. The c-value used for these land cover types was based on the values recommended by David (1988). The support practice (P) factor represents the reduction in soil erosion due to conservation measures such as contour farming, strip cropping and terracing. In this study, no information on the extent of support practices was available, and therefore the value of P was pegged to the value 1.

Estimated soil loss in tons/hectare/year was generated in raster mode using ArcGIS10 software. To differentiate soil erosion risk values, the vulnerability results was categorized into five classes: very low (<5 t/ha/yr), low (5-15 t/ha/yr), moderate (15-30 t/ha/yr), high (30-50 t/ha/yr), and very high (>50 t/ha/yr). The generated soil erosion map is shown in **Figure 4-1**.

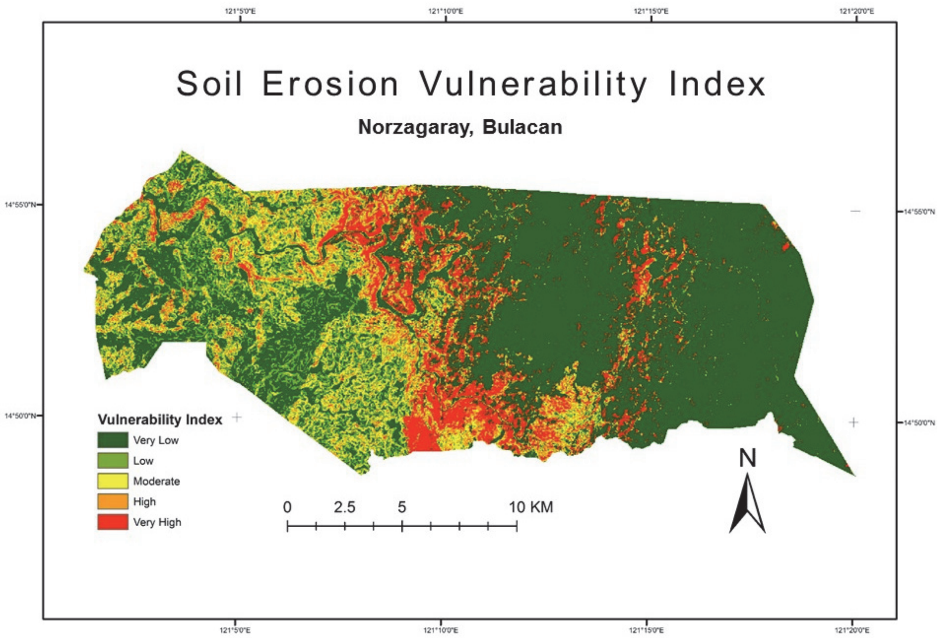


Figure 4-1. Generated soil erosion vulnerability map for Norzagaray, Bulacan

4.4. WTP Results

The perceived levels of different soil-related risks were examined on how they influence farmers' WTP. Previous studies have shown that respondents who perceive greater risk to their personal well-being can be more inclined to give higher WTP values (Yang, Zou et al. 2014, Khan and Damalas 2015). Aside from soil erosion, five other soil degradation risks were identified during the

FGD. These included nutrient loss, landslides, compaction, acidification, and waterlogging. Respondents were asked to identify which soil-related risks they perceived to be of personal concern. The summary of results is presented in **Figure 4-2**.

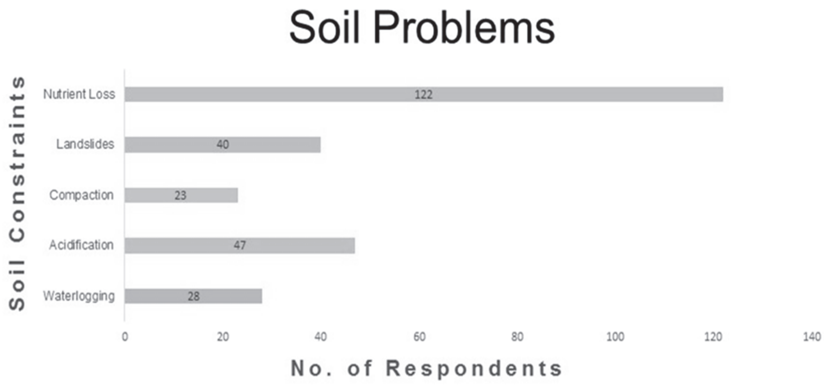


Figure 4-2. Chart showing number of respondents claiming soil problem type

4.4.1. WTP and socio-demographic determinant

Of the 200 surveyed respondents, 174 questionnaires were filled entirely in reflecting a completed response rate of 87%. The respondents' general profile (age, household size, education, income and farm size) was largely comparable to the profile of the community. There was a slight oversampling of union members and farmers with irrigated lands compared to municipal data. As for the other attributes, there was no econometric data available from the Norzagaray local offices for comparison. The summary of the respondents' socio-demographic composition is shown in **Table 4-1**.

The mean WTP, which was packaged as a yearly contribution, was calculated to be P192.04 ($4.29) with a standard error of 18.75. The mean WTP for Dataset2 representing all non-zero WTP responses was P200.09 ($4.47) with a standard error of 19.29. The mean WTP excluding zero and extreme values was P153.70 ($3.43) with a standard error of 12.38. For context, the minimum daily farm wage for the province is ₱390 ($8.71). The WTP estimates indicate that there is considerable support for the soil conservation fund.

Various socio-economic, and farming-based indicators were examined using non-parametric correlation analysis to test their impact on WTP. The Spearman-rho correlation coefficient was used to test the null hypothesis of no relationship between WTP and the different explanatory variables. Tests were conducted on all three datasets, and the summary of results is presented in **Table 4-2**. In all datasets, gross income, support from the government, farm ecosystem, and the method of farm possession were found to have significant effect in explaining WTP variability. To further analyze how these demographic attributes affect WTP, the mean WTP for each of their categories were computed and analyzed. Gross income was shown to be determinative of WTP heterogeneity. The mean WTP for farmers earning more than P200,000 ($3800) was significantly lower than the rest of the other income-groups. Higher income households are more likely to have more discretionary expenditure which they can spend for conservation measures. They are also perceived to have greater stakes in ensuring soil health as compared to those who earn less.

Table 4-1. Socio-demographic make-up of the respondents

Paramater Type	Characteristics	Details
Demographics	Gender	Male (89%); Female (11%)
	Age	18-30(4%); 31-40 (11%); 41-50 (16%); 51-60 (33%); >60 (36%)
	Household size	1-2 (15%); 3-5 (48%); 6-8 (25%); >8 (12%)
	Education	Did not finish high school (52%); high school (26%); technical school (16%); college/university (6%)
	No of Years in Agri-industry	1-5 years (10%); 5-15 years (17%); 15-25 (20%); >25 years (53%)
Income	Gross Annual Income	<P100,000 (60%); P100,000-P200,000 (22%); P200,000-P300,000 (8%); >P300,000 (10%)
	Percent of Income from farm	>90% (45%); 70-90% (30%); 50-70% (19%); >50% (6%)
External Support	Govt Assistance	Receiving Govt assistance (73%); Not receiving Assistance (27%)
	NGO Support	receiving support (14%); not receiving support (86%)
	Coop Union Membership	member of coop (26%); non-member of coop (74%)
Farm Details	Size of farm	<2 hectares (71%); 2-4 hectares (22%); >6 (7%)
	% of land used in Agriculture	>80% (48%); 60-80% (42%); <60% (10%)
	Farm Ecosystem	Irrigated Farms (24%); Rainfed (76%)
	Crop Type	Mainly Rice (50%); Mainly Corn/Vegetables (45%); Mixed (5%)
	Terrain	Flat (43%); Gentle (40%); Hilly (17%)
	Ownership Type	owned thru purchase (25%); owned thru inheritance (25%); lease/rented (22%); owned thru judicial patent/homestead (28%)

Farmers supported by government mainly through financial subsidies and free training were shown to give lower WTP. Government support is implicitly linked with household income and years of farming experience: low-income farmers generally become recipients of financial support, while those with less farming experience tend to be the beneficiaries of technical training. The WTP difference between those receiving and not receiving government support could be explained in a number of ways. While financial constraints could arguably be a factor, reliance on government to finance conservation could have contributed to the lower WTP of beneficiaries. Coupling sustainability instruction in government projects should be promoted to curtail the culture of total dependence on government support.

With regards to agricultural ecosystems, farmers of irrigated lands provided higher WTP than those working on rainfed farms. Irrigation can affect crop productivity, crop options, and economic efficiency. But research on Philippine agricultural ecosystems found that rainfed farms differed only slightly in productivity against irrigated farmlands over time (Mariano, Villano et al. 2010), which also holds true for Norzagaray. Economic efficiency, therefore, could not be used to account for the difference in WTP; instead, spatial factors such as topography, geography, and proximity to environmental amenities need to be analyzed further to contextualize WTP variability.

Table 4-2. Spearman's rho correlation coefficients for different factors

	N=174 Correlation	N=167 Correlation	N=159 Correlation
Gender	0.140	0.123	0.099
Age	0.137	0.152	0.142
Household Size	0.175 *	0.194 *	0.164 *
Education	0.001	0.005	-0.001
Gross Income	0.225 ***	0.236 ***	0.250 ***
Period as Farmer	0.162 *	0.173 *	0.149
Income Dependency	0.087	0.113	0.097
Government Support	0.253 ***	0.251 ***	0.209 **
NGO Support	-0.011	- 0.044	-0.084
Membership Support	0.043	0.018	-0.008
Size of Farm	0.150 *	0.115	0.080
Percent of Land used for Farming	0.018	0.010	0.013
Farm Ecosystem	-0.232 ***	-0.261 ***	-0.237 ***
Type of Crop	0.049	0.038	0.049
Terrain	-0.049	-0.135	-0.099
Ownership	-0.408 ***	-0.414 ***	-0.351 ***

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

With regards to land ownership, farmers who acquired their lands from agrarian reform and homestead patents gave significantly higher WTP values, while land tenants were more likely to provide lower values. Similar to the results of previous research showing the effects of land tenure to soil conservation (Tefera and Sterk 2010, Sklenicka, Molnarova et al. 2015, Ayamga, Yeboah et al. 2016, Lovo 2016), the type of land ownership was found to have significant correlation with WTP values. This suggests that land reform and tenancy systems should have implicit social and environmental underpinnings.

4.4.2. WTP and perceived erosion risk

Respondents were asked if they considered soil erosion to be a particular problem in their farmlands. Those who answered yes were then requested to rank their level of concern from 1 (moderate) to 3 (very high). Their perceived level of concern on soil erosion was then compared with their WTP. The results of the Spearman correlation analysis as shown in **Error! Reference source not found.** for Datasets 1 and 3.

Those who did not consider soil erosion to be a concern gave significantly lower WTP values compared to those who did. However, when the 'yes' replies were further analyzed, the given ranked responses did not indicate meaningful differentiation. This suggests that while perceived threats from erosion could have considerate influence on people's willingness to pay for conservation, the perceived intensity of risk may not be as determinative. People have different perception level of what constitutes as being moderate or very high, and this result in high variability and weak correlation with WTP.

Table 4-3. Perceived Level – Erosion as Problem

	Dataset 1		Dataset 3	
Erosion Category	N	WTP	N	WTP
No	101	134.36	96	104.90
Yes	73	266.73	63	235.64
1 – Moderate	23	275.00	19	227.63
2 – High	21	286.67	18	278.89
3 – Very High	29	258.62	26	211.54
	R = 0.316*		R = 0.386**	

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

4.4.3. WTP and Soil Threats

Among the soil threats that were asked from the respondents, two were found to have considerable effect on WTP – nutrient loss and landslides. The summary of the correlation analysis between perceived soil threats and WTP is provided in **Table 4-4.** Respondents who considered nutrient loss or landslide risk to be of personal concern gave higher WTP than those who didn't. Nutrient loss is considered as one of the most easily recognizable consequences of soil erosion that have a direct effect on long-term productivity (Samarakoon and Abeygunawardena 1996, Martínez-Casasnovas and Ramos 2006). Essential soil minerals such as nitrogen, phosphorus, and potassium need to be substituted using fertilizers that could result in higher costs (Bertol, Guadagnin et al. 2004). Conservation measures that can minimize nutrient loss could be used as a cost-efficient alternative to fertilizer overuse. This could explain why perceived threat of nutrient loss is determinative on the willingness to pay for conservation.

Likewise, the threat from landslides was shown to be positively correlative with WTP. People who felt concern from landslide risks gave higher WTP values compared with those who didn't perceive it to be a concern. Landslides have the potential to inflict enormous casualties and substantial economic losses especially in the mountainous regions (Dai, Lee et al. 2002). Similar to nutrient loss, soil conservation could reduce landslide risks and prevent devastating losses, which could explain the proclivity for higher WTP values.

Table 4-4. Correlation of different soil related problems and WTP averages

		Dataset 1		Dataset 2		Dataset 3	
		N	Mean	N	Mean	N	Mean
Nutrient loss	Correlation (r)	-0.203 **		-0.177 *		-0.189 *	
	Yes	122	214.39	119	219.79	113	178.36
	No	52	139.62	48	151.25	46	103.48
Landslides	Correlation (r)	-0.201 **		-0.174 *		-0.112	
	Yes	40	297.25	40	297.25	35	182.57
	No	134	160.63	127	169.49	124	149.4
Acidification	Correlation (r)	-0.103		-0.115		-0.080	
	Yes	47	252.34	45	263.56	41	192.71
	No	127	169.72	122	176.68	118	144.53
Compaction	Correlation (r)	0.052		0.054		0.057	
	Yes	23	150.87	22	157.73	21	117.62
	No	151	198.31	145	206.52	138	162.64
Waterlogging	Correlation (r)	-0.107		-0.112		-0.103	
	Yes	28	241.61	27	250.56	25	190.60
	No	146	182.53	140	190.36	134	150.37

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

4.4.4. WTP and modelled erosion average annual erosion

Aside from inquiring perceived levels of soil threats, modeling soil degradation could be used to explain the variation in WTP responses. In this study, soil erosion was modeled and was used to analyze the effect of soil erosion vulnerability on stated value. Using Spearman-rho correlation analysis, the results revealed a strong monotonic association between simulated erosion and WTP for both Dataset1 (r=0.631) and Dataset3 (r=0.584). Farmers with low and very low erosion levels provided significantly lower WTP, while those with high and very high vulnerabilities gave much higher responses. The summary of mean WTP is shown in **Table 4-5**.

Table 4-5. Mean WTP for Datasets 1 and 3 at Different Erosion Categories

Erosion Category	Dataset 1		Dataset 3	
	N	WTP	N	WTP
Very Low	49	63.78	44	71.02
Low	47	91.49	46	93.48
Moderate	22	170.91	22	170.91
High	30	377.67	26	301.15
Very High	26	419.23	21	280.95
	r = 0.631 ***		r = 0.584 ***	

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

To ensure that the erosion figures are comparable to farm-level conditions, the modeled estimates were resampled to be comparable to the average farm sizes in Norzagaray (0.75 – 3 hectares). Each data point (ground area = 900m^2) was recomputed by taking the mean value of the surrounding cells (visualization provided in **Figure 4-3**): first by using a 3x3 grid (ground area =8100m^2) and then second, by using a 5x5 grid (ground area=22500m^2).

Using Spearman's rho analysis, both resampled erosion estimates showed significant correlation with WTP. The summary of results is shown in Table 4-6. The values from the resampled 3x3 grid showed moderate monotonic association with WTP (r=0.516) which was similar to the results of the initial estimation. The 5x5 grid estimates, however, exhibited only weak association between the modeled erosion values and WTP. While this study does not make any determination on which model provided the most accurate representation of erosion risk, what could be recommended from the results is that the 3x3 grid resampling provides a viable option when analyzing WTP heterogeneity. The main problem with using just the value of the primary cell is that it might not be comparable to the general conditions of the whole farm. It may also be highly susceptible to spikes in value caused by errors from data gathering or image classification. The 5x5 resampled grid markedly modulated erosion estimates that it reduced the monotonic correlation between WTP and erosion risks. For these reasons, the averaging of values of the neighboring cells can provide a more pragmatic approximation.

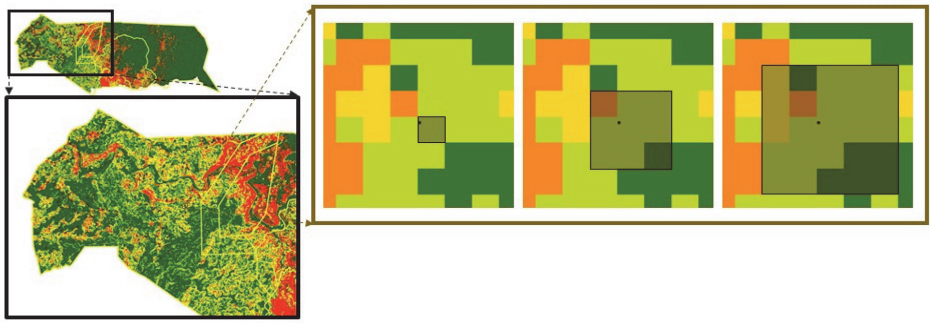


Figure 4-3. Visualization of aggregation technique used in analyzing single cell (900 sq m), 3x3 cell (8100 sq m) and 5x5 (22500 sq m) cell

Table 4-6. Summary of correlation coefficients for 3x3 and 5x5 grid resampling

	Neighborhood (3x3)				Neighborhood (5x5)			
	N	WTP (all)	N	WTP (159)	N	WTP (all)	N	WTP (159)
Very Low	37	108.24	33	91.06	35	108.71	30	93.5
Low	51	101.67	47	89.04	44	143.64	40	108
Moderate	44	216.93	40	163.63	59	208.64	56	166.25
High	34	295	33	273.64	29	283.79	27	249.26
Very High	8	581.25	6	358.33	7	392.86	6	291.67
	r = 0.516 ***		r = 0.515 ***		r = 0.371 ***		r = 0.366 **	

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

4.4.5. **Cluster Analysis**

Cluster analysis could be a useful mechanism to further investigate the variability of WTP especially in areas with fuzzy subgroupings. Environmental variables are naturally distributed in space which means that analyzing their geographic distribution is crucial in spatial analysis (Hutchinson 2008). In this study, the k-means clustering (non-hierarchical) analysis was employed in regrouping the sample population into three subsets to account for the geography and topography. The Euclidian distance between subjects x_i and x_j was computed using:

$$\boldsymbol{d_{ij}} = \sqrt{(\boldsymbol{x_{iN}} - \boldsymbol{x_{jN}})^2 + (\boldsymbol{x_{iE}} - \boldsymbol{x_{jE}})^2 + (\boldsymbol{x_{iS}} - \boldsymbol{x_{jS}})^2} \qquad \textbf{Eq 4.10}$$

where *N*, *E* and *S* are the spatial parameters (geographic coordinates and slope) of two points x_i and x_j standardized using converted z-scores. The clustering is shown in **Figure 4-4**, and described as follows:

- Group 1 = southwest, with gentle to rolling terrain
- Group 2 = east, with generally rolling to hilly topography
- Group 3 = northwest, with generally flat topography, near to the urban center

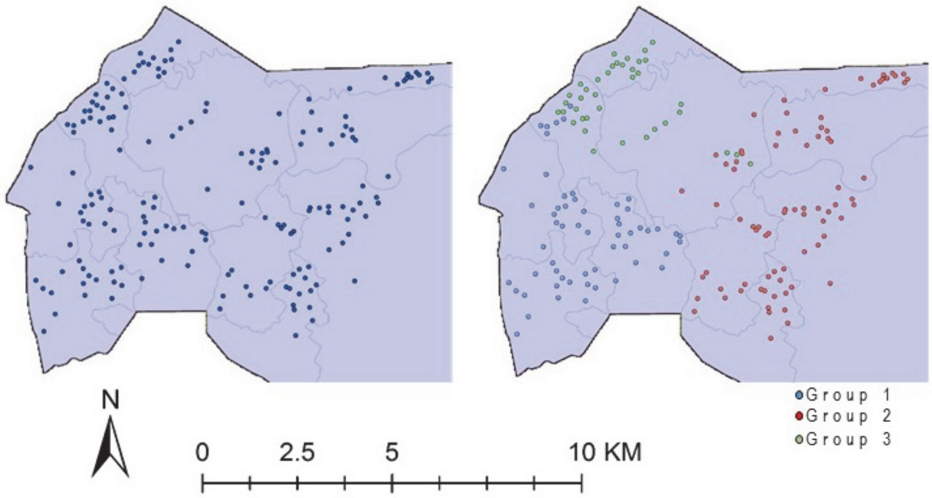


Figure 4-4. Map showing the resulting spatial grouping using K-nearest neighbor with spatial constraints

Correlation analysis using the sub-sectors generated from cluster analysis showed how some parameters influence WTP differently based on location. The summary of bivariate correlation statistics is presented in **Table 4-7.** Farm ecosystem and ownership types were found to have significant association with WTP for areas with more uneven topography, but not for the flatter region (Grp 1) where more farmlands are irrigated and leased. Other parameters were found to have significant association for particular regions which include: terrain (Grp 3), income level (Grp 2), nutrient loss (Grp 2), landslides (Grp 1) and acidification (Grp 3). This suggests that further analysis with regards the effects of topography and geographical location on stated value (WTP) should be further investigated.

4.4.6. **Tobit Model**

For the final econometric analysis, the Tobit model was used to censor extreme values in the dataset. Multiple linear regression was calculated to assess WTP using the various socio-demographic and soil threats as explanatory variables. Using stepwise regression, seven determinants (with $p<0.05$) were included in the model estimation, which was able to explain 68% of the variance. A significant regression equation was found ($F(7,166)=20.940$, $p<0.001$) with an R^2 of 0.469. The summary of model estimates is provided in **Table 4.8**.

Two demographic factors were found to be significant determinants in the regression model: income (β=0.25) and household size (β=0.12). Both were positively correlated with WTP, meaning that higher income earners and those from larger household sizes were more likely to give higher WTP values. The type of farm ecosystem (β=-0.19) was also found to be a significant indicator, with farmers from irrigated lands providing higher WTP values. The number of years spent in farming was also found to have a positive correlation with WTP. Those who have been involved in agriculture for much longer periods tend to have higher WTP values.

Some soil-related threats were also found to be significant determinants in the regression model: erosion (β=0.36), landslides (β=-0.24), and nutrient soil loss (β=-0.16). Vulnerability to soil erosion was shown to be a robust indicator of WTP, with those more vulnerable to higher rates of erosion providing higher WTP values. Likewise, similar to the findings of the correlation analysis, farmers who perceived landslides and nutrient loss as major concerns were shown to have higher willingness for soil conservation.

Table 4-7. Summary of correlation for selected determinants to WTP by subgroups

	Group 1: SW	Group 2: E	Group 3: NW
Parameters			
Size of Farm	0.197	0.119	0.127
% of Land for Farm	-0.049	-0.004	0.097
Farm Ecosystem	-.419***	-0.252*	-0.051
Type of Crop	0.092	-0.032	0.151
Terrain	0.016	-0.078	-0.345*
Ownership	0.304*	0.429***	-0.057
Income	0.125	0.297**	0.239
% Income from Farm	0.097	0.121	-0.004
Nutrient Loss	-0.132	-0.242*	-0.198
Landslide	-0.288*	-0.124	-0.311
Compaction	-0.175	0.192	0.263
Acidification	-0.059	-0.013	-0.333*
Waterlogging	0.045	-0.197	-0.119
Erosion	0.449***	0.587***	0.516***
Mean WTP			
Computed In PhP (P)	P 212.11	202.72	139.74
Converted in Dollars ($)+	$ 4.74	4.53	3.12

**** for p < 0.005; ** for p < 0.01; * for p < 0.05*

+ PhP = Philippine Peso (1 PhP = 0.2233 USD, March 31, 2014)

Table 4-8. Summary of model coefficients using MLR

	B (Std Err)	Std β		95% CI (LB, UB)	t-value
(Constant)	164.15				
	(90.27)			(-14.08, 342.37)	1.82
Erosion	59.15				
	(9.54)	0.36	***	(40.31, 77.99)	6.20
Income	47.34				
	(11.54)	0.25	***	(24.55, 70.13)	4.10
Landslide Threat	-107.00				
	(25.97)	-0.24	***	(-158.28, -55.73)	-4.12
Period as Farmer	44.82				
	(10.63)	0.24	***	(23.84, 65.80)	4.22
Farm Ecosystem	-84.71				
	(26.23)	-0.19	***	(-136.50, -32.91)	-3.23
Nutrient Loss Threat	-64.68				
	(23.89)	-0.16	**	(-111.85, -17.51)	-2.71
Household Size	25.29				
	(12.61)	0.12	*	(0.39, 50.18)	2.01

*R = 0.685; R2 = 0.469; F = 20.940****

4.5. Conclusion

This study contextualized soil value using an open-ended contingent valuation approach that elicited the stakeholders' willingness-to-pay for conservation. Its primary goal was to determine how socio-demographic attributes and perceived soil threats could influence WTP values. The WTP estimate was used as a proxy indicator for stated value, more specifically the explicit conservation value. The valuation framework used here is the cost-based assessment framework, which is most applicable when the valuation focuses on estimating the cost of degradation or the cost of conservation. This is particularly useful in establishing a reference (baseline) value for future studies using different valuation techniques.

The results of the Contingent Valuation show that the respondents are willing to pay for soil conservation. The estimated mean WTP from the Tobit Model was P192.04, with only 4% of the respondents giving zero value as response. Income levels, household size, farm ecosystem and farming experience were shown to be significant socio-demographic and farm-related indicators of WTP. Similarly, modeled and perceived soil threats such as erosion, landslides, and nutrient loss were shown to significantly influence the stakeholders' willingness for conservation. This information could be beneficial in planning for future conservation projects, in environmental management and policymaking. Clustering analysis showed the importance of analyzing WTP using additional spatial information, given how some parameters affect value differently based on location.

While this study provides a number of meaningful results, there were several uncertainties and methodological limitations in the estimation. As previously discussed, CVM's hypothetical nature can lead to an inflated upward hypothetical bias. Restrictive factors such as limited personal experience and incomplete information can cause disparity between the stakeholders' normative and solicited preference. In determining the effect of erosion vulnerability to WTP, there were no calibration and validation processes undertaken to increase the model accuracy due to limitations in available data and measurements. Future CVM studies specifically those assessing soil value can further investigate these areas of constraint, and propose new approaches to address them.

Chapter 5. Analyzing Soil Value and Price Determinants using PC and DC CVM

This chapter continues in exploring stakeholders' willingness to pay (WTP) for conservation and examines the use of other contingent valuation formats to limit some of the inherent constraints of the open-ended structure. This chapter assesses the effects of environmental awareness and geographical determinants on stakeholders' WTP. Given the limited research in soil valuation, the findings and methodology can be used towards a more comprehensive soil value characterization that integrates spatial analysis in econometric modeling.

5.1. Introduction

Environmental valuation of soil is essential in efficiently managing soil resources and provides a substantial argumentation towards conservation and sustainable use. The successful implementation of soil policy relies heavily on high participation among the various stakeholders. Active involvement of different user groups is crucial in increasing the likelihood of success in any management strategy (Mutekanga, Kessler et al. 2013). At the stakeholder level, valuation promotes sustainable use, explicitly linking economic decision making with the various direct and indirect benefits derived from nature (Salles 2011). A farmer will be more likely to adopt green agronomic practices when he learns of long-term costs and potential risks from unsustainable farming methods. On the governance and policy-making side, community leaders and elected officials will be better informed of economic consequences of potential changes and its impact on different sectors of the community (Jollands 2006).

Contingent valuation method (CVM) is one of the most dominant approaches used in valuing the environment. This type of stated preference (SP) technique takes a "whole good" approach that estimates economic value, measured as willingness to pay (WTP) or to accept (WTA) compensation, by directly questioning the affected population (Colombo, Calatrava-Requena et al. 2006). This main advantage of directly soliciting welfare estimates from stakeholders is also the method's biggest challenge. Many of CVM's critics often raise the issues of unreliability and methodological weaknesses of CVM in estimating value. Many economists are particularly biased towards revealed preference techniques, where economic values are inferred rather than explicitly stated, although this view is not necessarily shared by other social science fields (Carson, Flores et al. 2001). Despite some criticisms, CVM has been widely used by policymakers, environmental planner, economists, and research consultants for a variety of environmental applications and valuation research. Previous studies have shown that welfare estimates from CVM are consistent and reliable as long as the survey design and implementation are properly executed (Griffin, Briscoe et al. 1995). Aside from estimating WTP, most CVM studies are designed to identify value determinants, particularly the respondents' demographics, and how these attributes are able to influence value. Previous studies have shown the relationship of WTP with socio-economic indicators such as gender, age, household income, and education level (Wang, He et al. 2013, Khan, Brouwer et al. 2014).

While these demographic attributes have been shown to influence WTP, other value determinants need to be analyzed to further explain preference heterogeneity. Variables such environmental awareness, geographic location, level of environmental risk and proximity-to-amenities are some of the possible determining factors that could have significant influence on stated value. How these particular factors influence the WTP estimates will be examined in this chapter.

Analyzing environmental awareness is crucial to understanding the differences in knowledge levels among sectors on how they perceive environmental degradation and how they respond to their environment (Ziadat 2009). Perceptions of environmental use and degradation are fostered based on a socio□economic, cultural process and the level of technological development in the community (Atiqul Haq 2013). Past studies have shown how environmental awareness is determined not just by external forces but also by the individual's socio-cultural attributes. These include income level/social class, place of residence, age, gender, family size, education, and political ideology (Liere and Dunlap 1980, Ono and Maeda 2014). Another essential value determinant that can explain stakeholder heterogeneity is the geospatial aspect. Spatial dimensions have been shown to dramatically modify stakeholders' cognition and preference (Moreno-Sanchez, Maldonado et al. 2012, Andreopoulos, Damigos et al. 2015). Economic value and location-specific environmental attributes often demonstrate relationships of spatial dependency (Bateman, Day et al. 2006). However, geospatial analysis has had limited success in catching up with prevailing discussions in environmental economics (von Haaren and Albert 2011). For instance, many SP studies failed to assimilate the spatial effects in estimating value despite its potential influence on WTP aggregation (Schaafsma, Brouwer et al. 2012).

In recent decades, the increasing use of geographic information systems (GIS) in social sciences has made geospatial data to be crucial in resource economics (Hermann, Schleifer et al. 2011). A number of environmental valuation studies have demonstrated the benefits of incorporating spatial data and physical models in econometric studies (Bockstael 1996, Anselin 2001). Their results suggest that significant spatial and environmental attributes contribute to the formation of stakeholder cognition and preference, and can be used to explain some level of WTP heterogeneity. Frameworks that make spatial variability more explicit have been proposed, such as fusing spatial indicators in making empirical decisions and linking landscape functions with spatial policy data (Willemen, Verburg et al. 2008). With the rise of new spatial models with varying levels of flexibility and spatial demands, the role of spatial data and geo-information will become even more prominent in environmental valuation, planning, and management.

5.2. CVM Formats: Payment Card and Dichotomous Choice

Two of the most popular CVM systems are the **payment card** (PC) and **dichotomous choice** (DC) formats (Carson 2011). The payment card is a multiple bids format that allows the respondents to choose their WTP from a range of values. Generally, it has been shown to be more systematically related with explanatory variables (i.e., socio-demographic characteristics) and is consistent with the respondent's normative preferences (Cummings, Brookshire et al. 1986, Loomis 1990). Increasing the number of bids can theoretically increase efficiency by narrowing down the range at which the respondent's WTP falls. But this can result in additional analytical complexity for the respondents that may be counter-intuitive to the process. Determining the number of choice options and the value intervals between bids is therefore crucial in PC-CVM. The values can be defined using *a priori* assumptions which can then be calibrated using pilot-testing. In this study, the results of the open-ended CVM were used to generate the choice bids.

Similar to the econometric model used in other CVM studies, the welfare estimation for this PC and DC format is based on the utility variation model as proposed by Hanemann (1984):

$$\boldsymbol{y_i}^* = \boldsymbol{\beta x_i} + \boldsymbol{\lambda_i} \qquad \textbf{Eq 5-1}$$

where y_i^* is the individual's WTP variable, x_i is a vector of the individual's attributes, β is the coefficient for the attributes, and the λ_i is the error term with the mean equal to zero (Alberini 1995). In estimating the WTP values for the PC-CVM, the econometric model can be analyzed using ordinary least square regression by assuming a normally distributed random term (ε) and postulations for some covariates ($\beta\gamma$) and. The normally distributed WTP can then be calculated by:

$$\mathbf{WTP} = \boldsymbol{\mu}\mathbf{WTP} + \boldsymbol{\varepsilon} = \boldsymbol{\beta\gamma} + \boldsymbol{\varepsilon} \qquad \textbf{Eq 5-2}$$

For the dichotomous choice or referendum CVM, the respondent is given a scheme with an ascribed cost which the respondent will either accept or reject. Its main advantage is that the respondents are more likely to be familiar with this type of referendum-like mechanism (Evans 2008). However, the welfare estimation in the DC-CVM is relatively more complex. By correlating the respondents' characteristics with his choice preference, statistical analysis is used to calculate the likelihood that the respondent will agree to the proposed scheme (s) and its associated cost. Each response specifies whether a given price bid (c_i) is greater or less than the respondent's WTP (Cameron 1988). Simply put, the dichotomous response for the individual I_{is} is given:

$$\begin{aligned} &\boldsymbol{I_i = 0\ (response\ is\ no)\ if\ WTP_{is} < c_{is}} \\ &\boldsymbol{I_i = 1\ (response\ is\ yes)\ if\ WTP_{is} \geq c_{is}} \end{aligned} \qquad \textbf{Eq 5-3}$$

Assuming the individual (i) knows his valuation distribution, the probability that the individual would agree (yes = 1, no=0) with the offer given a price bid (c_i) is:

$$\mathbf{P_i = Prob\big(y_i^* \geq (c_i|x_i)\big) = 1 - F(c_i|x_i) + \lambda_i} \qquad \textbf{Eq 5-4}$$

where λ_i is the error term with a zero mean value, and a standard variance of δ^2 which is constant for individual respondent but fluctuates across the total respondents. The dependent variable P_i takes a value between 0 to 1 which can be considered as a continuous variable. Equating the specific function form $F(\cdot)$ into a standard normal distribution probability density ($\phi(\cdot)$) with a standard variance σ_i, the model becomes:

$$\mathbf{P_i = 1 - \phi\left(\frac{c_i - x_i\beta}{\sigma_i}\right) + \lambda_i} \qquad \textbf{Eq 5-5}$$

The main purpose of evaluating the equation is to analyze the mean value of y_i for each individual respondent as a function of personal information. The WTP distribution can then be calculated using a number of approaches.

One such approach is the Turnbull's estimator, which is a fully-nonparametric technique that is used to find the expected lower bound of the willingness to pay. This simple approximation of mean WTP uses a distribution-free estimator that depends on asymptotic properties. The Turnbull estimator uses the probability of acceptance for each price bid that mimics a survival function. The WTP estimate is calculated by adding the products of the lower bound bid and the change in density (Hamed, Madani et al. 2016).

Another approach is based on Hanemann, Loomis et al. (1991) that considers the mean WTP in the interval from zero to the maximum price bid. Using a parametric modeling approach, the Spike model (Kristrom 1997) which considering respondents with zero WTP could be used:

$$\boldsymbol{P_i(1) = \Lambda\big(\Delta V(A)\big) = \begin{cases} [1 + exp(\alpha)]^{-1} & A = 0 \\ [1 + exp(\alpha - \beta A)]^{-1} & A > 0 \end{cases}} \qquad \textbf{Eq 5-6}$$

where ΔV(A) is utility difference function, α and β are variables that could be approximated using maximum likelihood method, and A denotes the price bids. The Spike model emerges to be particularly applicable when there is considerable portion of the population choosing zero price bids (Ramajo-Hernandez and del Saz-Salazar 2012). The WTP approximation for the spike model is given by the equation (Kristrom 1997)

$$\boldsymbol{E(WTP) = -\frac{1}{\beta} ln(1 + e^{\alpha})} \qquad \textbf{Eq 5-7}$$

Both the parametric and non-parametric welfare estimate approximations were computed and analyzed in this study.

5.2.1. **Survey Design**

The contingent valuation was conducted using a door-to-door questionnaire survey from January to March 2015, with support from the municipal government of Norzagaray and its agriculture office. Before the survey, focus group discussions were organized with the various communities, including the local and provincial government representatives and Barangay officials (community leaders), focusing on stakeholder's farming techniques, environmental concerns, and local understanding on soil issues including use, management, and conservation. A draft questionnaire (presented in **Appendix G**) and survey plan were developed and finalized after a round of pre-testing with personnel from the local agriculture's office. It was then pilot tested with a small group of local farmers to ensure comprehensibility of questions and to estimate time requirements.

A stratified random sampling approach was developed in choosing the 300 heads of agricultural households as sample population in this study. The respondents were informed that a valuation study was being conducted in support of soil conservation measures that will supplement current land management projects initiated by the local government. The questionnaire utilized a payment card (PC) CVM format soliciting WTP on a voluntary payment, and a dichotomous choice (DC) CVM format for a mandatory fee.

The payment vehicle was in the form of a fee to be collected for financing projects and initiatives aimed at minimizing degradation in the farmlands. The fund was to be used in supplementing government efforts to reduce erosion rates at the farm level through public financed soil conservation measures especially targeting poorer agricultural households. In the PC format, a proposed community fund was to be set up on a voluntary capacity, and the respondents were asked to choose for their WTP from among nine price bids (₱0.00 – ₱200). For the referendum CVM, the respondents were asked whether they were amenable to the imposition of a mandatory fee, ranging from ₱50, ₱100, ₱150, and ₱200. The respondents were then inquired for their willingness on a follow-up bid: if the respondent answered yes on the initial bid, the succeeding bid was raised by ₱25; if no, the value was decreased by ₱25. The section concluded with a self-evaluation test measuring the individual's propensity for farm-based soil management. The responses were converted into a score from 1 to 5 which was then averaged and used as the agricultural sustainability consciousness index (ASCI) variable, reflecting the respondents' behavior and perception towards soil conservation.

5.2.2. **Spatial Analysis**

The respondents' geographic coordinates were determined mainly with the use of handheld GPS, which were then entered into the geodatabase. A different set of spatial analyses were then used for the two CVM formats in analyzing the effect of the respondent's spatial location to WTP. For the payment card-derived WTP, soil erodibility and landslide hazard map were used to assess WTP heterogeneity. The soil erodibility factor (K) was generated using the geologic/soil map of the Angat Watershed, with additional soil texture data from the Bureau of Soil and Water Management. Using ArcGIS 10.5, the erodibility map was generated using the equation (Foster, Mccool et al. 1981):

$$100K = 0.1313[(2.1m^{1.14} \times 10^{-4}(12 - a)) + (3.25(b - 2)) + (2.5(c - 3))] \quad \textbf{Eq 5-8}$$

where m is (silt (%) + very fine sand (%)) (100-clay (%)); a is organic matter (%); b = structure code used soil classification; c = soil permeability class. The landslide susceptibility map used in this study was provided by the Mines and Geosciences Bureau through Bulacan's Provincial Disaster Risk Reduction and Management Council.

For the dichotomous choice format, the influence of topographic effects and proximity-to-amenity were evaluated. Topography was characterized by the elevation map and the slope map, which were generated from Advanced Spaceborne Thermal Emission and Reflection Radiometer-Global Digital Elevation Model (ASTER-GDEM). The ASTER-GDEM images were downloaded, mosaicked and processed using ENVI 5.3 software. For elevation, three dichotomous groupings were utilized: (a) less than 50 meters, (b) less than 100 meters, and (c) less than 150 meters. Similarly for slope, three dichotomous categorizations were set: (a) less than three-degree slope, (b) less than eight-degree slope, and (c) less than 15-degree slope. For proximal analyses, the distance to water tributaries, and the distance to forest reserves were used. In delineating the proximal regions, a landcover map was generated from Landsat 8 images and processed using ArcGIS 10.5. Three river proximal regions were generated: (a) within 500 meters, (b) within one-kilometer, and (c) within 1.5 kilometers. Likewise, three forest proximal zones were also used: (a) within two kilometers, (b) within four kilometers, and (c) within six kilometers.

Table 5-1. Summary of socioeconomic composition of the respondents

Parameter		Value
No. of Respondents		276
Gender	Male – Female	85.14% – 14.86%
Education	w/o High school Diploma	48.55%
	finished High school	29.35%
	Technical school	12.68%
	College	9.42%
Annual Income	< P40,000	47.83%
	P 40,000 - P 69,999	13.04%
	P 70,000 - P 99,999	18.84%
	> P100,000	20.29%
Age	Mean	54.36
	Lowest – Highest	22 – 81
Household Size	Mean	4.87
	Lowest – Highest	2 – 8
Type of Land Ownership	Owned through Rights	11.96%
	Owned through Purchase	21.38%
	Owned through Inheritance	19.93%
	Rented / Leased	16.30%
	Farm worker	30.43%
Farm size (in Has)	Mean	1.41
	Lowest – Highest	0.25 – 6.70

5.3. Results and Discussion

From the 300 agricultural families chosen to participate in the survey, 24 responses were excluded from the analysis due to incomplete socio-economic data, missing spatial information of farms and multiple marked-responses. The socio-demographic breakdown is shown in **Table 5-1**. The average age of the respondents was 54, and the average household size was five. There was an overwhelming number of male respondents due to sample population being heads of household. There was good spread in the other demographic groupings, with some apparent proclivity to specific categories (i.e. without HS diploma and <₱40,000 earners), fairly proportional to the town's demographic composition.

5.3.1. Environmental Awareness

A self-evaluation test measuring the individual's propensity for farm-based soil management was included at the end of the questionnaire. The responses were converted into a scoring system from 1 to 5 which was then aggregated and used as the agricultural sustainability consciousness index (ASCI) variable. The ASCI was used to score the individual's environmental awareness, reflecting the farmer's behavior and perception towards soil conservation. **Figure 5-1** presents the questions used in assessing the ASCI and the summary of results.

About 93% of the respondents agreed that soil protection was an essential component in their farm operations, while 72% agreed that the local government has the responsibility to enforce soil conservation measures for the community in general. Majority of the respondents said that they invest in farm-based soil conservation measures (81%) and that they continually seek additional training to learn more about conservation methods (68%). Post-survey discussions revealed that the additional training and technical support for soil protection had been provided mainly by the Municipal Agriculture Office. Almost three in every five of the respondents (59%) either agreed or strongly agreed that regulations and penalties for non-compliance of soil conservation measures are justifiable, while only one in every two respondents (52%) agreed on the imposition of additional fees towards soil conservation.

Table 5-2. Pearson correlation coefficients for WTP and one-way ANOVA for discrete explanatory variable

			ANOVA		
	Corr Coefficient	Sig.	F	Sig	
Gender	0.056	0.353			
Age	0.073	0.229			
Farm size	0.109	0.069			
Household size	0.034	0.569			
ASCI	0.152	0.012			*
Education	0.225	0.000	4.627	0.000	***
Income	0.332	0.000	4.888	0.000	***
Ownership	0.306	0.000	5.357	0.000	***

**** Significant at 0.001 level; ** significant at 0.01 level; and * significant at 0.05 leve*

5.3.2. Payment Card

The mean WTP per household was estimated to be P79.98 ($1.80) per year, with 25 respondents (9%) expressing zero bids. Previous studies have argued on the advantages of censoring protest bids from the econometric analysis (Lindsey 1992, Whittington 1998) But others have cautioned against excluding zero bids as it may lead to unjustifiable bias towards increased welfare estimates (Jorgensen, Syme et al. 1999, Madureira, Nunes et al. 2011, Grammatikopoulou and Olsen 2013). After excluding zero bids, the mean WTP was P87.95 ($1.98). About 77% of the respondents selected price bid of P100 or less, with the bulk of respondents choosing P50 (37%) as WTP.

The Pearson product-moment correlation coefficients were computed to determine the relationship between the willingness-to-pay for soil conservation and the different respondent attributes. **Table 5-2** presents the summary of results. There was significant moderate correlation between WTP and education ($r=0.225$, $p<0.001$), WTP and income ($r = 0.332$, $p<0.001$), and WTP and ownership ($r=0.306$, $p<0.001$). Environmental awareness measured using the ASCI was also found to be positively correlated with WTP, with those who consider soil conservation as essential in their decision making more likely to give higher WTP values. Additional analysis of variance (ANOVA) was performed between WTP and the three discrete explanatory variables, generating similar results of significant relationships.

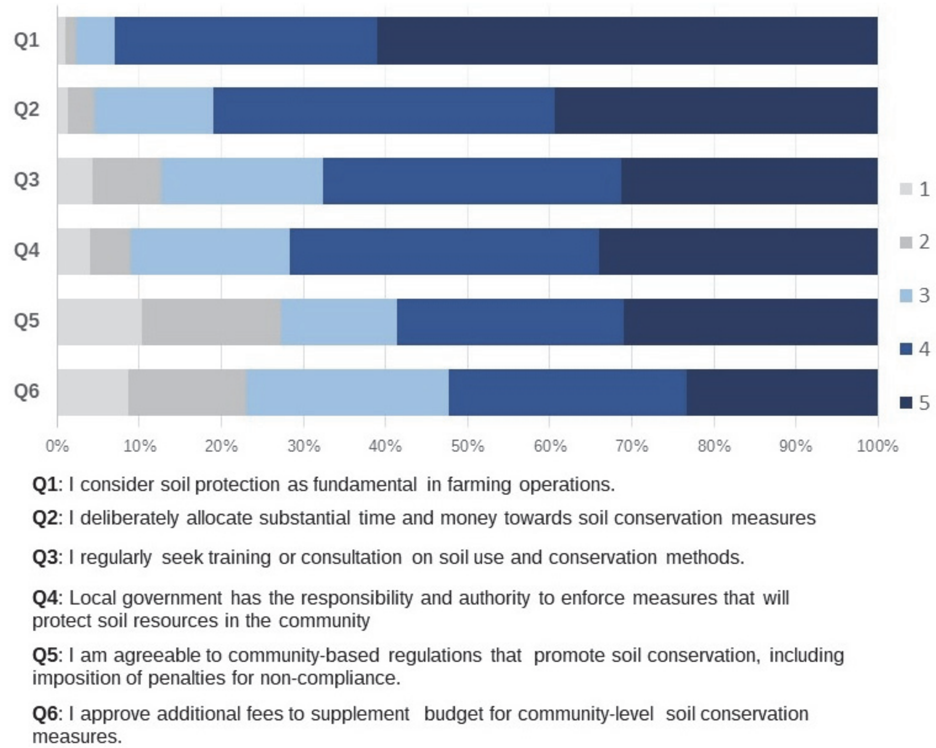


Figure 5-1. Chart showing respondents' environmental awareness index

Table 5-3. Regression model results of the PC-CVM

	Model A-I				Model A-II			
	Coeff			t-value	Coeff			t-value
	β	Std Err	Std β		β	Std Err	Std β	
(Constant)	-21.976	25.851		-0.850	-21.362	16.850		-1.268
Education	8.870	3.410	0.157	2.601**	7.320	3.326	0.130	2.201*
Income	8.647	2.932	0.189	2.949**	9.601	2.907	0.210	3.302***
Owner	7.787	2.364	0.198	3.293***	8.142	2.356	0.207	3.456***
ASCI	11.183	3.692	0.165	3.029**	11.852	3.700	0.175	3.203**
Gender	-18.959	8.531	-0.122	-1.022				
Age	0.328	0.257	0.073	1.278				
Farm size	3.791	3.445	0.060	1.100				
Household	0.402	2.075	0.110	0.194				
R^2	0.214					0.191		
F	9.060					15.959		

**** Significant at 0.001 level; ** significant at 0.01 level; and * significant at 0.05 level*

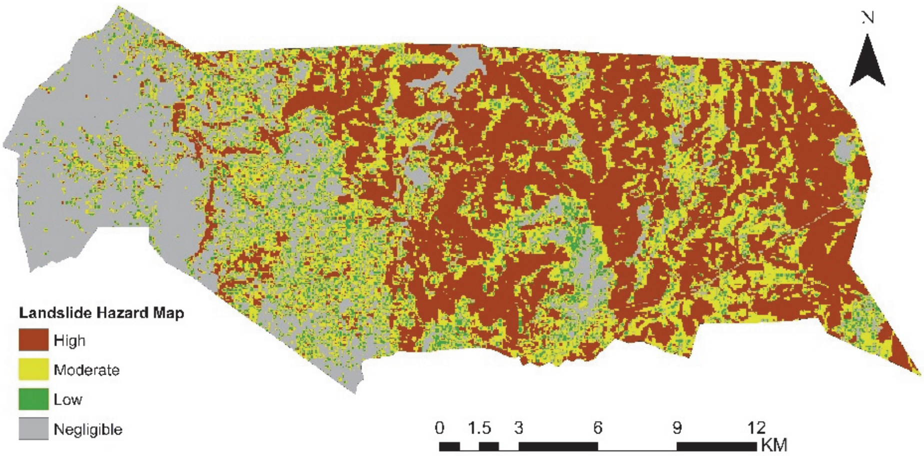


Figure 5-2. Norzagaray's landslide susceptibility map
(adapted from the Bulacan Provincial Disaster Risk Reduction and Management Council)

Modeling the relationship of WTP with all the explanatory variables, an ordinary least squares regression was calculated and the summary of findings is presented in Table 3. The results indicate there was a collective significant effect between WTP and the list of independent variables ($F = 9.06$, $p<0.000$, $R^2=0.21$). Individual predictors were further examined which showed four of the eight independent variables were found to have significant influence on

WTP: type of land ownership (t=3.29, p<0.001), income (t=2.95, p<0.01), ASCI score (t=3.03, p<0.01), and education (t=2.60, p<0.01). Minimizing the model with only significant regressors, a stepwise linear regression model was constructed and the summary of results is presented in **Table 5-3** (Model A-II). The resulting model (F=15.96, R^2=0.19) included four significant explanatory variables, similar to the results of the OLS model. Assessing the relationship between WTP and the environmental consciousness score (ASCI), there was a positive albeit smaller correlation coefficient (r=0.152, p<0.05).

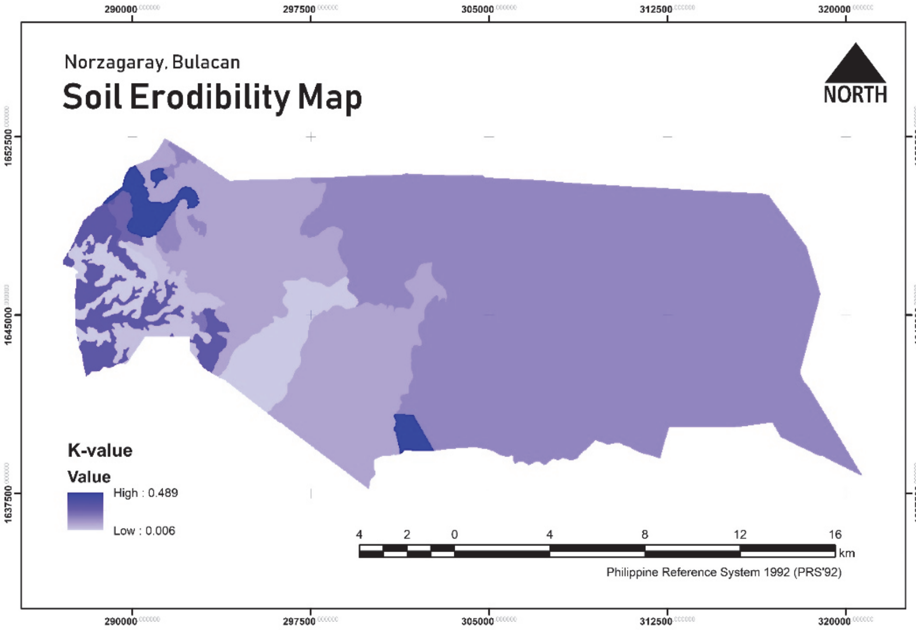


Figure 5-3. Soil erodibility map of Norzagaray

Previous research have suggested that significant correlation between income and willingness-to-pay value is highly indicative that the respondents have taken the WTP question seriously (Zhao, Liu et al. 2013). Aside from income, WTP was correlated with the type of land ownership suggesting that land tenure play a significant influence on respondents' WTP. Similar to previous findings (Gebremedhin and Swinton 2003, Fraser 2004, Sklenicka, Molnarova et al. 2015), landowners were more likely to provide higher WTP and invest more on soil conservation compared to tenants and farm-workers. Education and environmental consciousness score were also found to have significant positive correlation, with those having more formal education and scoring higher in ASCI correspondingly providing higher WTP responses.

After incorporating the spatial coordinates of the respondents into the geodatabase, the landslide classification index (**Figure 5-2**) and soil erodibility values (**Figure 5-3**) were analyzed to check whether these variables are related to WTP. To test differences in WTP values between landslide classes, a one-way analysis of variance was made. The summary of results is shown in **Table 5-4**. There was no statistically significant difference between group means as determined by one-way ANOVA ($F(3,272) = 1.248$, $p = 0.29$). Likewise, the Pearson coefficient was computed between stakeholders' WTP values and soil erodibility. The results indicate no significant correlation ($r=0.109$, $p=0.07$). This may indicate erodibility values do not necessarily influence a respondent's WTP, or that that erodibility values particularly in this region and given the map scale is not a good measure for actual erosion risks.

Table 5-4. ANOVA Results for WTP and Landslide Hazard Map Index

		N	WTP Mean	Std. Error	95% Conf. Int. Lower	Upper
1	Negligible	161	75.78	4.129	67.62	83.93
2	Low Risk	26	75.96	10.785	53.75	98.17
3	Moderate Risk	59	86.02	7.699	70.61	101.43
4	High Risk	30	94.17	11.477	70.69	117.64
Fixed Effects				3.330	73.43	86.54
Random Effects				4.090	66.96	93.00

5.3.3. Dichotomous Choice

The summary detailing the acceptance rates at the various price bids is presented in **Table 5-5**. The Turnbull WTP value was then estimated by summing the products of the lower bound value and the density change and was calculated to be P99.47 ($2.24). The median value (50th percentile) was within the price bid range of the P75-P100.

In analyzing the DC-WTP results, a logit regression model that included the stakeholders' attributes was generated. This model was to determine which factors influenced the respondent's decision-making in valuing for soil conservation. The results of the double-bounded dichotomous choice logit model are shown in **Table 5-6**. The generated logit model was able to predict 76% of expected probabilities. The results of the model show that price bid and the income level are both significant in affecting the probability of the respondent's willingness to pay. The income coefficient being a positive value indicated that high-income earners had a higher probability accepting the bid proposal. The negative price coefficient implied that the higher the proposed fee, the less likely it would be accepted. These findings were in agreement with previous findings of related WTP studies (Brugnaro 2010).

Table 5-5. WTP responses and acceptance rate for the dichotomous choice CV

WTP value	₱25	₱50	₱75	₱100	₱125	₱150	₱175	₱200	₱225
Total	19	70	90	71	88	67	69	68	10
Yes (Accepted)	17	51	55	32	21	11	12	10	1
No (Rejected)	2	19	35	39	67	56	57	58	9
Accept Rate	89.47	72.86	61.11	45.07	23.86	16.42	17.39	14.71	10

The elevation (a), slope (b), water buffer zone (c) and forest buffer zone (d) maps are presented in **Figure 5-4**. In interpreting the spatial effects to the stated value, the mean WTP and analysis of variance were analyzed using a logit model, and the summary of results is presented in **Table 5-7**. Attribute variations were tested at 5% significance (p=0.05), 1% (p=0.01), and 0.10% (p=0.001) levels and were considered statistically significant if the p-value was less than 0.05. Results from the logit model revealed that topographic effects did not significantly influence stakeholders' responses. However, the results of the logit model showed that proximity to amenities had some significant effect on WTP values. Those who lived within one kilometer from the river system had significantly higher mean WTP (₱92.50) compared to those living outside (₱71.69). Similarly, those living close to the protected forest reserves substantially had higher WTP values. The two-kilometer and four-kilometer those living inside these zones having higher mean WTP than those living outside.

Table 5-6. Parameter estimates of the double bounded logit model for the DC-CVM

Variable	B	S.E.	Wald	
Constant	0.649	0.884	0.539	
Price Bid (WTP)	-0.023	0.002	102.226	***
Income	0.334	0.099	11.522	***
Gender	-0.062	0.307	0.040	
Age	-0.007	0.009	0.631	
Household Size	0.001	0.070	0.000	
Education	0.121	0.114	1.116	
Ownership	0.128	0.081	2.523	
ASCI	0.177	0.129	1.880	

**** Significant at 0.001 level; ** significant at 0.01 level; and * significant at 0.05 level*

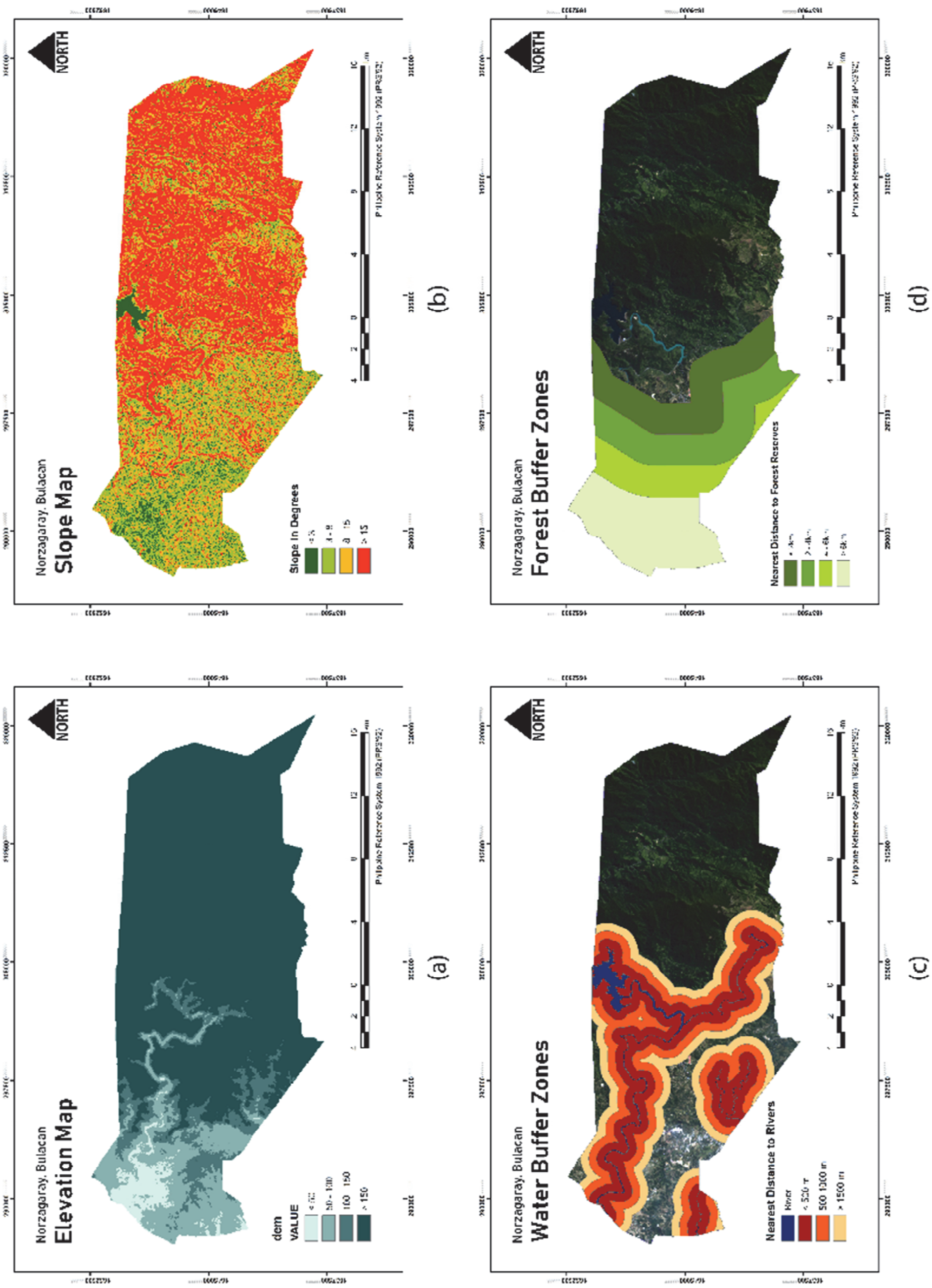


Figure 5-4. Spatial maps of Norzagaray Bulacan: (a) elevation map; (b) slope map; (c) water buffer zone map; and (d) forest buffer zone map

Table 5-7. Summary of results of logit model for spatial variables

		Mean WTP (PhP)		Logit Model		
	N (%)	within zone	outside zone	Wald - WTP	Exp (B)	p-value
Water zones						
(a) within 500m	23.19%	90.23	76.89	2.788	1.004	0.095
(b) within 1km	39.86%	92.50	71.69	8.957	1.007	0.003
(c) within 1.5km	61.59%	83.68	74.06	1.932	1.003	0.165
Forest zones						
(a) within 2km	16.67%	98.91	76.20	6.177	1.007	0.013
(b) within 4km	39.49%	88.53	74.40	4.182	1.005	0.041
(c) within 6km	53.26%	84.01	75.39	1.637	1.003	0.201
Elevation Classes						
(a) <50 meters	22.10%	88.11	77.67	1.656	1.003	0.198
(b) <100 meters	61.23%	78.25	82.71	0.418	0.999	0.518
(c) <150 meters	80.80%	80.16	79.25	0.011	1.000	0.915
Slope Category						
(a) <3degree slope	14.49%	94.38	77.54	3.067	1.005	0.080
(b) <8degree slope	58.33%	83.07	75.65	1.185	1.002	0.276
(c) <15degree slope	88.41%	78.89	88.28	0.798	0.997	0.372

5.4. Conclusion and Policy Implications

There is a general recognition that the success and sustainability of environmental conservation depend primarily on the efficient use of resources and on the concerted effort of the various stakeholders to follow the strategies. The valuation of soil and other environmental public goods have been shown to provide an effective means of understanding the environmental contributions to human well-being. By highlighting the various environmental functions and services it provides, soil protection and its sustainable use are brought to the decision-making table alongside other economic and policy matters. However, approaches in soil valuation still remain tenuous given a variety of factors, including the absence of a market to serve as a pricing mechanism, a stakeholder base with heterogeneous preferences and cognition, and its intrinsic uniqueness providing both private benefits and communal amenities. The contribution of this study is to move the needle forward in soil valuation science by providing methodological and analysis improvements.

The results of the multi-stage modeling approach integrating PC and DC CVM indicate a positive attitude towards soil conservation, which was estimated using the payment card format at P79.98 ($1.80) per household, or P99.47 ($2.24) calculated using the dichotomous choice format. When the zero bids were excluded in the PC-CVM calculations, the mean WTP increased to ₱87.95 ($1.98). For context, the region's daily minimum wage for farm workers was ₱319 at the time of the survey. This means that the average WTP roughly translates to one-fourth of a farm worker's daily wage. Comparative analysis of the two methods also showed high similarities of results, both with the WTP calculations and the significant stakeholder attributes affecting stated value. The findings of the study corroborated previous studies that found respondents from higher income and education brackets were more likely to choose higher price bids.

Other factors were also found to significantly influence choice preference, including the type of land ownership and the degree of personal environmental consciousness. Land tenure was shown to have significant influence on WTP, with landowners more willing to invest for conservation than land tenants. People who have greater attachment and stake with the health of the land will be more compelled to protect it. This suggests that land right is not just a social issue but also an environmental matter. Likewise, those with higher environmental and ecological consciousness were more likely to spend for soil conservation and sustainable use. This highlights the need for education to be integrated in government policies, which will underscore the importance of soil conservation and influence stakeholder perception.

The primary objective of this paper was to assess how environmental consciousness, demographic and spatial factors influence the stakeholders' willingness to pay for soil conservation. The overall findings suggest that while WTP estimates were highly dependent on the respondent's income level, other socio-demographic and spatially dependent variables can influence preference, such as education, type of land ownership, and environmental consciousness of the individual. Spatial analysis also revealed proximity to river and forest amenities had a significant positive influence on WTP estimates. Several observations were noted in the planning, implementation, and analysis of results in this study. They are summarized as follows:

- One of the main challenges in stated preference studies is ensuring that the respondents provide truthful responses which reflect their normative preferences. Stakeholder engagement, particularly with farmers and farm workers, is a complex and tedious process. Aside from persuading chosen respondents to participate in the project, building trust is one of the primary factors affecting the accuracy of results. Gaining the support of the local government, community leaders, and the farmers' organization was crucial in carrying out this research, from communicating with the respondents to providing assistance and security when required.
- Stakeholder engagement should be further promoted in environmental valuation. In this study, the participation of stakeholders was not limited to the implementation of the survey, but throughout the development

process which includes questionnaire improvement and post-evaluation discussion. Moreover, collaborative research with stakeholders promotes participation that is needed when developing and implementing policies.

- The questions to measure the environmental awareness was explicitly designed for farmers and serves only as a preliminary design. It would need to be further improved to elicit a more descriptive and comprehensive picture of the respondent’s awareness level.
- Only a limited number of spatial determinants were included in this study. Future studies can explore other spatial parameters that could influence WTP values, including pedometric attributes.

Chapter 6. Exploring Indirect Use Value through Discrete Choice Experiment

This chapter analyzes stakeholder WTP heterogeneity for soil's indirect use-value by assessing the various socio-demographic and spatial determinants influencing preference variation. This chapter provides a methodology for the use of discrete choice experiment (DCE) in soil valuation, which is a non-market based multi-attribute valuation approach that can estimate the value of attribute change of direct and indirect utility. By attributing the various pecuniary contributions of EPG towards human-well-being, the results would be crucial in developing policies that would integrate the value of indirect services along with the more tangible soil amenities and can contribute to the growing literature on environmental economics and soil science in general, specifically on the under-represented economic valuation of soil conservation.

6.1. Introduction

While considerable developments have been advanced in the field of environmental economics over the years, much work is still required to deal with understanding the complex nature of environmental worth. In the previous chapters, soil's economic value has been measured using contingent valuation methods (CVM) which estimates value through direct solicitation. CVM has been the most dominant approach in environmental valuation because of its flexibility that allows the valuation of a broad range of non-market goods and enables the assessment of total value including the passive use value (Carson, Flores et al. 2001). One of its primary limitations though is its inability to sequester the contributions of specific parameters and amenities of an economic good. This results in value ambiguity which questions whether respondents are able to comprehend and appropriately value the full contributions of the environment, particularly its indirect use values. Ecological functions that indirectly provide amenities and services are oftentimes vague and intangible. For soils, in particular, many pedological functions may have undefined value because of their non-consumptive and non-excludable nature. In order to better rationalize the economic value of soil's indirect functions, alternative econometric approaches that are able to discriminate between individual amenities and evaluate each separately need to be used. One such approach is the discrete choice experiment.

6.1.1. Theoretical Design of DCE

The use of discrete choice experiments (DCE) is particularly suited to explore the variability in stakeholder decision making particularly to distinguish preference for individual amenities or resource attribute. DCE is a survey based stated preference technique that asks the respondents to choose from different choice sets containing mutually exclusive hypothetical alternatives. Considered as an advanced stated preference approach, DCE has been used in a variety of environmental valuation applications, including in biodiversity enhancement (Bartczak and Meyerhoff 2013, Zander, Signorello et al. 2013, Yao, Scarpa et al. 2014), environmental restoration (Bienabe and Hearne 2006, Alvarez-Farizo, Gil et al. 2009, de Rezende, Kahn et al. 2015, Lienhoop and Brouwer 2015), health-risk aversion (Veronesi, Chawla et al. 2014, Vidogbena, Adegbidi et al. 2015), climate change adaptation (Nguyen, Robinson et al. 2013, Andreopoulos, Damigos et al. 2015).

The standard choice modeling technique assumes that the respondents' utility defined over a clearly defined array of attributes, including price (Colombo, Christie et al. 2013). Unlike other non-market valuation approaches such as contingent valuation or travel cost method, DCE provides the estimation of value change in a number of attributes, as well as the compensating surplus measures of multiple changes in attribute levels (Viteri Mejía and Brandt 2015).

And with the inclusion of cost attribute, the marginal utility can be translated into estimates of the willingness to pay (or accept) attributed to the change in an environmental attribute.

Two main principles provide the foundations for choice experiments in linking choice behavior being assessed in the survey and the respondents' preferences over a variety of attributes: Lancaster's Utility theory (Lancaster 1966) and the Random Utility Theory (McFadden 1974). Lancaster's Utility Theory states that the value consumers attribute to a particular good is based on the different attributes of products or services from which consumers derive utility, rather than directly from the good. Random Utility Theory suggests that for a respondent n selecting the option j from a choice set i = 1, …, J in a situation t, the individual indirect utility (U_{njt}) can be decomposed by the utility model:

$$\boldsymbol{U_{njt} = \beta_n{}' x_{njt} + \varepsilon_{njt}} \qquad \textbf{Eq 6-1}$$

where $\beta_n{}'$ is the coefficient vector, and x_{njt} is the vector of attribute levels of option j. The value of the stochastic component ε_{njt} reflects the utility function describing the difference between a person's actual utility and the measurable aspect of the utility. This utility component is assumed to follow a type 1 extreme value distribution, so that the probability P_{nit} for the respondent choosing one option over other alternatives ($j \neq i$) is

$$\boldsymbol{p_{nit} = \frac{exp(\beta_n{}' x_{nit})}{\sum_{j=1}^{J} exp(\beta_n{}' x_{njt})}} \qquad \textbf{Eq 6-2}$$

Some econometric approaches can be used to analyze the probability equation. When the random component ε is assumed to be independently and identically distributed (IID) with an extreme-value distribution, the choice model can be estimated using the McFadden (1974) conditional logit (CL) model. The CL model assumes the scale parameter to be constant, corresponding to the respondents' having similar choice behavior. The implicit assumption is that the respondents have a homogenous taste for the attributes as presented in the choice experiment.

Another approach that has been used in discrete modeling studies is the random parameter logit (RPL) method (Train 1998, Colombo, Christie et al. 2013). Unlike the standard CL models, preference heterogeneity is accounted for in RPL by allowing the parameter vector to vary among individuals with values dependent on the underlying distribution that captures the respondents' taste (Veronesi, Chawla et al. 2014). It relaxes the independence of irrelevant alternatives (IIA), which can represent any substitution pattern and can explicitly account for unobserved heterogeneity (Gelo and Koch 2012). Both the CL and RPL models produce estimates of the coefficient vector β, and can be interpreted as the average utility weights of the attributes from the choice tasks (Borger, Hattam et al. 2014). Although the assumptions for β are defined differently, both models produce them using maximum likelihood functions. The marginal WTP for the attribute can then be calculated using the ratio of the coefficients of the attribute and the cost bid attribute:

$$\boldsymbol{WTP_k} = \frac{\boldsymbol{\beta_k}}{-\boldsymbol{\beta_{cost}}} \qquad \textbf{Eq 6-3}$$

Where β_k and β_{cost} are the coefficients of the k-attribute and cost respectively. For the random parameter logit,, the β_k represents the mean of the distribution of the coefficient for each attribute, and the mean marginal WTP value can be calculated by taking the average over the sample distribution of WTP coefficients.

6.1.2. **Spatial Parameters**

Spatial modeling and mapping of economic welfare from ecosystem services for varying policy alternatives have received much attention in environmental research (Termansen, McClean et al. 2013). Numerous studies in environmental valuation have exhibited the advantages of incorporating spatial data and physical models into econometric studies (Bohlen and Lewis 2009, Kousky and Walls 2014, Tapsuwan, Polyakov et al. 2015). Spatial variations in environmental functions have a significant influence on the availability and quality of ecosystem services and may have a consequential impact on stakeholder preferences. For instance, proximal and distal effects from anthropogenic improvements and environmental amenities have been shown to influence the formation of stakeholder cognition and preferences (e.g., Cai, Cameron et al. 2011, Borchers and Duke 2012, Bowman, Tyndall et al. 2012).

In this research, proximity-to-amenities and hazard-susceptibility were utilized to explain stakeholders' choice heterogeneity. Proximity to the river and forests have previously been identified as significant determinants influencing economic value (e.g., Snyder, Kilgore et al. 2007, White and Leefers 2007, Pfluger, Rackham et al. 2010, Tapsuwan, Polyakov et al. 2015). Nearness to waterways and forest areas have been associated with having positive effects on property prices (Tapsuwan, Polyakov et al. 2015, Nicholls and Crompton 2017). Environmental hazards can also affect housing markets and the economic value of environmental goods. The potential loss or disruption of the use of environmental amenities due to hazards can significantly alter value cognition and stakeholder preference. Potential costs associated with natural disasters come in the form of human losses, infrastructure damages, and significant disruptions in economic activities (Shrestha, Okazumi et al. 2016). For instance, high risks of flooding and landslides have been shown to influence the marginal WTP for conservation and mitigation measures (Daniel, Florax et al. 2009). Analyzing the complicated relationship of spatial attributes on stakeholder choices and their willingness to pay for soil amenity improvements helps policymakers in evaluating options and costs more efficiently in allocating resources.

6.2. Methodology

6.2.1. Survey design and implementation

The questionnaire design in this study followed the guidelines specified by Bateman, Carson et al. (2002). A preliminary focus group discussion was conducted with seven representatives from the local government, agriculture office, environment office, and farmer-leaders in December 2014. Before the conduct of the survey, focus group discussions were organized with the various communities, including the local and provincial government representatives and barangay (community) leaders. The discussions centered around environmental risks, logistical concerns, and the local understanding of soil issues including use, management, and conservation. Quotas for several socio-demographic parameters (e.g., age, income, education) were established to resemble the demographics of the head of the family for the entire population. A draft questionnaire and survey strategy were developed and finalized after a round of pre-testing with personnel from the local agriculture's office and barangay officials.

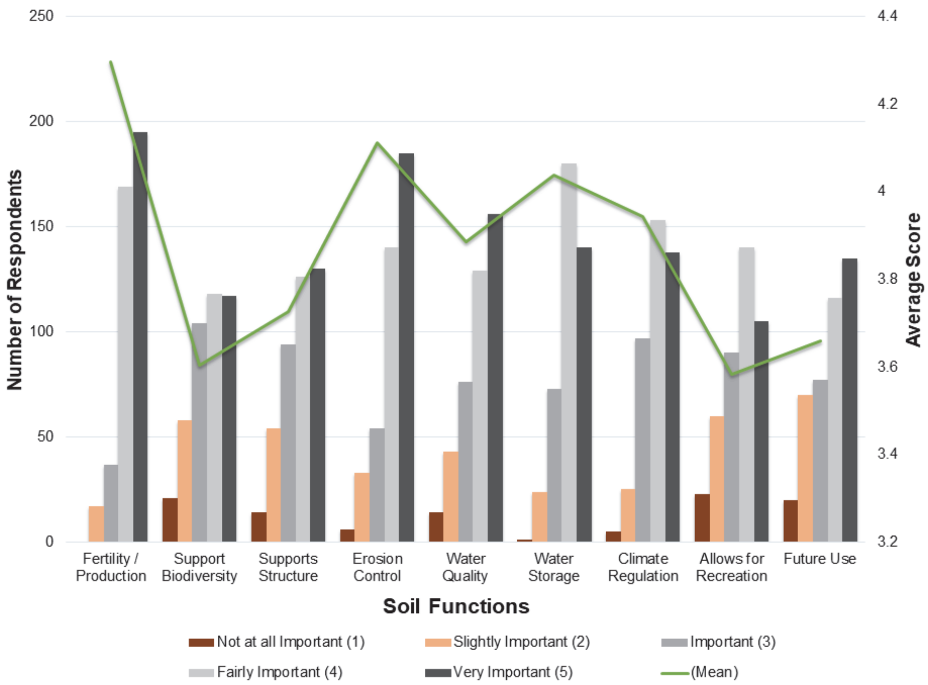


Figure 6-1. Results of the preliminary survey asking the respondents to gauge the level of importance of the different soil functions

a preliminary survey conducted a year earlier regarding which soil functions the respondents considered as directly significant to their lives. Through a Likert-scale study, the respondents were asked to rate a list of soil functions from ***not at all important*** (1) to ***very important*** (5). The results of the preliminary survey are shown in **Fig. 6-1**.

CASE 1	OPTION 1	OPTION 2	STATUS QUO
Water Storage Capacity Increased water infiltration and holding capacity thereby regulating discharge and mitigates flooding	L3: High Improvement (greatly increases soil's water holding capacity)	L1: Low Improvement (Status Quo)	L1: Low Improvement (Status Quo)
Erosion & Sediment Yield Control Reduced soil loss and the resulting sediment yield	L3: High Improvement (Reduces soil loss by 20%)	L2: Mod Improvement (Reduces soil loss by 10%)	L1: Low Improvement (Status Quo)
Carbon Sequestration Capacity Increased SOC through reduced C-loss and increased carbon inputs in soil	L1: Low Improvement (Status Quo)	L2: Mod Improvement (Moderately reduces C loss)	L1: Low Improvement (Status Quo)
Additional Cost (annual cost for every household)	P 150.00	P 50.00	P0.00 No Additional Cost
Choice for this scenario case: (please choose your preferred option)	☐	☐	☐

Figure 6-2. Example of the choice set

A D-efficient partial profile[10] choice design was implemented, having some guaranteed accuracy in parameter estimation and good prediction capability (Kessels, Jones et al. 2011). There were 54 choices generated, partitioned in 18 questionnaires, with each respondent presented with three case sets with varying levels of improvements in soil amenity. Three levels of amenity improvements were used: low (L1), moderate (L2) and high (L3). A fourth attribute was 'cost' which was also given in four levels: ₱0, ₱50, ₱100 and ₱150. The payment vehicle (cost) to be used in calculating welfare estimates, was in the form of an annual watershed mandatory fee, which has frequently been used in environmental DCE (Morse-Jones, Bateman et al. 2012). The community fund collected from every household would be used to fund additional conservation strategies aimed at improving soil amenities from the watershed. This was considered as the ideal option with regards to the credibility of the created hypothetical market since the respondents are very familiar with this type of fiscal instrument. Each respondent was asked to

[10] D-efficient (or d-optimal) design provides straight optimization based on a chosen criterion and the model that will fit it. Instead of using classical designs of fractional factorials, the d-efficient matrices are often non-orthogonal and come with correlated effect estimates.

answer three randomly generated case sets, with each case set having two randomly generated scenarios and a third option representing the status quo. To minimize ordering effects, the order of the choice sets and the arrangement of soil attributes were altered in each choice set. The status quo alternative became the default third choice option with all Level 1 improvements and no associated financial burden (cost = ₱0.00). **Fig. 6-2** presents an example of choice sets involving two alternatives and the status quo option.

To help explain stakeholders' heterogeneity of preference, the socio-demographic characteristics, spatial effects, and environmental consciousness scoring interacted with the attributes of soil ancillary functions were included. A five-point Likert-scale test was used to measure environmental awareness (EA) score, which centered on the respondents' openness for community initiatives to deal with soil conservation, and perceived awareness regarding soil functions (shown in **Appendix H**). Supplemental questions were included to mask protest responses. Those who answered 'no' were then excluded from the analysis and deemed as protest responses. Pilot testing was conducted with 18 residents to estimate the time requirements, comprehensibility of questionnaires, and other logistical considerations. This allowed us to approximate the amount of time needed for each interview, and to prepare the schedule and other requirements of the survey proper.

6.2.2. **Survey Implementation**

The actual CE survey was conducted from February to April 2015 using face-to-face interviews, with support from the municipal government and Norzagaray's agriculture office. Eleven local folks were trained and employed as support staff for data collection. Each questionnaire took an average of thirty minutes to complete, implemented by the lead researcher and at least one support staff. Before the survey proper, a verbal presentation explaining the overview and overall research objectives were provided to the respondents and local leaders (barangay officials and farmer groups).

Stratified random sampling was performed in selecting 450 residents conducted via house visits. To ensure the sample population reflected the agricultural versus non-agricultural demographics of the community, two-thirds of the respondents were targeted to come from agrarian households. Nine of the eleven barangays with substantial agricultural farms were selected as part of this study, with each having 50 households (30 agricultural, 20 non-agricultural) chosen as respondents. Aside from completing the survey questionnaires, the geographic coordinates of the respondent were also indicated. A handheld GPS was used to acquire the spatial coordinates, which was taken on the entrance of the respondent's landholding. As an additional check on the spatial location, a printed remote sensed image of the area was used as a base map to identify the respondent's location.

6.3. Results

Out of the total 450 respondents, 38 responses were eliminated as protest bids which came from respondents who signified their unwillingness to contribute financially to the project. The demographic breakdown of the remaining 412 responses is shown in **Table 6-1**. The survey was implemented at an average of 20 minutes per respondent. The average age of the respondents was 52, and an average household size of 5.0. For industry type, all non-agricultural employment types were combined into a single class due to limited representation. The other socio-demographic attributes (education, income, age and household size) were reasonably representative of the Norzagaray population.

Table 6-1. Socio-economic characteristics of respondents and attributes of their environment

Parameter	Description	Mean	Std Dev
Age	Age of respondent in years	52.35	12.32
Household Size	Number of family dependents including head	4.60	1.44
Environmentalism Score	1 - Very Low; 2 - Low; 3 - Moderate; 4 - High; 5 - Very High	3.96	0.60
Industry Type	1 - agricultural; 2 - non-agricultural	1.33	0.47
Education	Without high school diploma – 39.81% With HS diploma – 29.85% Technical/trade school – 20.87% College degree – 25.24%		
Annual Income	Less than ₱40,000 – 39.56% ₱40,000 to ₱69,999 – 14.32% ₱70,000 to ₱99,999 – 20.87% More than ₱100,000 – 25.24%		
Forest Zone	1 - w/in 1.5km from forest reserve; 2 - 1.5-3.0km zone; 3 - 3.0-4.5km zone; 4 - beyond 4.5km zone	3.28	0.02
Water Zone	1 - w/in 1km zone; 2 - 1.0-2.0km zone; 3 - beyond 2.0km zone	1.79	0.02
Landslide Risk	0 - no risk; 1 - low risk; 2 - moderate risk; 3 - high Risk	0.65	0.02
Erosion Susceptibility	0 - negligible to very low; 1 - low; 2 - moderate; 3 - high; 4 - very high	1.26	0.03
Flood Risk	0 - no risk; 1 - low risk; 2 - medium risk; 3 - high risk	0.35	0.02

The environmental self-assessment survey revealed that majority of the respondents (66%) had high environmental awareness with an average score of at least 4.0. The majority of the respondents indicated their awareness of the regulating and supporting functions of soil (91%), and stated they personally benefitted from soil protection (79%). Sixty-two percent of those surveyed concurred acceptance for additional remuneration towards watershed preservation. Likewise, the majority of respondents indicated their agreement regarding the protection of watershed as part of their social responsibility (57%) and that its protection was essential for the future generation (64%).

6.3.1. Utility Model Estimation

The parameter estimates from the empirical analyses are shown in **Table 6-2.** Both models were estimated for comparison, to analyze for the presence of significant random effects.[11] Both models generated mean parameter estimates that were statistically significant on at least the 10% level. The signs of all attribute estimates were consistent on both models. The primary soil attributes showed a priori positive coefficients, while the cost attribute coefficient had a negative sign. These results are in line with basic economic principles indicating behavioral preference for higher quality goods and lower prices.

Table 6-2. Summary of estimates for conditional logit (CL) and random parameter logit model (RPL-I)

	CL	RPL-I	
Term	Coefficient (Error)	Mean Coeff (Error)	Std Dev (Error)
ASC	-1.4517 (0.2412)**	-1.4561 (0.2420)***	0.0188 (0.0074)***
Water Storage	0.0338 (0.0501)**	0.0362 (0.0502)**	-0.0558 (0.0475)***
Erosion Control	0.3141 (0.0502)**	0.3112 (0.0504)**	-0.0880 (0.0530)***
Carbon Sequestration	0.0276 (0.0062)**	0.0287 (0.0064)**	0.0142 (0.0541)**
Cost	-0.0053 (0.0010)***	-0.0053 (0.0010)***	
AIC	2223.640	2226.992	
BIC	2249.189	2272.922	
Log Likelihood	-1106.796	-1104.423	

∗∗∗ Significant at 1%-level; ∗∗ Significant at 5%-level; ∗ Significant at 10%-level; ASC = Alternative Specific Constant; AIC = Akaike Information Criterion; BIC = Bayesian Information Criterion

[11] *RPL is the more mathematically complex option, and relaxes much of the assumptions in CL. If they have very similar results, the presence of random effects does not necessitate further investigation.*

Table 6-3. RPL (RPL-II) with interaction effects with socio-economic and EVI covariates

Attribute	Mean	Std Dev	x AGE	x HHO	x INC	x EDU	x IND	x EVI
ASC	1.912**	0.011*	-0.010*	-0.294	-0.2608*	0.3949*	-0.2309	0.0132*
	(0.150)	(0.009)	(0.004)	(0.181)	(0.1204)	(0.2072)	(0.5631)	(0.0092)
Water	0.320**	-0.058**	0.003*	0.0004	0.0773**	0.0534*	0.0362	0.0201**
Storage	(0.155)	(0.018)	(0.004)	(0.039)	(0.0259)	(0.0217)	(0.1173)	(0.0307)
Erosion	0.743**	-0.092**	0.001	-0.031	0.0182**	0.0603*	0.0662	-0.0204
Control	(0.253)	(0.035)	(0.004)	(0.038)	(0.0046)	(0.0308)	(0.0260)*	(0.0686)
Carbon	0.722**	0.004	0.005	0.079*	-0.0415	0.0735**	-0.0868	0.0171*
Seqtn	(0.212)	(0.056)	(0.003)	(0.030)	(0.0516)	(0.0190)	(0.1359)	(0.0099)
Cost	-0.030***		-0.0004***	0.001*	-0.0004	0.0027**	-0.0001	0.0058***
	(0.010)		(0.0001)	(0.0003)	(0.0009)	(0.007)	(0.0024)	(0.0014)

*** *Significant at 1%-level;* ** *Significant at 5%-level;* * *Significant at 10%-level*

Table 6-4. RPL (RPL-III) with interaction effects of attributes with spatial covariates

Attribute	Mean	Std Dev	x FOR	x WAT	x LAN	x ERO	x FLO
ASC	-3.0804*	0.0304**	0.1128	0.0004*	0.1109	0.0657	-0.1065
	(1.3232)	(0.0081)	(0.1509)	(0.0003)	(0.2758)	(0.2279)	(0.3713)
Water	0.1775**	-0.0555	0.0411*	-5.71E-05*	0.0781**	0.0533*	0.1003**
Storage	(0.0656)	(0.0482)	(0.0298)	(5.37E-06)	(0.0073)	(0.0266)	(0.0187)
Erosion	0.2388**	-0.0921	0.0563	7.75E-05	0.0756	0.0258**	0.0260*
Control	(0.0804)	(0.0539)	(0.0305)	(5.62E-05)	(0.0562)*	(0.0075)	(0.0796)
Carbon	0.2089***	0.0053	0.0101	-9.34E-06	0.0293	-0.0669	-0.0135
Seqtn	(0.0807)	(0.0549)	(0.0144)	(5.81E-06)	(0.0652)	(0.0535)	(0.0911)
Cost	-0.0208**		-0.0011**	1.11E-06	-0.0025*	0.0036**	0.0028*
	(0.0056)		(0.0006)	(1.12E-06)	(0.0012)	(0.0010)	(0.0017)

*** *Significant at 1%-level;* ** *Significant at 5%-level;* * *Significant at 10%-level*

For the RPL, various distributions were undertaken and showed minimal effect. Thus, a normal distribution was selected for the survey attributes, except for the cost parameter which was treated as a fixed variable. The number of observations was 1236 given that every respondent answered three choice sets each. The parameter estimates for the RPL models were determined using 2000 random draws. The estimates of the RPL model indicate significant preference heterogeneity, as the standard deviations for the three soil attributes were found to be significant.

A second RPL model (RPL-II) is presented in **Table 6-3,** elaborating on the interaction effects of the respondents' socio-demographic attributes. The results suggest that individual preferences may be related to auxiliary factors particularly education, income, and environmental awareness (EA). The EA score was found to have a positive and significant impact on cost and water regulation attribute. Respondents with higher EA scores preferred higher level improvements for water storage capacity and were willing to spend more for soil improvements. There was significant preference for improving soil's carbon sequestration among highly-educated respondents, while higher-income respondents favored improvements for water storage and erosion control. Age was also found to have significant impact on the stakeholders' decision-making process. Older respondents showed significant preference for lower-priced options. As for industry type, there was no substantial evidence indicating statistically significant difference in preference from agricultural and non-agricultural respondents.

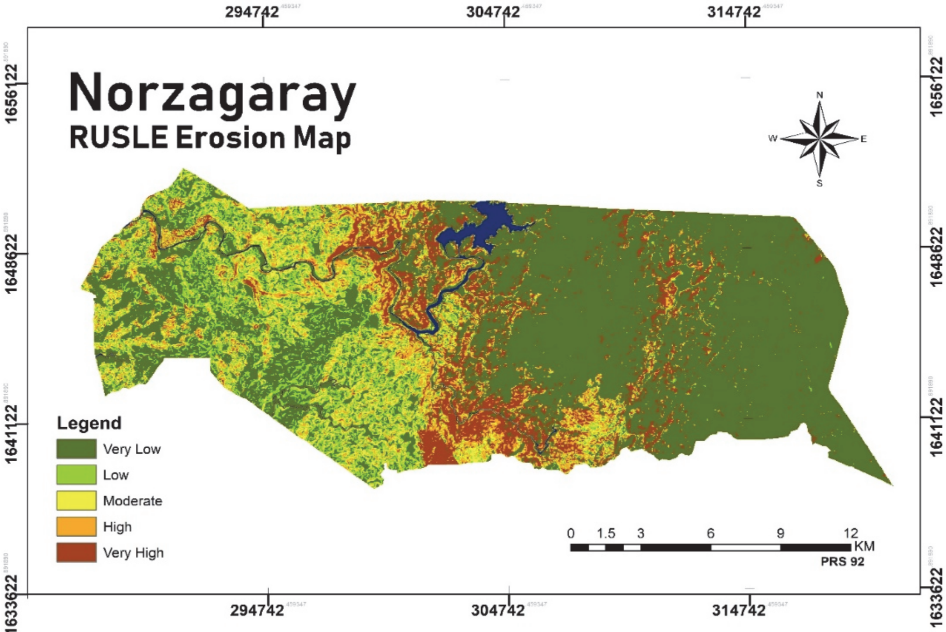


Figure 6-3. Generated soil erosion vulnerability map

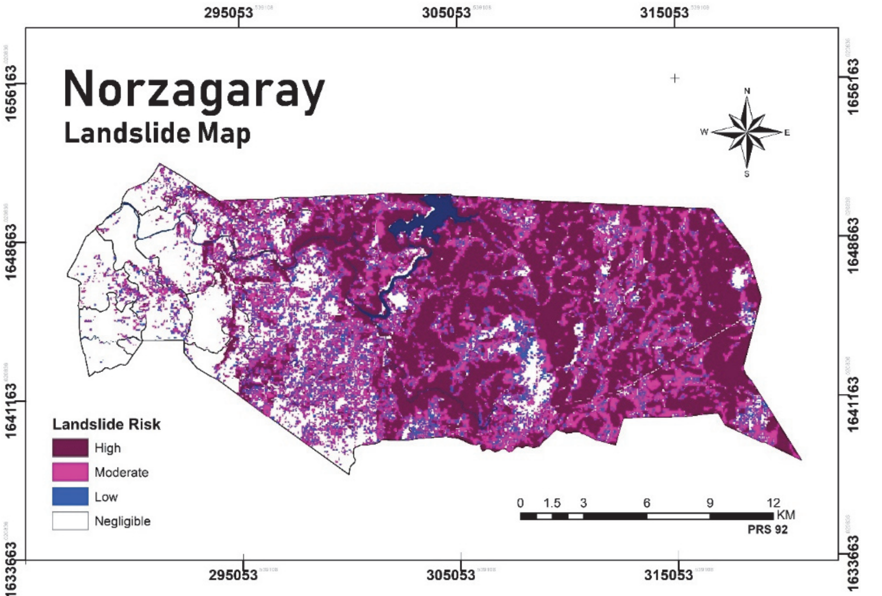


Figure 6-4. Landslide Map for Norzagaray
(adapted from the Bulacan Provincial Disaster Risk Reduction and Management Council)

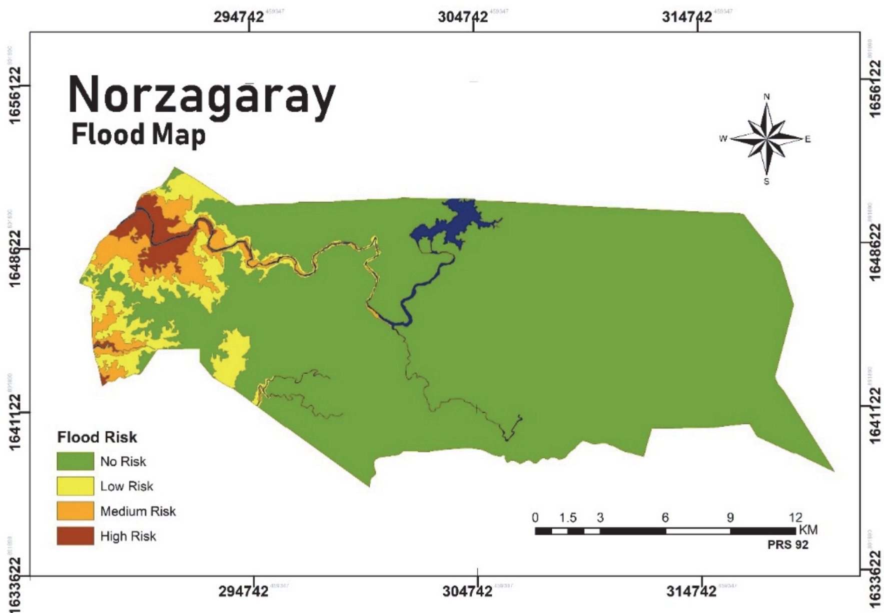


Figure 6-5. Flood Map for Norzagaray
(adapted from the Bulacan Provincial Disaster Risk Reduction and Management Council)

To further explain the respondents' choice heterogeneity, spatial effects were integrated into the empirical choice model. Risk/hazard maps in erosion, landslides, and flooding were utilized to determine significant effects in stakeholder preferences. The soil erosion map, as shown in **Figure 6-3**, was generated based on the RUSLE model. The landslide map (**Figure 6-4**) and flood map (**Figure 6-5**) were acquired from the provincial government of Bulacan through the Provincial Disaster Risk Reduction and Management Council (PDRRMC). Proximity analyses from major waterways and forest reserves were also executed to show significant effects on preference heterogeneity. The results of the random parameter logit (RPL-3) with spatial covariates are summarized in **Table 6-4**. Respondents living near rivers and in flood risk areas were found to significantly favor improvements in water storage capacity. Those living in areas highly vulnerable to erosion were found to prefer improving erosion control.

6.3.2. Marginal Willingness-to-Pay

The marginal welfare estimates simulated from the RPL model are reported in **Table 6-5**. The soil attributes were treated as ordinal data to calculate the marginal WTP for each improvement level. L1 levels in attribute improvements and ₱0.00 in cost were used as baseline values. Welfare estimation revealed that the respondents were willing to pay significantly more for greater levels of improvements on all three soil amenities. The water storage function was shown to elicit the highest marginal WTP, which was followed by erosion control. Superior improvement levels were valued higher for each of the soil amenities, which would indicate that the respondents were able to discriminate and value all three soil parameters distinctively.

Table 6-5. Average Marginal WTP estimates

Attribute	Level	Improvement	Mean Coefficient	MWTP	Lower 95%	Upper 95%
Water Storage	L2	Moderate	2.249 (0.175)	₱64.61 (5.00)	₱54.81	₱74.42
	L3	High	2.757 (0.191)	₱143.82 (6.38)	₱131.32	₱156.32
Erosion Control	L2	Moderate	1.403 (0.180)	₱40.32 (4.83)	₱30.84	₱49.79
	L3	High	1.959 (0.157)	₱96.59 (5.44)	₱85.93	₱107.25
Carbon Seqstn	L2	Moderate	1.438 (0.140)	₱41.30 (3.83)	₱33.80	₱48.81
	L3	High	1.029 (0.182)	₱70.87 (5.58)	₱59.93	₱81.80

All parameter estimates were computed relative to baseline levels (L1).
$1 (in April 2015) = PhP 44.63

Further analysis was conducted to determine preference variations for soil amenity improvements among the different user groups, specifically between agricultural and non-agricultural households. **Table 6-6** shows the summary of WTP estimates for these stakeholder groups. Both groups were shown to have the highest WTP on water storage capacity, with no significant difference between groups. However, there was significant difference in estimated WTP values for erosion control and carbon sequestration. Agricultural households were estimated to have significantly higher WTP for erosion control, while non-agricultural respondents had higher marginal WTP for carbon sequestration improvements.

Table 6-6. Marginal WTP for soil improvements comparing values from agricultural vs non-agricultural respondents

		Farmer			Non-Farmer		
	Level	Parameter Coefficient	Change in WTP	Std Error	Parameter Coefficient	Change in WTP	Std Error
Water Storage	L2	2.472	66.55	5.92	1.898	67.23	10.28
	L3	2.956	146.14	7.48	2.185	144.62	13.23
Erosion Control	L2	1.737	46.77	5.66	0.776	27.48	9.58
	L3	1.775	94.54	6.32	2.115	102.40	11.27
Carbon Seqtn	L2	1.430	38.50	4.37	1.400	49.60	8.17
	L3	1.109	68.36	6.61	0.762	76.58	11.32

6.3.3. Spatial Effects

The data points were regrouped into binary clusters to investigate the effects of spatial attributes (see **Appendices I – M** for additional tables). The presence of environmental risks was found to significantly affect stakeholder preferences. The risk of soil erosion was especially determinative in increasing WTP values. Across all the different soil attributes, respondents in erosion-prone areas were willing to pay more for improvements. There was moderate but significant difference detected for moderate level improvements in erosion control, with those living in low-incidence areas providing higher WTP values. For the erosion susceptibility parameter, the results showed significant WTP difference in L3 improvements for water storage capacity and erosion control. This suggests that experiencing the impact of soil erosion firsthand can significantly influence the perception regarding the importance of soil services. Landslide risk was also found to influence stakeholder preference. Respondents in high-risk areas indicated significantly higher WTP values for high-level improvements in water storage capacity. Residents living in areas with high risks of landslides were willing to pay significantly higher to improve the soil's water storage capacity and reduce erosion rates. This could be indicative of how the respondents view the relation of the soil's water holding capacity, soil stability, hydrological peak flows, and landslide risks. While landslide remains a highly complex and multi-faceted process, the relationship of water retention capacity with landslide risks is more intuitively apparent based on its effect on overland flow and soil stability. This understanding, together with the prevalence of massive landslides in Norzagaray, could have contributed to the significance of landslide risks on influencing preference heterogeneity. For the flood risk parameter, there was no

significant difference found in the mean WTP estimates in any of the soil parameters. This suggests that the risk of flooding was not a significant determinant for WTP variability.

For proximity analyses to environmental features, the 1km distance from water tributaries and the 1.5km distance from forest reserves were used as demarcation to test their spatial effects in WTP variability. In the river-proximity analysis, no significant difference in the marginal WTP values was observed among the respondents within and outside the demarcation line. As for the forest proximity analysis, significant differences in marginal WTP were estimated between the clusters, for moderate improvements in erosion control, and high improvements for water storage capacity, erosion control, and carbon sequestration. Residents living nearer to the forest reserves were willing to pay more for improvements in all soil amenity types, which make proximity-to-forest a prime determinant for WTP. Similar findings from previous studies have found proximity to improvement sites or areas of concern (AOC) as having a significant effect on property prices, economic value and WTP (e.g., Tyrvainen 1997, Anthon, Thorsen et al. 2005, Braden, Taylor et al. 2008, Mueller, Springer et al. 2018).

6.4. **Discussion**

Land policies and conservation initiatives often include efficiently managing trade-offs and preserving the long-term economic utility of land (Rasul 2009, Laurans, Rankovic et al. 2013). In many instances, soil management of complex watersheds requires understanding the macro-ecology, balancing the needs of various stakeholders, and understanding the socio-economic constraints of the proximate communities. From the policy and decision-making perspective, identifying the various ecosystem services and soil amenities contributing directly and indirectly to human wellness is of valuable interest. Together with the direct use values, the indirect use values provide crucial metrics required in a more comprehensive examination of management options. In this study, we evaluated several soil functions using choice experiment analysis. The analyses of preference and cost for each soil service allowed further investigation of WTP and preference heterogeneity among the stakeholders.

The results of our RPL models showed the highest marginal WTP was towards improving the soil's water retention capacity. Post-survey discussions with stakeholders revealed that the respondents associated the improvement as a means to avert flooding events and to help replenish the water supply during the dry season. Given Norzagaray's well-pronounced dry and rainy seasons, the town's residents are well familiar with water-related problems such as flooding and water shortage, even on a 'normal' meteorological year. During El Nino and La Nina years, where there is unusual warming or cooling of the waters in the equatorial Pacific, Norzagaray experiences extended very dry and very rainy seasons which are disastrous to the town's agriculture industry. At the time when the survey was being implemented, the town and the rest of the country were making preparations for El Nino projected to occur in the last quarter of 2015. This could have influenced the respondents' preferences given that the last time El Nino occurred in 2010, Angat Reservoir reached historically low water levels which significantly affected much of the Central Luzon region.

The other soil attributes were also estimated with considerable WTP values. The respondents showed willingness to pay for improvements to minimize soil loss and lessen the sediment discharge into the water systems, even if the national government mainly finances the costs of maintaining the watershed and dredging operations. Post survey discussions with stakeholders suggest that this could be motivated by a combination of self- interests and altruistic desires to protect the watershed. Reducing soil erosion rates and sedimentation of the reservoir not only extends the economic life of the dam but is viewed by some Norzagaray residents as part of their social responsibility. Carbon sequestration was the least valued among the listed soil amenities, which coincidentally is the least perceivable. Our findings indicate that while the respondents were well-versed with climate change issues and showed they were willing to pay for improvements, they viewed carbon sequestration least compared with the other two.

Overall, the respondents showed positive attitude towards spending for soil improvements. The low incidence of status quo responses showed openness to spend more on improving soil amenities. This is despite having a population made up mainly of poor and low-income households, which contradicts the notion that impoverished communities have weak inclination for environmental improvements. Income, education and environmental consciousness were found to have direct effect on the respondents' preferences. Higher income and more educated respondents indicated higher WTP values, comparable to the results of similar choice experiments. Higher income residents are more likely to have available funds for incidental expenditures that can be used for soil management and conservation schemes. The results of our econometric modeling demonstrate that administrators can elicit greater community acceptance for public financed conservation measures if they can demonstrate direct or personal linkages to specific soil amenities. It also shows how spatial and socio-demographic factors could potentially impact stakeholder preferences, which could be later used in strategic planning.

6.5. Conclusion

Using a choice modeling experiment, the stakeholders' willingness to pay for soil conservation was measured which was used as economic metrics for the soil's stated value. A variety of socio-economic and spatial attributes were found to have significant effect on the explicit valuation of soil in the town on Norzagaray, Bulacan, which can be used in crafting suitable land use policies and conservation plans specific to the area. Understanding the nuances that help formulate people's value perception on ecosystem services help decision makers to consider trade-offs in policy and management approaches. Despite some methodological and data limitations, this study pushes our understanding of how stakeholders perceive and value soil amenities which can then be used for improving soil policies and land-use management. Developing effective policy designs is critical in attaining high participation in soil management, and in reaching conservation goals. This is particularly helpful for rural communities especially those with limited pecuniary capabilities. Priority preferences diverge among stakeholder groups which can be influenced by socio-demographic and spatial determinants.

In using the discrete choice experiment in estimating soil value, the study was able to estimate the stated value based on how much people would be willing to spend on conservation on specific soil functions. The innovative use of discrete choice experiment in soil valuation allowed the isolation of value for particular amenities, which showed that the respondents attributed economic worth to individual soil functions. The results revealed that for this particular study site, the respondents had preference for improving the soil's water holding capacity, with their preferences affected by various factors including income, education and environmental awareness. The inclusion of environmental risk factors and proximity to environmental amenities also provided additional rationalization to explain preference heterogeneity, which can be further explored in future soil valuation studies. Soil Valuation studies, especially those that focus on indirect utilities and non-use values, help provide a more inclusive understanding of the underlying nature of soil amenities on human welfare. This is especially critical for soils and other environmental public goods whose economic value have often overlooked.

Chapter 7. **Soil Value Assessment using Replacement Cost Method**

This chapter discusses the use of cost-based method in estimating the economic value of soil. Given the dualistic nature of soil of having non-marketable services as well as marketable amenities, the use of cost-based techniques in assessing the impact of degradation is available as an alternative valuation technique. Previous chapters have dwelt on pricing soil's economic value highly dependent on the stakeholder's willingness and ability to contribute, but commonly overlooking the actual costs associated with use and the contribution soil degradation.

7.1. Introduction

Soil conservation in watersheds has been directly and indirectly associated with the protection of water supply and energy production. However, implementation and management of soil conservation have been challenging. Local communities (e.g., farmers) who could introduce much of the conservation measures do not always see their direct economic benefits. All around the world, the increased sedimentation of reservoirs adversely impacts the water capacity which results in the decreased water supply to be used for domestic and irrigation purposes. Dams with hydro-electric power plants are also negatively impacted by reducing power generation efficiency and gutting the dam life by years.

The costs associated with the maintenance and repair, disruption of services, reconstruction of new facilities, and rehabilitation are the most direct and relatable types of values that people can associate with. The need for investments in environmental protection and sustainability of use can be defended in business perspective by estimating some of these costs. There has been a number of studies that have tried to estimate the costs associated with environmental degradation: several approaches to value have been applied to soil resources and have become part of cost-based approaches to soil value (Dimal 2015). One of the cost-based approaches that can be used to estimate the economic impact of soil conservation/degradation is the replacement cost method.

Replacement cost method (RCM) estimates the value of environmental damage based on the price needed to restore the environment from its previous undamaged state. The erosion and degradation of farmlands affect not only agricultural production upstream, but may also degrade reservoirs, contaminate water supplies, cause sedimentation in dams, or disrupt ecosystems downstream. The costs of rehabilitating the upstream farmlands, dredging sediment-filled reservoirs and decontaminating polluted water supplies would be tallied, which would then be used as an estimate. A modified RCM follows the use of possible projects to provide an equal alternative to the environmental good or service that would have been lost due to degradation. The different costs of the shadow project would then be calculated and then used as the estimate for the value of the environmental good. RCM has often been criticized whether the estimated value is reflective of the real cost of damage. Some argue that once the environment has been damaged, it would be unlikely that any amount would be able to restore it from its pristine state. Others fear that by using the replacement cost method, the assessment would only be reflective of the short- and medium-term consequences of environmental degradation while sacrificing the long-term impacts.

The main advantage of using cost-based techniques is that these are often less data- and resource-intensive compared to preference-based (i.e., stated and revealed) approaches (Notaro and Paletto 2012). Most of the data required in RCM are either readily available or can be generated through indirect measurement (e.g., beta-transfer analysis), which makes the use of the RCM a less costly alternative, which may work particularly well for the large scale (e.g., watershed) applications. One of its main limitations is that the estimated value of replacement may not always be a reliable metric for the benefits derived from the environment. One of the principal assumptions in RCM is that it considers secondary benefits or indirect amenities as irrelevant; otherwise, there would have an overstatement of value if this assumption does not hold (Notaro and Paletto 2012). Artificial substitutes that are used in estimating the values in RCM do not always provide the same level of services or similar array of benefits compared the benefits they are compared with. This commonly leads to the underestimation of the ecosystem's value. RCM, therefore, becomes more appropriate for estimating the economic value of single ecosystem services, or at least a limited number of services.

In this chapter, RCM was used to calculate the economic impact of soil degradation on reservoir management particularly on the resulting sedimentation occurring in the Angat watershed. The term 'replacement cost' in this case is interpreted as the damage cost, which is the cost needed to restore soil amenities that have been damaged due to soil degradation. Reservoir sedimentation was calculated from differences in bathymetry in three moments in time and compared with the cumulative estimated sediment yield from the catchment from intervening years (modeled with the RUSLE and different sediment delivery ratio methods). The catchment sediment yield was then compared to land use changes to project future costs. The costs of various watershed rehabilitation measures were also calculated to determine how much is needed to restore a variety of soil services in the upper watershed region.

7.2. Related Literature

Estimating sediment yield is a crucial factor in designing and managing hydroelectric power plants, dams, and reservoirs for flood control. Sedimentation reduces a reservoir's holding capacity, which results in lower economic efficiency and minimized capability against flooding. In reservoir management, dredging is a common practice to extend the economic viability and use of reservoirs that serve as water supply, flood control and in power generation. Dredging is the removal of loose sediments and debris from the bottom of the lakes, rivers, harbors and open seas to restore and lengthen the economic viability of water systems for specific uses. This is a short-sighted solution to the capacity loss problem, especially since the cost of dredging operations is astronomical. While not all reservoirs require immediate and constant dredging, a growing number of highly critical reservoirs have been rapidly losing their storage capacity with increasing rates of soil erosion. In the

Philippines, dredging is handled by the Department of Public Works and Highways (DPWH), and in coordination with other relevant agencies including the National Power Corporation (NAPOCOR), National Irrigation Authority (NIA), and the Metropolitan Water and Sewage System.

One of the main tasks in dredging operations is estimating the amount of sediments needed to be excavated. At the reservoir, sediment yield can directly be estimated using hydrographic surveys or by the sediment load measured at a specific point of interest (Ponce 1989). But In the absence of actual measurements, physical models and empirical approximations using statistical analysis can be used as alternatives. One such statistical approach makes use of the sediment delivery ratio (SDR) which estimates the amount of soil materials delivered to a specific point in the drainage system from the gross eroded materials detached from the whole watershed. The general equation for the SDR is expressed as:

$$SDR = SY / E \qquad \textbf{Eq 7-1}$$

where SDR is the sediment delivery ratio; SY is the sediment yield; and, E is the estimated total eroded material. The total eroded material (E) can be estimated by multiplying the predicted soil loss per unit area (A) (i.e., using the USLE/RUSLE to estimate A: see Chapter 3) with the size of the watershed.

SDR is affected by a variety of factors including sediment texture, proximity to the main channel, watershed area, channel density, land use and cover, and rainfall-run-off factors. The relationship between the SDR with the size of the drainage area is known as the SDR curve. Large watersheds with vast drainage sizes and fields with long distances to the streams have low SDR values because large areas have greater chance of trapping sediments along the way before they reach the end of the water channel system. At the regional scale (large and very large watersheds), the most commonly used method of estimating the SDR is through the SDR-catchment size power function given by the generic formula:

$$SDR = \alpha A^{\beta} \qquad \textbf{Eq 7-2}$$

where A is the area of the catchment, and α and β are empirical parameters (Maner, Saffan et al. 1962). Given that SDR values decrease with increasing catchment size, the parameter β is assumed to be negative, ranging from -0.01 to -0.025 (Ferro and Minacapilli 1995). Currently, there is no one universally accepted methodology used in estimated the SDR, although there have a been some studies and handbooks developed that propose some methodologies. Renfro (1975) formulated an SDR equation making use of observations of sediment yield in 14 watersheds in the Blackland Prairie Area in Texas (R^2= 0.92). Renfro's equation (Ward, Trimble et al. 2015) for SDR (in percent) estimation is:

$$\log_{10}(SDR\%) = 1.7935 - 0.14191 \log_{10}(A_K)$$ **Eq 7-3**

where A_K is the watershed's drainage area in square kilometers. The USDA-SCS 1979 developed their SDR estimation using measurements from the Blackland Prairie in Texas. The USDA equation for SDR (in ratio) is:

$$SDR = 0.5663\, A_K^{-0.11}$$ **Eq 7-4**

where A_K is the watershed's drainage area in square kilometers. Vanoni (1975) employed 300 watershed data from all over the world to develop the relationship model. This equation is considered as a more generalized estimation of SDR. The equation from Vanoni is:

$$SDR = 0.47305 A_K^{-0.125}$$ **Eq 7-5**

where A_K is the watershed's drainage area in square kilometers. While SDR has been used as an approximation technique in estimating the sedimentation for waterways, its use and application have to be taken with caution. In particular, the SDR-area formulations do not consider local particularities, such as topography, climatic conditions, soil and water channel attributes, land cover/use, and vegetation cover (Ponce 1989).

7.3. Study Area

The study was conducted in the Angat-Ipo Watershed, a critical reservation in the Philippines that supports the water and electricity needs for the country's capital region. The Angat-Ipo Watershed is located in the province of Bulacan and covers the towns of Dona Remedios Trinidad (also known as DRT), Norzagaray, and San Jose Del Monte. Located on Luzon Island, the Angat Watershed reservation is 62,309 hectares found at the southern tail of the Sierra Madre Mountain Range. Due to its critical nature, the Angat Watershed has been declared as a protected forest reserve for watershed purposes and cannot be subject to sale or settlement. In the 1990s, the Angat Watershed was considered as an "exceptional watershed in the Philippines." But from various anthropogenic pressures, the watershed has regularly been suffering numerous environmental problems.

The Angat hydroelectric power plant has been in operation since 1967. The main power house discharges water to the irrigation diversion dam, while an auxiliary power house discharges water into the Ipo Dam which is used as the source of domestic water supply for Metro Manila. It also provides the irrigation needs for much of Bulacan's rice and other agricultural production. This water source supplies more than 90% of the water needs the capital region's water needs, supports 2% of the country's rice production and supplies 2% of the Luzon Grid. The potential build-up of sediments threatens the economic life and viability of the dam to provide services. In managing the Angat Dam, sedimentation of the reservoir is extremely critical since it provides 95% of the

Metropolitan Manila's 11 million residents and irrigation needs of millions more in the downstream. It also produces 256MW of electricity used as a peaking power supply, and services as flood defense during extreme rainfall events.

Given the high volume and variability of rainfall events throughout the year coupled with the increasing demands of a growing population, ensuring high water capacities of reservoirs is important to provide greater flexibility of water supply being managed through dams. However, due to accelerated rates of erosion and sedimentation of main reservoirs, the water capacity in several dams have greatly diminished to the point that dredging has become a staple in reservoir management. The use of dams for power generation and water supply control have made soil management in the uplands even more critical. Currently, the Angat Watershed is administered by the National Power Corporation, while the Ipo Watershed is run by the Metropolitan Waterworks and Sewerage Systems (MWSS).

For administration and management purpose, the Angat Watershed has been subdivided into ten subcatchments (see **Figure 7.1**). These subcatchments can be further regrouped based on their location relative to the Angat Dam. The Upper basin, which provides the inflow water into the Angat Reservoir, is comprised of the following subcatchments: Magusong, Katmon, Maputi, Matulid, Talagio, Macua, and Angat Reservoir. The Lower Basin, which receives the outflow waters from the Angat Dam, includes Pako, Ipo, and Angat River. For this valuation study, the cost estimation from sedimentation of the Angat Main Reservoir is based on the soil loss contributions from the upstream subcatchments (Upper Basis Subcatchments). Similar to the approaches used in the previous chapters, the Revised Universal Soil Loss Equation (RUSLE) was used to approximate the rate of soil erosion in the Angat Watershed. The RUSLE model was implemented in ARCGIS10.5 in 30-meter grid cell raster maps.

Translating the quantities of removed sediments into sedimentation at the reservoir, several delivery models were utilized to estimating the sediment yield of the Upper Angat-watershed. While there are other physical variables to approximate sedimentation rates, the drainage area method has been the most widely acceptable empirical approach in estimating the SDR of a given watershed. In this particular study, three models were chosen to used to determine the SDR and estimate the sedimentation at the Angat Dam Reservoir.

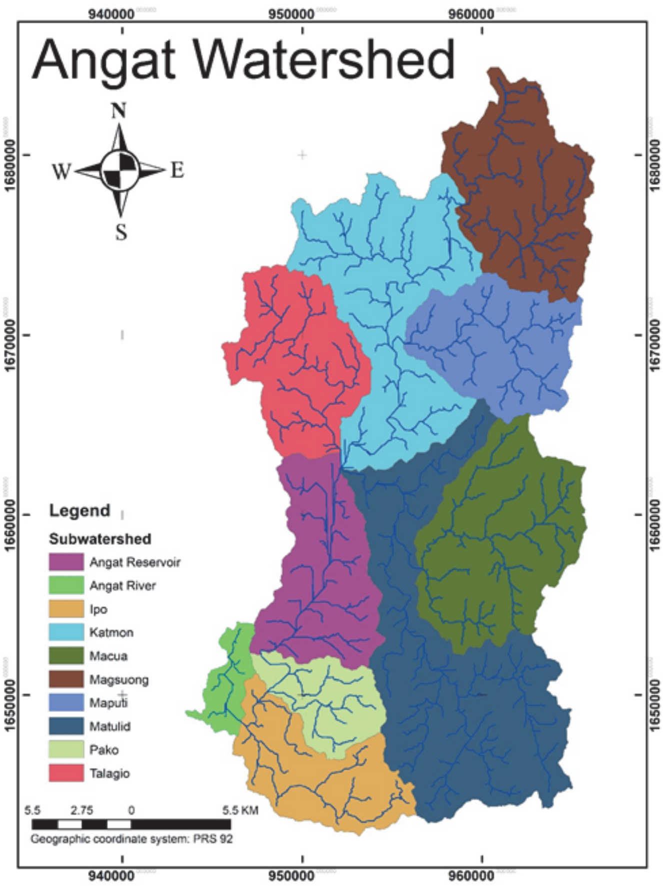


Figure 7-1. Different Angat Subwatersheds

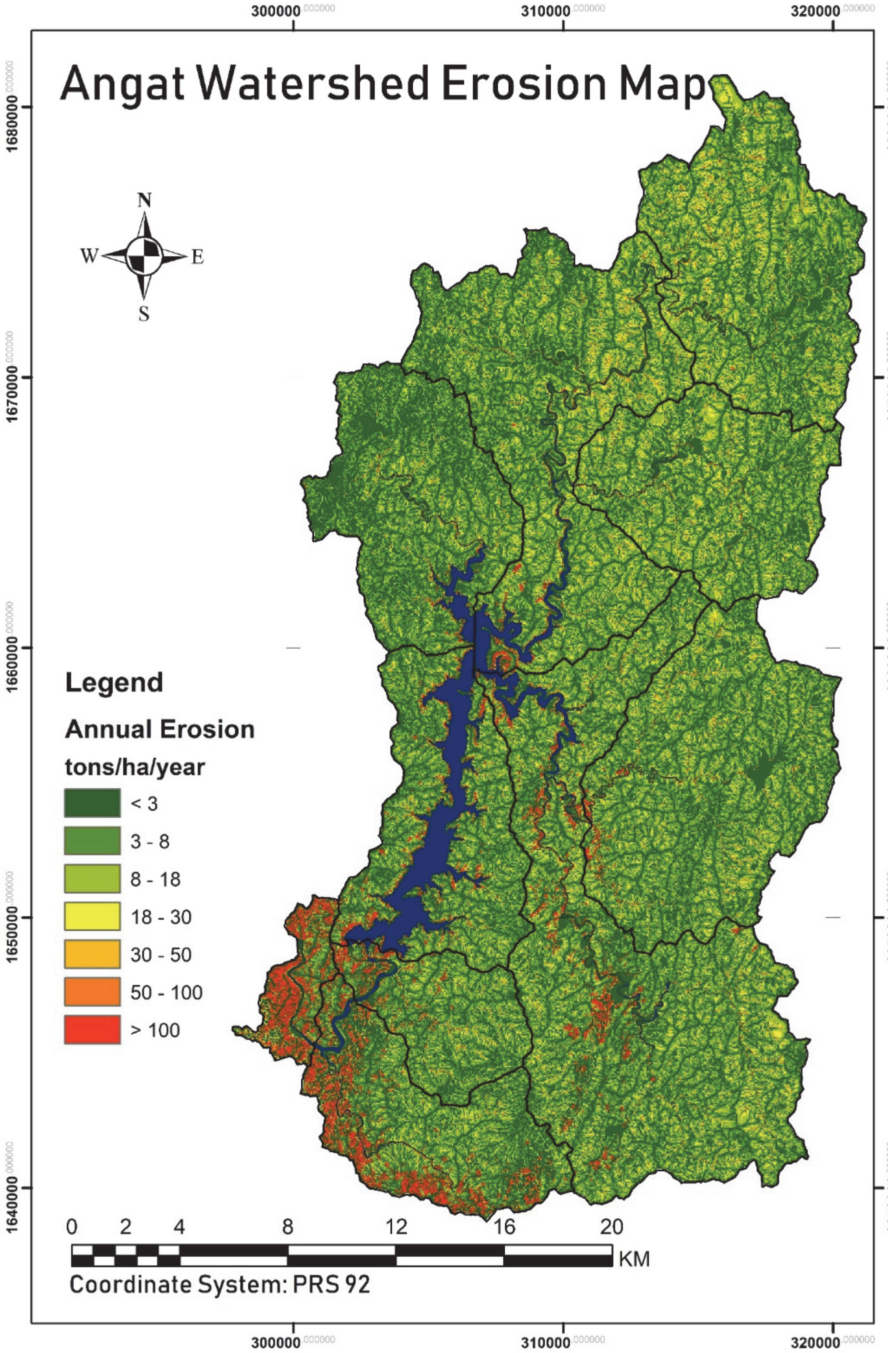


Figure 7-2. Soil Erosion Map of Angat Watershed

7.4. Estimating Cost from Dredging Operations

The factors used in the RUSLE model were evaluated using **Eqs 7.7 – 7.10**. **Figure 7.2** shows the generated soil erosion map for Norzagaray. Mean Annual Erosivity (***R***) values ranged from 1024.30 to 1399.17, while the soil erodibility (***K***) values ranged from 0.043 to 0.085. the cover management factor (***C***) ranged from 0 (water) to 1 (barren soil), with 90.91% of the total land area classified as forest lands. The combined slope length and steepness (***LS***) values ranged from 0 to 468.61.

Table 7-1. Erosion and Sediment Yield Estimates for Angat Reservoir

Watershed Name	Area (has)	Mean (tons/ha/yr)	Std Dev	Gross Eroded Mats (tons)	Eroded Vol (MCM)	Est. Sed Yield (MCM)		
						Renfro	Vanoni	USDA
Magsuong	6975.22	8.48	17.69	59131.82	2.37	0.589	0.501	0.660
Maputi	5334.45	8.11	14.06	43264.80	1.73	0.431	0.366	0.483
Talagio	5631.95	8.49	62.83	47820.88	1.91	0.477	0.405	0.534
Macua	8170.28	10.93	63.20	89280.25	3.57	0.890	0.756	0.996
Reservoir	5607.28	26.60	144.93	149157.14	5.97	1.487	1.263	1.665
Matulid	12843.69	22.39	123.42	287525.59	11.50	2.866	2.434	3.209
Katmon	9512.99	13.05	79.87	124137.74	4.97	1.237	1.051	1.385
TOTAL	54075.87			800318.23	32.01	7.976	6.775	8.932

MCM = million cubic meters

The sedimentation in the main reservoir was then approximated using SDR estimates from the three models. The SDR value using the Renfro Equation (**Eq 7.3**) was 24.92%, yielding an approximate value of 7.976 million cubic meters of sediment (MCM). Vanoni's Equation (**Eq 7.4**) produced an SDR estimate of 21.16% which is about 6.775 MCM total reservoir sedimentation, The USDA Equation (**Eq. 7.5**) had an estimated SDR value of 27.90% resulting in a yield of 8.932 MCM in sediments. The summary of results is shown in **Table 7-1**.

Table 7-2. Bathymetric survey results for 1994 and 2008 for the Angat Main Reservoir

Elevation (meters)	Reservoir Volume, MCM 1994	2008	Difference, MCM
160	137.4	124.75	12.65
155	104.89	97.26	7.63
150	79.94	69.77	10.17
145	59.2	50.75	8.45
140	42.32	31.73	10.59
135	28.93	20.35	8.58
130	18.64	8.97	9.67
125	11.09	4.73	6.36
120	5.87	0.50	5.37
115	2.59	0.25	2.34
110	0.81	0.00	0.81
105	0.11	0.00	0.11
100	0.00	0.00	0.00
		TOTAL:	82.73
		Annual SD:	5.91

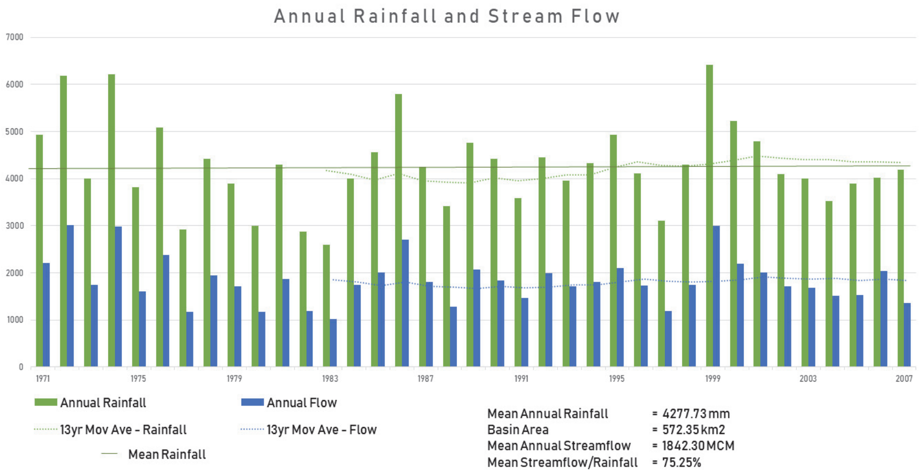


Figure 7-3. Annual rainfall and streamflow measured at the Angat Dam Station; also shown are their long-term and short-term averages

To validate the empirical estimates generated from SDR values, the results were compared with sedimentation values computed from previous bathymetric surveys. The bathymetric survey results from 1994 and 2008 were used in calculating the reservoir (water) volume difference, which was then converted as the volume of sediment materials trapped within the reservoir. The 1994 bathymetric survey was conducted as part of the Umiray Flood and Reservoir Operation Study (funded by the Asian Development Bank), while the

2008 bathymetric survey was from the Reservoir Operation Study by Japan International Cooperation Agency (JICA). The summary of results of the bathymetric surveys for 1994 and 2008 is presented in **Table 7-2**.

The total sediment build-up from 1994-2008 was 82.73MCM which reflected an annual sediment yield of about 5.91MCM. This sedimentation value from direct bathymetric measurements should be used in caution when comparing with the estimates from RUSLE-SDR estimates[12]. The results from bathymetric surveys pertained to sedimentation occurring from 1994 to 2007, while the empirical estimates made use of the 2008 land classification map and average rainfall values from 1971 to 2007. Any direct comparison between these two estimates would have two be predicated on two important premises: rainfall averages should be comparable, and land use/land cover did not dramatically change between the two dates. The summary of annual rainfall and stream flow values for the Magat Dam Station is presented in **Figure 7.3**. The 36-year (1971-2007) mean annual rainfall was 4277.73mm/yr, while the 13-year (1994-2007) average rainfall was 4348.46mm/yr. The absolute difference of 70.73 between the two rainfall averages (or about 1.65%) provides a reasonable argument that the values could be assumed as comparable.

For the purpose of comparison, the landcover maps for 1996 and 2016 were also generated from Landsat Images (April 1996 and May 2016), and are shown in **Figure 7.4**. Visual comparison of the generated land-cover maps already indicated a significant amount of cover change, particularly on the southern half of the watershed. To simplify the quantification of landcover change within the region, three landcover clusters were used: (1) forest, (2) agricultural and grasslands, (3) barren, and urban/peri-urban areas.

[12] *The rainfall value used in the RUSLE computations was based from rainfall measurements from 1971 to 2007.*

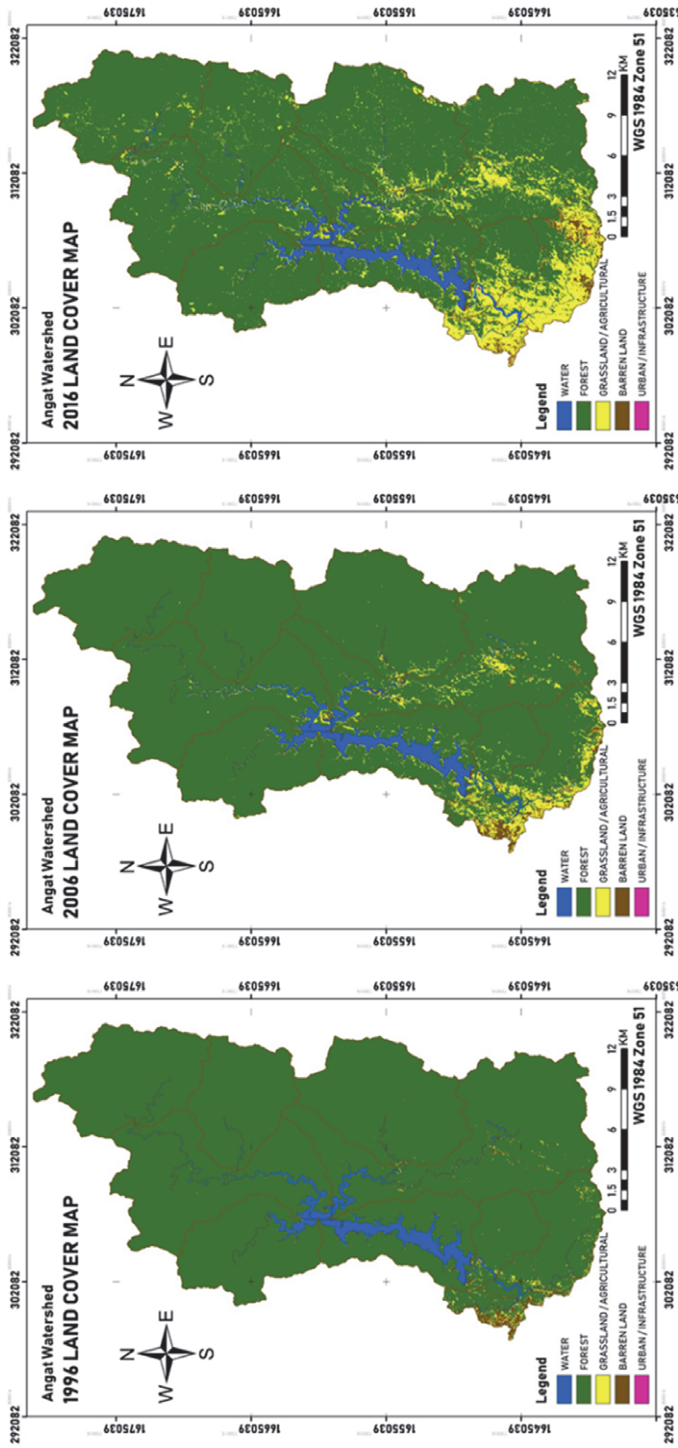
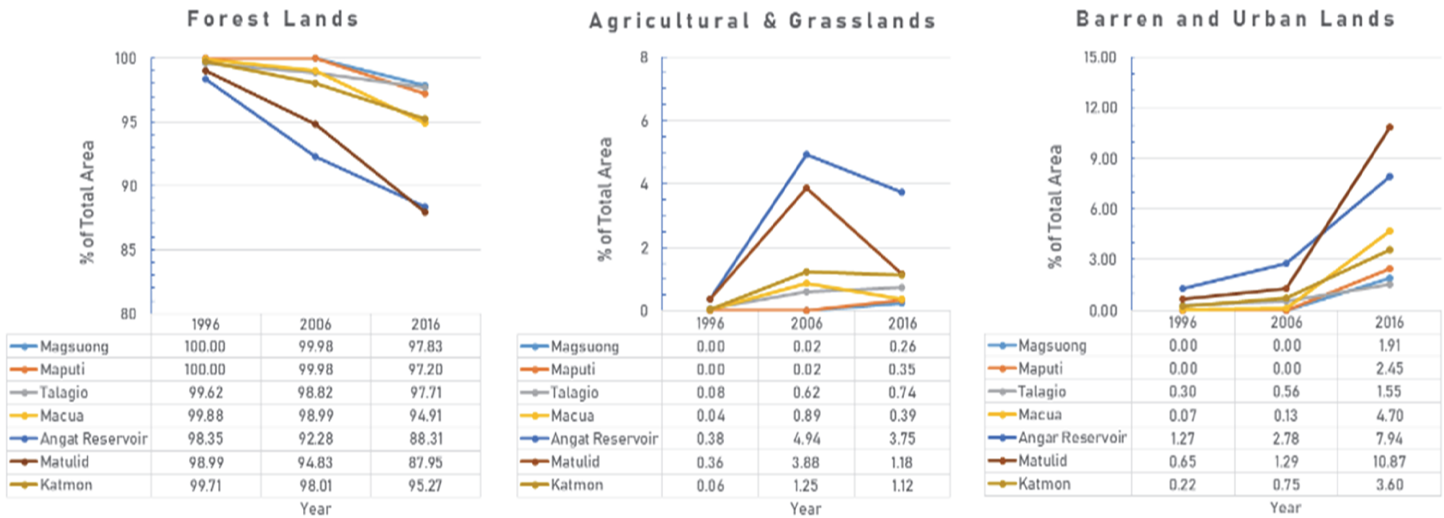


Figure 7-4. Changes in landcover for Angat Watershed in the last 30 years

The area covered by each cluster was estimated for every time period for each subcatchment, as presented in **Figure 7.5**. The downstream sub-basins of Angat River, Ipo and Pako, showed the highest cover changes overall, while the Angat Reservoir and Matulid subcatchments had the most significant change among the upstream sub-basins. The over-all trend showed significant decrease in forest cover and concomitant increases in agricultural and grasslands, and in barren and urban areas over the years. According to government reports (Municipal Government of Norzagaray 2010, Pascual 2013), the degradation of forest cover has been mainly caused by the unauthorized conversion of forestlands to a variety of land uses, particularly towards agriculture, timber, mining, and charcoal production. The change into agricultural and grasslands was particularly noticeable from 1996-2006 when a substantial number of forest lands had been converted for agricultural production. These numbers partially contracted in 2016, when stricter implementation of watershed management protection laws was enacted and when alternative sources of livelihood other than farming became more profitable (Pascual 2013).

The expansion of urban and peri-urban areas has been increasing, particularly in the more accessible downstream subcatchments. Open-pit Mining and quarrying operations also intensified during this period, particularly in the Ipo and Angat River subbasins. These cover changes particularly in the mountainous uplands of the watershed could greatly affect soil detachment rates, and thus constrains the direct comparison of RUSLE-SDR values with bathymetric results. To factor in the apparent effect of land cover change in erosion and sediment yield estimation, the amount of eroded materials was calculated for 1996 and 2016. Assuming RKLS to be constant, the cover management variable (C) was modified to reflect the land cover classification for the specified year. The mean erosion rates and the estimated total eroded sediments were then computed for the three time periods, and are presented in **Table 7-3**. The total soil loss (tons/yr) for 1996 and 2016 were estimated to be 431931.49 and 1379130.55 tons respectively.

Linear regression was used to model the increase erosion rate from the land cover change using the computed mean erosion rates (tons/ha/yr) for 1996, 2006 and 2016. The projected slope was 0.876 (see **Figure 7-6**) which reflected the estimated annual increase in erosion rate for the entire watershed assuming that all the other parameters remain unchanged and the projected land cover change remains uniform. If this estimate holds true and remains constant, this upsruge in erosion rates could be economically and environmentally catastrophic to the Angat watershed, particularly to the dam and reservoir management.



Forest Lands	1996	2006	2016
Magsuong	100.00	99.98	97.83
Maputi	100.00	99.98	97.20
Talagio	99.62	98.82	97.71
Macua	99.88	98.99	94.91
Angat Reservoir	98.35	92.28	88.31
Matulid	98.99	94.83	87.95
Katmon	99.71	98.01	95.27

Agricultural & Grasslands	1996	2006	2016
Magsuong	0.00	0.02	0.26
Maputi	0.00	0.02	0.35
Talagio	0.08	0.62	0.74
Macua	0.04	0.89	0.39
Angat Reservoir	0.38	4.94	3.75
Matulid	0.36	3.88	1.18
Katmon	0.06	1.25	1.12

Barren and Urban Lands	1996	2006	2016
Magsuong	0.00	0.00	1.91
Maputi	0.00	0.00	2.45
Talagio	0.30	0.56	1.55
Macua	0.07	0.13	4.70
Angar Reservoir	1.27	2.78	7.94
Matulid	0.65	1.29	10.87
Katmon	0.22	0.75	3.60

Figure 7-5. Graphs showing the changes in land cover in the Angat Watershed for the years 1996, 2006 and 2016.

The leftmost graph shows a general trend of decreasing forest cover in the region, while the rightmost graph pertains to the barren and urban lands shown to be increasing fast. The middle graph shows lands classified as agricultural and grasslands, which show a mix of increases and decreases over the years.

Table 7-3. Summary of mean erosion rates per hectare and estimated total eroded sediments using RUSLE

		Mean Erosion Rates			Estimated Total Eroded Sediments		
		1996	2006	2016	1996	2006	2016
Magsuong	6975.22	7.56	8.48	16.93	52704.74	59131.82	118066.06
Maputi	5334.45	6.43	8.11	18.07	34326.17	43264.80	96396.72
Talagio	5631.95	5.45	8.49	12.07	30683.63	47820.88	67956.09
Macua	8170.28	8.88	10.93	23.12	72537.79	89280.25	188871.44
Angat Res'r	5607.28	8.90	26.60	37.92	49915.59	149157.14	212639.87
Matulid	12843.69	9.37	22.39	37.19	120308.03	287525.59	477603.94
Katmon	9512.99	7.51	13.05	22.87	71455.54	124137.74	217596.44
	AVERAGE	7.99	14.80	25.50			
	TOTAL				431931.49	800318.23	1379130.55

*Mean annual erosion rates are in tons/hectare; Total eroded sediments are in tons

To compare the effects on the estimated amount of sediment delivered to the main reservoir, the average sediment yield for 1996 and 2006 were calculated using the three SDR equations. The summary of yield estimates is shown in **Table 7-4**, including the difference of each with the measured sedimentation from bathymetric measurements. The Renfro equation was shown to provide the closest estimate with a slight overestimation of about 3.90% from the measured hydrographic results, while the Vanoni and USDA estimates were lower than 11.75% and greater than 16.35% respectively. While the accuracy estimates could be seen as encouraging, caution should be taken in making generalizations from these results given the amount of uncertainties and assumptions introduced in the model.

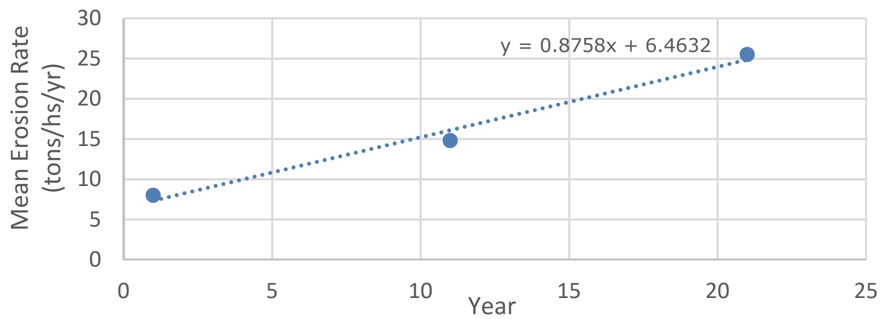


Figure 7-6. Estimated change in erosion rate from landcover change

Table 7-4. Summary estimates of sediment yield (in MCM) for 1996, 2006 and 2016 using the Renfro, Vanoni and USDA equations.

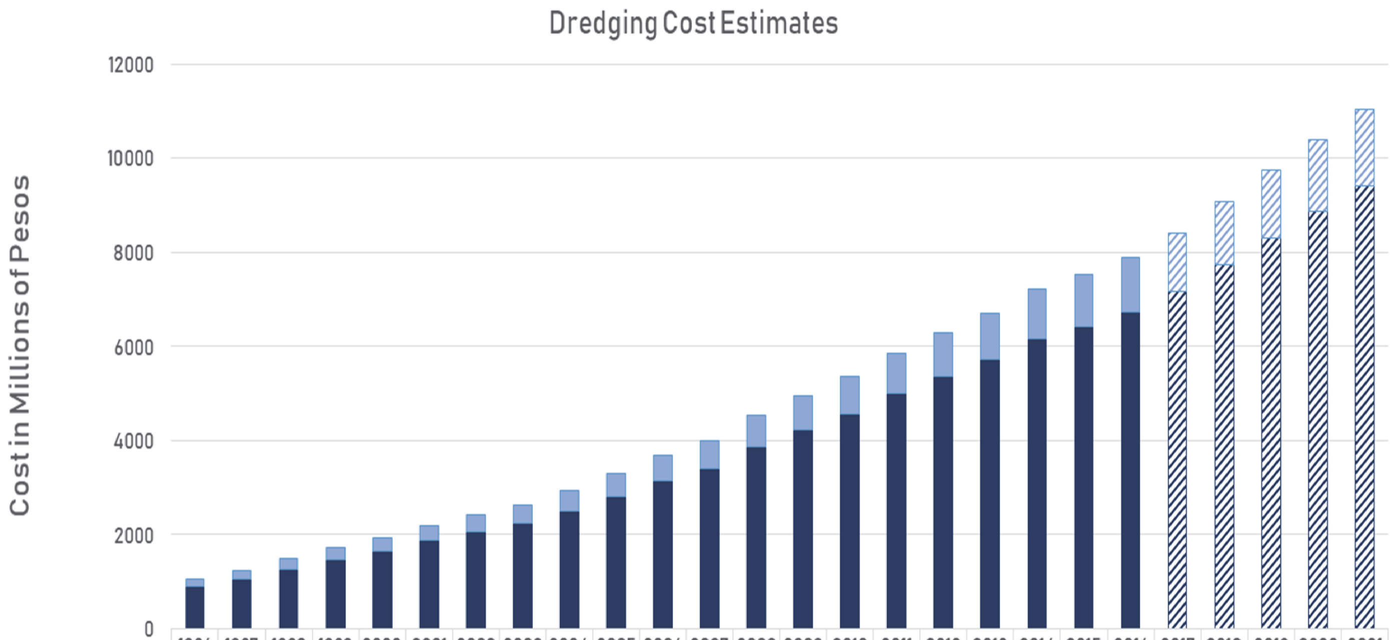
	1996	2006	2016	Average (1996-2006)	Diff from Bathymetric	% Diff
Renfro	4.30	7.98	13.75	6.14	0.23	3.90
Vanoni	3.66	6.78	11.68	5.22	-0.69	-11.75
USDA	4.82	8.93	15.39	6.88	0.97	16.35

The cost of dredging was then computed using the estimated sediment yield from the Renfro Equation. The cost of dredging operations, combining the price of sediment excavation and material disposal, was based on the costing estimates from the Philippine Department of Budget (2014). Inflationary rates were used to past estimates and future projections of total dredging costs from 1994 to 2021. The summary of dredging cost estimates is presented in **Figure 7-7**. The estimated dredging costs from sedimentation in 2014 was ₱7.21B when the cost estimates for sediment excavation and disposal were ₱480.24/cu.m and ₱80.23/cu.m respectively. The total dredging cost is projected to increase to ₱11.02B in 2021, which is almost double the cost in just 10 years.

7.5. Estimating Costs of Alternatives to Upland Rehabilitation

There are several policies and programs that can be used to respond to soil loss reservoir sedimentation. A well-maintained watershed that is able to minimize the rates of soil erosion in the uplands has been considered as an effective and cost-efficient technique in mitigating soil loss (Rawlins, Aggabao et al. 2017). Aside from being fiscally sound, the rehabilitation and the protection of the natural environment promote greater sustainability in supporting various soil ecological services (see **Chapter 2**). In this section, two cost estimates, namely conservation development value and maintenance costs, would be discussed and evaluated. Given that soil conservation (erosion mitigation) measures have always been cheaper than remediation, this estimate could be understood as being the lower bound of soil's replacement cost value.

As shown in earlier figures (**Figure 7-4** and **7-5**), portions of Angat's forest cover has shown stripped and converted into various forms of land cover. The region's unchecked urban sprawl, emerging agricultural and mining operations, and growing local populations have created enormous environmental pressures that have intensified the erosion rates in the region.



	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Disposal Costs	156	183	220	254	286	324	356	389	433	488	543	588	668	729	791	864	928	989	1064	1111	1165	1239	1338	1436	1533	1627
Excavation Costs	902.	1057	1269	1465	1650	1871	2058	2244	2499	2817	3136	3396	3860	4213	4571	4994	5359	5716	6149	6419	6730	7160	7731	8298	8856	9398

Figure 77-7. Change in costs from excavation and disposal costs from sedimentation in the Angat Reservoir

This was observed in the analysis of land cover change and confirmed by the various site visits conducted from December 2015 to March 2016. To prevent further soil degradation, mitigation measures have to be put in place and will have to maintain over time. But estimating the cost of soil erosion control measures can vary dramatically primarily from variations in project design. In order to simplify the calculations but still provide a logical project estimate for erosion control, two of the most commonly used mitigation techniques in the Philippines were used as a basis: reforestation of denuded forest cover, and cocomat installation to areas prone to landslides (Rawlins, Aggabao et al. 2017). To estimate the areas that would need erosion measures, the land cover classification was used to classify the region into four main categories:

(1) Areas of extreme concern – degraded forests with a very steep slope (slope greater than 45 degrees)
(2) Areas of concern –degraded forests located in flat to steep slope areas (slope less than 45 degrees)
(3) Agricultural and urban areas – parts of the watershed have been converted to disposable lands and have been used mainly for agricultural production; and,
(4) Forest lands (including closed forests and open forests).

The summary of computations is presented in **Table 7-5**. Under this proposed design, areas under Category 1 would require the installation of cocomats to serve as the immediate protection from landslides and extreme erosion rates. Cocomat is a form of erosion blanket made from coconut fiber which has been used in the country for steeply sloped areas needing immediate erosion control (Cereno 2009). The cost estimate used for cocomat installation follows its current market price of ₱1900/20m^2 or ₱950k/hectare. For areas under categories 2 and 3, reforestation would be undertaken that will not only reduce soil loss but will also promote the long term viability of a number of soil functions. The cost estimate used here is based on the reforestation expenditure used by the ABS-CBN Bantay Kalikasan Foundation, which is a private foundation towards the reforestation of Philippine forests (Vista, Cororaton et al. 2016). Note that this value considers the required supervision costs for reforested regions for at least the next three years (Luna 2016). Combining the reforestation and cocomat installation costs would yield the estimated soil conservation development value, which was computed to be about ₱484.1 million. Note that this data should be viewed with combined caution and urgency, given that further degradation of forest cover particularly in landslide prone areas would require further use of cocomats to provide immediate erosion control. Since the use of protective coconets is 12x more expensive than replanting the area, further need for immediate remediation will exponentially increase costs. For example, a 100-hectare increase in cocomat-use would increase the total cost by 18%, while a 500-hectare rise in cocomat usage will propel the costs by 90%. This detail should be taken more closely in forming the medium- and long-term soil conservation and watershed planning.

Table 7-5. Cost estimates (in PhP) for the soil rehabilitation and management of Angat Watershed

	Description	Area (in has)	Reforest'n Costs (in '000)	Coco Mat Install (in '000)	Watershed Management (in '000)
I	Deforested lands with very steep slope (>45deg)	292.11		277,504,500.00	156.28
II	Deforested lands flat to steep (<45 deg)	2,648.14	206,554.92		1,416.75
III	Agricultural/ peri-urban/ urban	364.64			195.08
IV	Forest lands	48,749.22			26,080.83
		Total	₱484,059,420.00		₱27,848,948.85

- *Reforestation cost used by ABS-CBN Bantay Kalikasan Foundation is P78,000/hectare*
- *Current cocomat market price is P950,000/hectare (October 2018)*
- *Yearly cost to ensure protection and preservation of land cover estimated to be P535/hectare*

The maintenance costs, on the other hand, reflect the value needed to ensure the infrastructure developed for soil conservation will be maintained, and that environmental policies and laws are strictly enforced. This estimate is strikingly much smaller than the conservation development value because this cost must be allocated on a yearly basis. For the computations, the municipal environmental and agriculture officers of Bulacan were surveyed with the help of the Provincial Planning Department to determine a suitable value for maintaining and protecting the upper watershed. The computations revealed that the annual cost would be about ₱27.8 million pesos. This value represents the maintenance of the status quo, and does not include rehabilitation or soil erosion control. This price tag should be taken as a reminder that soil conservation requires constant maintenance, and on the long run can dramatically decrease potential future costs from degradation.

7.6. Conclusion

Under the replacement cost method, the value of soil is estimated based on the costs associated by the mitigation efforts used to substitute amenities or services lost. For the value to be applicable, two important assumptions must be met: the alternative to the soil benefit provides the same level of service and costs much cheaper; and, the community is prepared to spend money to replace the services if it were to be lost. In this study, two alternative mitigation strategies were evaluated. Between dredging and reforestation alternatives, reforestation provides a more cost-efficient and sustainable technique to reduce reservoir sedimentation, which is only 7% of the dredging costs even with the inclusion of coco-fiber nets in high risk areas. This strengthens the argument for comprehensive reforestation efforts to reduce soil loss and sediment yield would be the more cost-efficient and sustainable means. Other soil functions, such as carbon sequestration and supporting biodiversity. would be better protected through reforestation compared to the more costly dredging option. And while the Angat Watershed still can be considered as a healthy watershed given that it still has 95% of its area covered with forests and that the Angat Dam is far from requiring dredging for its continued operations, valuation studies like this provide a clearer understanding of the potential costs if degradation of soil resources are allowed to continue. It would therefore be beneficial for utilities and watershed administration to incorporate soil services modelling and valuation in the short- and long-term planning.

Chapter 8. **Synthesis**

This study investigated the economic value of soil from various stakeholder perspectives by utilizing different non-market based approaches and a number of value types to encapsulate the worth of soil amenities. Throughout the research, potential challenges and areas of growth for the development and progress of soil valuation. This chapter is mainly divided into two section. The first part offers a succinct discussion on the primary issues in soil valuation that were revealed in the previous chapters. The second half examines the various applications and potential usage of soil valuation particularly in the decision-making process.

8.1. Research Findings

The results of this soil valuation study have yielded some methodological, empirical and policy-related implications. These include: (1) the emphasis on the suitability of valuation framework to match the needs of the study rather than a push for a universal structure; (2) the importance of matching valuation techniques with the intended value type and the contexts of the study area; (3) new typology of soil pricing mechanisms; (4) the critical role of stakeholder participation in the valuation process and soil use management; and (5) the expanding part of spatial data and modeling in the valuation process.

8.1.1. Unified Valuation Framework

Over the last decade, a number of soil valuation frameworks have been proposed to provide a system of value assessment for soil resources. Some of them have recommended focusing only on final marketable goods which they argued as the most logical and most closely aligned with conventional economic principles. Some frameworks have explored the inclusion of intermediary soil services in the valuation framework, to underscore the comprehensive nature of soil amenities and to correct the undervaluation of environmental goods. In particular, the total economic value (TEV) of environmental systems has been one of the focal aims in the growing science of environmental economics. The TEV combines the financial effects of the more easily perceived direct utility values of ecosystems, with the regulating and supporting services that have often been overlooked.

For soils, the TEV can be very useful for the long-term sustainable use of soil resources given that, in most instances, the indirect soil amenities often take a backseat against the more prominent soil production functions. However, the experiences from this study suggest that conducting such an undertaking may not always be as useful as valuing particular soil service or amenity for a specific group(s) of stakeholder. The various components of the TEV may require comprehensive analysis which includes the use of intensive data gathering and laborious field work. Theoretically, TEV could be a potential 'unifying' framework to establish soil value; but pragmatically speaking, its implementation requirements make the framework too exhaustive which greatly diminish its feasibility.

To answer the question of whether it is recommended to have a unified framework for soil valuation for all scenarios, the answer is no – at least not for all scenarios. Although a unified framework would indeed make a comparison of values more straightforward and greatly support benefit-transfer analyses, valuation specifications, including the framework, should conform to the intended goals of the study and to the study area. Similar to the valuation of other environmental public goods, soil valuation is complex and dynamic, and its applications are wide-ranging. Different soil valuation studies are used for a variety of purposes, which would entail the use of different types of economic values and derived using various valuation techniques. Promoting a single valuation framework to encompass all these variations might be counter-productive to the cause of soil conservation espoused in the valuation of soil resources.

8.1.2. **Framework and Techniques**

Deciding on which framework and approaches to implement is a crucial decision to be made at the beginning of any soil valuation study. At the start of this research, the design specifications initially included implementing all the three main valuation frameworks (fund-flow, TEV and cost-based), using various valuation approaches. However, after spending a year in the study area to conduct preliminary analysis and initiate stakeholder engagement, the proposal to implement a complete TEV assessment was deemed as not logistically practicable and is not necessarily essential. Not only would implementing such an undertaking be laborious, but the final result would also not necessarily befit the context of the study site. On this regard, I found the fund-and-flow framework to be well-suited to the needs of this particular research, and can be quite versatile to a variety of settings and study objectives.

As for the valuation methods, the six valuation approaches presented in this study provided reasonable value estimates for soil; four studies were excluded due to various reasons. Hedonic pricing was unsuitable for this study site because the agricultural property market wasn't mature enough to include soil (fertility) parameters in land valuation. Market pricing was also excluded because it failed to isolate the contribution of soil to the value of final marketable goods. The conjoint analysis was not included because the results showed that the respondents found the survey too complex that they were unable to articulate their preference and attitude. While the research does not conclude which of the various valuation techniques is most suitable, what was evident was that suitability of valuation method is highly dependent on the study area, spatial scale, stakeholder cognition, and the specific objectives of the valuation.

8.1.3. **New Typology of Soil Value**

Based on the notion that the valuation framework and techniques are highly dependent on research objectives, this research is proposing new typologies of soil value, other than the TEV. Often, a specific value type focusing on particular expenditure is more useful in soil use planning and policy development. Three value types that could be suitable for future soil-related valuation include: **conservation value**, **infrastructure value**, and **damage value**.

Conservation value reflects the cost associated with the protection for the continued utility of soil services. A number of valuation techniques can be employed to estimate soil conservation value. Using CVM or DCE, the stakeholders' WTP for soil protection can be used as the proxy value. Socio-demographic attributes of stakeholders, such as education, environmental consciousness, and income levels were found to have major influence on this value estimate.

Infrastructure value reflects the cost needed to execute a construction development that will prevent adverse effects of soil degradation. The costing is usually elicited either from a group of experts providing cost estimates or from actual expenditure from an already completed project. From the household production function, the value can be estimated implicitly based on farmers' defense expenditure. Aside from demographic characteristics, spatial features such as erosion risk levels and proximity to amenities could have significant effects on this estimation.

Damage value reflects the cost of damage control, or substitution, resulting from soil degradation. Using stated preference techniques, the WTP for damage control can be assessed from the various stakeholder groups, which includes private individuals and business enterprises. In this study, the damage value was based on normalized dredging operations costs and hypothetical rehabilitation costs and estimated using the replacement cost method.

8.1.4. **Stakeholder Participation**

The role of stakeholders is a significant component in the valuation process. The participatory approach focuses on a joint-decision making, wherein the primary stakeholders are knowledgeable about the problem and are willing to take part in the analysis. It has become a vital principle on development projects, with the support coming from different groups. Unlike conventional valuation methods where the whole process of decision-making is left to the discretion of experts, participation considers the stakeholders as collaborators, and they are given a significant say in the decision-making process. Dialogues between experts, policymakers, and primary stakeholders lead to an exchange of knowledge and experience, needed to analyze critical issues and to formalize solutions. Participation can be a tool to empower communities—empowering people to overcome challenges and influencing the community to take control of their lives are inherent to the participation process.

In integrating participation in the valuation framework, the first concern would be as to which conceptual approach would be most applicable and useful to which valuation framework. The different appreciation of environmental information by experts as compared with the public is an issue. When experts share environmental information, it may be assumed that they share a scientific worldview and that they know how to evaluate the quality of information, and they know how to assess them. Since stakeholders come from varying background and have different levels of competence, the one-size-fits-all policy cannot be used to elicit public participation.

Two important considerations in integrating public participation soil valuation include: (1) who are the relevant stakeholders, and (2) how to engage them properly. Determining who the relevant participants and acknowledging which stakeholder groups were to be considered were critical in the planning stage of the research. The difference in perspective, behavior, and cognition between stakeholder groups was at times stark and divergent. Upland farmers valued soil conservation much differently from farmers in the lowlands. Landowners viewed communal conservation fees more positively than land-leasing farmers, while farmers with longer agricultural experience were more likely to accept imposed conservation fees. Participants did not have homogenous perspectives but instead showed multiple identities, with different preferences and mindsets.

Another important issue is how to properly engage the stakeholders in valuing soil resources. Gaining the trust of the stakeholders while establishing neutrality of the researcher is an essential component that would allow free flowing discussion, elicitation of meaning responses and correct estimation of soil value. In this study, the objectives, delimitations, and constraints were properly disclosed with the stakeholders. Understanding the community dynamics and its organization was a useful starting point for planning the survey design and initializing engagement. It was essential that the facilitators were seen as being objective and impartial, and at the same time viewed with some level of authority that the respondents were willing to trust. The agriculture office, which has the primary responsibility of coordinating and developing projects with the local farmers, had been solicited to be a main research partner from the start. At the onset, this provided credence and rapport with the community.

8.1.5. Spatial Models and ES Value Mapping

The development of spatial technologies has provided a valuable advantage in environmental assessment and monitoring. Environmental attributes and economic value commonly exhibit spatial dependency. In this study, the use of spatial data provided additional perspectives on services, environmental risks, and determinants of economic value. Spatial attributes and environmental risk factors were shown to significantly impact the formation of stakeholder cognition and preference. The inclusion of spatial factors in economic valuation was especially critical in aggregating individual values and in explaining preference heterogeneity. Geographic-dependent information such as changes in LULC and population migration patterns, have been shown to help contextualize changes in soil ecosystem services affecting economic value.

Another group of parameters that were found to be very useful in soil valuation was the attributes relating to soil quality. Early frameworks lacked the flexibility to adequately differentiate the quality of soil benefits by simplifying soil as a homogenous entity. With additional perspectives from soil scientists, new valuation frameworks have characterized soil as a natural capital described through its biological, chemical and physical properties. However, data limitations, particularly the absence of up-to-date soil maps, is one of the biggest limitations hindering the further use of soil parameters in valuation. From my experience in gathering soil data in the Philippines for valuation purposes, while relevant government institutions were willing to assist in providing the necessary soil data, most of them were not up-to-date and provided only minimal information. With semi-detailed soil maps (1:50,000) produced in the 1980s and 1990s and detailed soil maps (1:10,000) produced in the 1970s, the current spatial and temporal resolution of Philippine soil data is very limited. Upgrading the soil map inventory is crucial in conducting a comprehensive valuation of soil resources and in assessing land use management and policy alternatives. Soil resource audits should also be updated to match growing applications specifically for soil valuation. Since assessment of soil value is undertaken at varying scales, soil data inventories would need to be modernized at the local, regional and national scales.

8.2. Applications of Valuation in decision making

Soil valuation provides an avenue to investigate a wide array of soil initiatives and policy alternatives that would reflect a socially optimal choice. Some potential applications to soil valuation include Payment for Ecosystem Services (PES), modifications to current property rights and tenure systems, and supporting sustainability goals.

8.2.1. Payment for Ecosystem Services (PES)

This research has shown people’s willingness to pay some into community funding towards soil conservation, to promote and protect soil amenities. This could be further extended through projects such Payment-for-Ecosystem-Services where governments can help promote soil conservation by financially rewarding private land-owners who practice sustainable agricultural practices and implement soil conservation measures. Payment for ecosystem services (PES) has gained much attention as an economic tool for promoting natural resource management recently. It consists of market-based policy strategies that promote the adoption of environmentally sustainable production practices through supportive and restrictive economic incentives. Economic valuation of soil would be useful in the development, assessment, and implementation of PES schemes. Valuation can be used to establish prices for PES to tackle the economic externalities of resource extraction and commodity production, enhancing both social and ecological conclusions. Through a comprehensive valuation of soil services, the externalities of environmental use would be represented in the PES scheme for the benefit of both the service beneficiaries and the payment recipients. Complexities related to uncertainty, power relations, distributional matters, and social embeddedness must be accounted to contextualize the variety of institutional settings in which PES operate. By integrating a more participative and comprehensive valuation process in the PES system, some of the institutional gaps and methodological challenges limiting PES use can be addressed.

8.2.2. Modifications to Property Rights

Soil valuation has accentuated the urgency to modify the current systems of property rights. Results from this research have shown that people's perception on soil value and on the use of sustainable farming practices and soil conservation measures are heavily tied on land tenure and their property rights. Through economic valuation, the cost of soil services and disservices can be thoroughly assessed which could then be developed into a more flexible form of property rights. This robust system would be able to commodify ecosystem services without having the need to privatize them. Diverse institutional arrangements can be expanded into various forms of property rights. Soil valuation can be used as a springboard for discussions among the many stakeholder groups to negotiate and adjust project structures to balance some of their diverging needs. This prevents environmental conflicts conceived as the competition for ecological distribution, such as access to natural resources and ecosystem services.

8.2.3. Supporting Sustainability Goals

One of the most tangible applications of soil valuation with global and local implications is its possible use towards fulfilling the Sustainability Development Goals (SDG) Adopted by the United Nations General Assembly in 2015, these Sustainability Goals have become an international social agenda with the main purpose of ending poverty, conserving the environment and ensuring prosperity gains are shared to all. Seventeen development goals have been adopted, with each goal having its specific targets to be realized within the next 15 years.

The valuation of the environment has a distinctive potential to be used as a tool to empower local communities and national governments. Understanding the value and costs associated with soil policies and programs provide decision-makers the ability to weigh the short- and long-term effect of competing alternatives, particularly in being able to achieve sustainability goals. It is important to note, however, that soils have no direct linkage with most of the SDGs, but rather contribute to the general ecosystem services that provide various services needed by society (Keesstra, Bouma et al. 2016). Among them, seven goals could be positively affected by the emerging use of soil valuation (see **Figure 8-1**). In particular, two SDGs have very strong links with soil value and can be supported by the growth of soil valuation science: (#2) Zero Hunger and (#15) Life on Land.

Figure 8-1. UN Sustainable Development Goals. Soil valuation has direct and indirect connection to seven SDGs which in the figure is printed in color.

The 2nd SDG is aimed towards ending hunger, achieving food security and improved nutrition, and promoting sustainable agriculture. Two of its main targets include increasing productivity of small-scale food producers through secured and equal access to land, productive resources and inputs, markets and opportunities (2.3), and ensuring sustainability of food production systems and implementing more resilient farming techniques that increase production whilst progressively improving soil and land quality (2.4). Increased awareness of soil's economic worth incentivizes the use of sustainable farming techniques and conservation methods that can help promote long-term food security. Given the diverse effects of soil quality and soil health in different stages of agricultural production, soil valuation can provide specific or comprehensive estimates of the economic impact of a particular food policy or program. Moreover, given that some soil use can pit communal welfare against private gains, or short-term benefits against long-term usage, soil value provides decision-makers various perspectives which is crucial in the decision-making process.

The 15th SDG is geared towards sustainably manage forests, combat desertification, reverse the degradation of land resources, and stop the losses of biodiversity. Many of its specific targets are well-suited by the growth of soil valuation particularly those that are aimed in ensuring conservation of terrestrial ecosystems (15.1), promoting sustainable management of forests (15.2), restoration of land and soil (15.3), guaranteeing the conservation of mountain ecosystems (15.5), and integrating the values of ecosystem and biodiversity into government planning and policy-making (15.9). One of the primary goals of soil valuation is to decouple the notion of environmental degradation with economic gains – that it is possible to grow the economy while also protecting the environment. Understanding the stated value of soil, particularly on its indirect uses, provides a picture of how stakeholders perceive the worth of soil, which is very important in crafting soil policies and programs. Moreover, the growth of soil valuation into mainstream use propels the notion of soil from being merely an input of production to being viewed as a crucial ecosystem in itself. The next time a question of what the value of soil is, the answer would not be simply, 'whatever the market value for potted soil is', but a more nuanced response that takes into account the economic impact of various soil functions.

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Appendix A. Summary of agricultural inputs of production and differentiation of values between irrigated vs non-irrigated fields

		Mean	Std Error	Min	Max	F	Sig
Seedlings (kg/ha)	Total	69.15	1.27	33.02	133.35	0.033	0.857
	Irrigated	69.53	2.49	33.02	118.04		
	Non-irrigated	69.01	1.49	36.02	133.35		
Fertilizer (kg/ha)	Total	212.99	4.89	60.58	449.89	8.266	0.005
	Irrigated	235.62	9.28	74.89	449.89		
	Non-irrigated	204.59	5.60	60.58	434.85		
Pesticide (L+kg /ha)	Total	2.02	0.06	0.38	5.00	87.982	0.000
	Irrigated	2.81	0.10	1.25	5.00		
	Non-irrigated	1.72	0.06	0.38	3.86		
Labor (md/ha)	Total	56.19	0.95	13.00	92.48	3.309	0.071
	Irrigated	59.01	1.54	33.35	92.48		
	Non-irrigated	55.14	1.16	13.00	90.02		

Appendix B. Gross Income as Response Variable and Socio-economic demographics as explanatory variables

	Gross Output (PhP/ha)	N	Std Dev	F	p-value
Size of Farm (interval)				8.223	0.005
Education (categorical)				0.233	0.873
w/o High School Diploma	70662.07	95	20733.88		
Graduated HS	70619.16	44	22077.15		
Technical School	71152.59	30	16402.30		
College Degree	75801.32	12	21667.14		
Farming Experience (ordinal)				11.655	0.000
<10 years	62711.63	20	15499.77		
11-20 years	59944.54	32	13422.55		
21-30 years	64256.89	31	15504.38		
>30 years	78570.53	98	21528.30		
Land Ownership Type (categorical)				3.421	0.019
Owned through Patent/AR	76578.55	45	24894.60		
Owned through Purchase	65996.28	48	15979.85		
Owned through Inheritance	66417.61	39	16532.52		
Rent / Lease	74697.76	49	20857.25		
Farm Ecosystem (categorical)				6.293	0.013
Irrigated	77215.35	49	22196.63		
Non-irrigated	68793.80	132	19230.31		
Terrain (ordinal)				1.190	0.307
Gentle (<8% slope)	68554.97	79	18052.44		
Moderate (8-30% slope)	73618.85	74	23149.06		
Rolling/ Hilly (>30% slope)	71453.44	28	18363.41		
Receiving Govt Assistance (dummy)				0.962	0.328
Yes	71940.46	135	20477.39		
No	68529.80	46	20021.96		

Farmgate Price for Rice used in calculation: PhP 18/kilogram

$1 (in April 2015) = PhP 44.63

Appendix C. Parameter estimates for model 1 using only the inputs of production (seedling, fertilizer, pesticide, and labor) as explanatory variables, and agricultural yield as response variable

Parameter		Estimate	Std. Error	95% Confidence Interval Lower	Upper
α	Intercept	9.002	11.100	-12.924	30.928
β_1	Ln Seedlings	1.237	2.471	-3.643	6.118
β_2	Ln Fertilizer	-1.839	1.890	-5.574	1.895
β_3	Ln Pesticide	0.761	2.194	-3.574	5.095
β_4	Ln Labor	0.842	2.371	-3.842	5.526
β_{11}	Ln Seedlings x Ln Seedlings	0.379	0.434	-.479	1.237
β_{12}	Ln Seedlings x Ln Fertilizer	-0.247	0.507	-1.249	0.755
β_{13}	Ln Seedlings x Ln Pesticide	0.575	0.597	-.604	1.754
β_{14}	Ln Seedlings x Ln Labor	-1.086	0.647	-2.364	0.191
β_{22}	Ln Fertilizer x Ln Fertilizer	0.585	0.216	0.159	1.011
β_{23}	Ln Fertilizer x Ln Pesticide	-0.974	0.483	-1.929	-0.019
β_{24}	Ln Fertilizer x Ln Labor	0.051	0.577	-1.088	1.190
β_{33}	Ln Pesticde x Ln Pesticide	0.068	0.394	-0.710	0.845
β_{34}	Ln Pesticide x Ln Labor	0.264	0.556	-0.834	1.362
β_{44}	Ln Labor x Ln Labor	0.386	0.253	-0.114	0.885

Pseudo R^2 (1 - Residual Sum of Squares / (Corrected sum of squares)) = **0.356**
Constraints:
$\beta_1 + \beta_2 + \beta_3 + \beta_4 = 1$
$\beta_{11} + \beta_{11} + \beta_{12} + \beta_{13} + \beta_{14} = 0$
$\beta_{22} + \beta_{22} + \beta_{12} + \beta_{23} + \beta_{24} = 0$
$B_{33} + B_{33} + B_{13} + B_{23} + B_{34} = 0$; $B_{44} + B_{44} + B_{14} + B_{24} + B_{34} = 0$

Appendix D. Parameter estimates for Model2 using inputs of production and socio-demographic attributes as explanatory variables, and the agricultural yield as response variable

Parameter		Estimate	Std. Error	95% Conf Interval	
				Lower	Upper
α	Intercept	7.827	10.718	-13.344	28.997
β_1	Ln Seedlings	1.051	2.402	-3.694	5.797
β_2	Ln Fertilizer	-1.481	1.855	-5.145	2.183
β_3	Ln Pesticide	0.243	2.176	-4.054	4.541
β_4	Ln Labor	1.186	2.319	-3.394	5.766
β_{11}	Ln Seedlings x Ln Seedlings	0.428	0.418	-0.397	1.254
β_{12}	Ln Seedlings x Ln Fertilizer	-0.155	0.480	-1.102	0.792
β_{13}	Ln Seedlings x Ln Pesticide	0.516	0.570	-0.610	1.643
β_{14}	Ln Seedlings x Ln Labor	-1.218	0.629	-2.461	0.024
β_{22}	Ln Fertilizer x Ln Fertilizer	0.517	0.216	0.090	0.944
β_{23}	Ln Fertilizer x Ln Pesticide	-0.778	0.477	-1.721	0.164
β_{24}	Ln Fertilizer x Ln Labor	-0.101	0.550	-1.188	0.986
β_{33}	Ln Pesticde x Ln Pesticide	-0.061	0.379	-0.810	0.689
β_{34}	Ln Pesticide x Ln Labor	0.383	0.541	-0.686	1.452
β_{44}	Ln Labor x Ln Labor	0.468	0.245	-0.017	0.953
γ_1	Educational Attainment	7.827	10.718	-13.344	28.997
γ_2	Farming Experience	1.051	2.402	-3.694	5.797
γ_3	Ownership Type	-1.481	1.855	-5.145	2.183
γ_4	Farm Ecosystem	0.243	2.176	-4.054	4.541
γ_5	Government Assistance	1.186	2.319	-3.394	5.766
γ_6	Terrain	0.428	0.418	-0.397	1.254
γ_7	Government Support	-0.155	0.480	-1.102	0.792

Pseudo R^2 (1 - Residual Sum of Squares / (Corrected sum of squares)) = **0.444**
Constraints: $\beta 1 + \beta 2 + \beta 3 + \beta 4 = 1$ || $\beta 11 + \beta 11 + \beta 12 + \beta 13 + \beta 14 = 0$ || $\beta 22 + \beta 22 + \beta 12 + \beta 23 + \beta 24 = 0$ ||
$B33 + B33 + B_{13} + B_{23} + B_{34} = 0$; $B_{44} + B_{44} + B_{14} + B_{24} + B_{34} = 0$

Appendix E. Parameter estimates for Model3 using inputs of production, socio-demographic attributes and environmental consciousness score and agricultural yield as response variable

Parameter		Estimate	Std. Error	95% Confidence Interval	
				Lower	Upper
α	Intercept	7.801	10.258	-12.460	28.063
β_1	Ln Seedlings Cost	0.984	2.306	-3.570	5.539
β_2	Ln Fertilizer Cost	-1.446	1.761	-4.926	2.033
β_3	Ln Pesticide Cost	0.750	2.087	-3.372	4.871
β_4	Ln Labor Cost	0.713	2.215	-3.663	5.088
β_{11}	Ln Seedlings x Ln Seedlings	0.548	0.398	-.237	1.334
β_{12}	Ln Seedlings x Ln Fertilizer	-0.352	0.457	-1.256	0.551
β_{13}	Ln Seedlings x Ln Pesticide	0.404	0.547	-0.677	1.485
β_{14}	Ln Seedlings x Ln Labor	-1.149	0.607	-2.348	0.051
β_{22}	Ln Fertilizer x Ln Fertilizer	0.525	0.205	0.120	0.931
β_{23}	Ln Fertilizer x Ln Pesticide	-0.773	0.453	-1.669	0.122
β_{24}	Ln Fertilizer x Ln Labor	0.075	0.524	-0.961	1.111
β_{33}	Ln Pesticde x Ln Pesticide	0.105	0.361	-0.608	0.819
β_{34}	Ln Pesticide x Ln Labor	0.158	0.525	-0.878	1.195
β_{44}	Ln Labor x Ln Labor	0.458	0.234	-0.005	0.920
γ_1	Educational Attainment	0.010	0.019	-0.027	0.047
γ_2	Farming Experience	0.039	0.018	0.004	0.074
γ_3	Ownership Type	-0.003	0.016	-0.034	0.029
γ_4	Farm Ecosystem	-0.066	0.054	-0.172	0.040
γ_5	Government Assistance	0.067	0.021	0.026	0.108
γ_6	Terrain	-0.024	0.026	-0.075	0.026
γ_7	Government Support	-0.029	0.042	-0.111	0.053
δ_1	Ln Conservation Cost	0.113	0.031	0.052	0.173
δ_2	Erosion Vulnerability	-0.033	0.013	-0.058	-0.007
δ_3	Ln EAS	0.092	0.106	-0.117	0.301

Pseudo R^2 (1 - Residual Sum of Squares / (Corrected sum of squares)) = **0.501**
Constraints: $\beta_1 + \beta_2 + \beta_3 + \beta_4 = 1$ || $\beta_{11} + \beta_{11} + \beta_{12} + \beta_{13} + \beta_{14} = 0$ || $\beta_{22} + \beta_{22} + \beta_{12} + \beta_{23} + \beta_{24} = 0$ ||
$B_{33} + B_{33} + B_{13} + B_{23} + B_{34} = 0$; $B_{44} + B_{44} + B_{14} + B_{24} + B_{34} = 0$

Appendix F. Summary of conservation expenditure

	Conservation Expenditure (PhP/ha)	N	Std Error.	F	p-value
Size of Farm (interval)				14.269	0.000
Education (categorical)				0.316	0.813
w/o High School Diploma	2378.28	95	162.22		
Graduated HS	2262.27	44	234.29		
Technical School	2194.42	30	245.85		
College Degree	2643.99	12	262.60		
Farming Experience (ordinal)				5.863	0.016
<10 years	2053.96	20	212.20		
11-20 years	1858.11	32	211.31		
21-30 years	2215.83	31	194.76		
>30 years	2589.87	98	174.77		
Land Ownership Type (categorical)				0.604	0.613
Owned through Patent/AR	2613.72	45	305.52		
Owned through Purchase	2202.28	48	236.83		
Owned through Inheritance	2233.98	39	211.23		
Rent / Lease	2336.08	49	162.60		
Farm Ecosystem (categorical)				20.960	0.000
Irrigated	1544.61	49	122.98		
Non-irrigated	2631.45	132	137.08		
Terrain (ordinal)				0.194	0.660
Gentle (<8% slope)	2340.02	79	146.71		
Moderate (8-30% slope)	2245.26	74	177.20		
Rolling/ Hilly (>30% slope)	2572.37	28	360.54		
Receiving Govt Assistance (dummy)				0.016	0.898
Yes	2328.90	135	134.64		
No	2361.65	46	190.00		

>> $1 (in April 2015) = PhP 44.63

Appendix G. Survey Questionnaire used in PC-CVM

The translated language of the CVM questionnaire are as follows:

The Municipal Government of Norzagaray through the Agriculture Office is creating a community fund that will be used to finance soil conservation measures catered to mitigating erosion in private farmlands.

A. Voluntary Payment System:

1. Would you be willing to participate/contribute soil conservation measure if it was going to be on a voluntary basis? ☐yes ☐no
2. If the community-initiated fund is to be set-up aimed at assisting farmers and farm-workers with soil conservation and rehabilitation, and it is voluntary, how much will you be willing to contribute annually?
   ☐0 ☐25 ☐50 ☐75 ☐100 ☐125 ☐150 ☐175 ☐200

B. Environmental Awareness Test: Please state if you agree or disagree with each of the following statement:

1. I consider soil protection as an essential consideration in farming.

☐strongly disagree ☐ disagree ☐neutral ☐agree ☐strongly agree

2. I deliberately allocate substantial time and money towards soil conservation measures

☐strongly disagree ☐ disagree ☐neutral ☐agree ☐strongly agree

3. I regularly seek training/consultation on soil use & conservation methods.

☐strongly disagree ☐ disagree ☐neutral ☐agree ☐strongly agree

4. Local government has the responsibility and authority to enforce measures that will protect soil resources in the community.

☐strongly disagree ☐ disagree ☐neutral ☐ agree ☐ strongly agree

5. I am agreeable to community-based regulations and ordinance that will promote soil conservation, which would include the imposition of penalties for non-compliance.

☐strongly disagree ☐ disagree ☐neutral ☐ agree ☐ strongly agree

6. I am amenable to the collection of additional fees that will supplement the budget towards community-level soil conservation measures.

☐strongly disagree ☐ disagree ☐neutral ☐ agree ☐ strongly agree

C. Compulsory Payment System:

1. If it was decided that a mandatory fee would be imposed, and each land-holding household will be taxed ______ amount annually, would you be willing to accept? ☐yes ☐no
2. If you answered YES to the previous question, and the amount was raised by ₱25, would you be willing to accept such plan? If you answered NO and the amount was lowered by ₱25, would you be willing to accept the plan? ☐yes ☐no

Appendix H. Part of Survey Questionnaire used in DCE

Self-Evaluation: Please state if you agree or disagree with each of the following statement

1. I am aware of the different indirect functions of soil such as its regulating and supporting functions.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

2. I personally enjoy the different benefits of protecting the watershed's soil from degradation.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

3. I consider it a personal obligation to contribute in protecting the watershed from soil degradation.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

4. Asking the residents to pay a community-agreed amount for the watershed's conservation measures is acceptable.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

5. Protecting the upper watershed from soil degradation is important for the sake of the future generation.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

6. I feel some degree of fulfillment when I do my part in the watershed's soil conservation.

☐strongly disagree ☐disagree ☐neutral ☐ agree ☐ strongly agree

The Norzagaray part of the Angat Watershed provides a multitude of direct and indirect benefits to our town. In support of the current efforts to preserve and rehabilitate protected areas of the watershed, a plan is proposed to implement an environmental support fund.

1. Would you be willing to contribute to the environmental support fund?

☐yes ☐no

2. Do you prefer that the fund be:

☐voluntary ☐mandatory and fixed ☐mandatory and progressive

3. If the environmental support fund was to be set-up, and each household would be as asked to pay an annual fee, how much do you think should the amount be set?

☐0 ☐25 ☐50 ☐75 ☐100 ☐125 ☐150 ☐175 ☐200

Appendix I. Summary of mean WTP for landslide-groupings

Group-A1 includes all points categorized with 'low landslide risk' (C_L1), 'moderate landslide risk' (C_L2), and 'high landslide risk' (C_L3). Group-A2 includes all points classified as 'no landslide risk' zones (C_L0).

	Group-A1 (C_L1, C_L2, C_L3)				Group-A2 (C_L0)			
Attribute	Estimate (Std Err)	CI [L95%, U95%]	MWTP	Std Error	Estimate (Std Err)	CI [L95%, U95%]	MWTP	Std Error
Water (L2)	2.25 (0.191)	[1.886, 2.634]	78.609	5.948	2.851 (0.34)	[2.214, 3.556]	67.617	6.055
Water (L3)	2.476 (0.218)	[2.06, 2.917]	165.11	7.585	3.163 (0.37)	[2.469, 3.931]	142.62	7.038
Erosion (L2)	1.067 (0.195)	[0.691, 1.455]	37.293	5.962	2.714 (0.369)	[2.021, 3.479]	64.351	5.914
Erosion (L3)	1.961 (0.18)	[1.614, 2.321]	105.79	6.489	1.662 (0.284)	[1.118, 2.236]	103.75	7.081
Carbon (L2)	1.427 (0.164)	[1.112, 1.755]	49.841	5.191	1.678 (0.248)	[1.207, 2.18]	39.782	5.140
Carbon (L3)	1.036 (0.216)	[0.617, 1.467]	86.031	7.056	1.206 (0.339)	[0.557, 1.894]	68.392	7.436

Appendix J. Summary of mean WTP for erosion groupings

Group-B1 includes all points categorized with 'low erosion risk' (C_E1), 'moderate erosion risk' (C_E2), 'high erosion risk' (C_E3), and 'very high erosion risk' (C_E4). Group-B2 includes all points with 'negligible to very low erosion risk' (C_E0)

	Group-B1 (C_E1, C_E2, C_E3, C_E4)				Group-B2 (C_E0)			
Attribute	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error
Water (L2)	2.243 (0.198)	[1.866, 2.643]	78.839	6.159	2.689 (0.299)	[2.123, 3.302]	69.665	69.665
Water (L3)	2.505 (0.225)	[2.077, 2.960]	166.88	8.065	2.836 (0.322)	[2.230, 3.496]	143.12	143.122
Erosion (L2)	1.381 (0.205)	[0.986, 1.792]	48.544	6.000	1.779 (0.300)	[1.206, 2.390]	46.082	46.082
Erosion (L3)	1.849 (0.184)	[1.495, 2.217]	113.53	6.968	1.846 (0.261)	[1.347, 2.372]	93.907	93.907
Carbon (L2)	1.333 (0.165)	[1.016, 1.663]	46.862	5.268	1.731 (0.237)	[1.282, 2.214]	44.835	44.835
Carbon (L3)	0.984 (0.221)	[0.556, 1.423]	81.431	7.224	1.175 (0.31)	[0.581, 1.798]	75.279	75.279

Appendix K. Summary of mean WTP for flood groupings.

Group-C1 includes all points categorized with 'low flood risk' (C_F1), 'moderate flood risk' (C_F2), and 'high flood risk' (C_F3). Group-C2 includes all points classified as 'no flood risk' zones (C_F0).

	Group-C1 (C_F1, C_F2, C_F3)				Group-C2 (C_F0)			
Attribute	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error
Water (L2)	2.924 (0.443)	[2.1, 3.861]	76.115	4.978	2.294 (0.178)	[1.953, 2.653]	75.041	9.433
Water (L3)	3.099 (0.485)	[2.203, 4.132]	156.800	6.213	2.524 (0.201)	[2.141, 2.928]	157.600	11.427
Erosion (L2)	2.239 (0.463)	[1.373, 3.214]	58.285	5.011	1.374 (0.182)	[1.023, 1.736]	44.941	8.567
Erosion (L3)	1.692 (0.365)	[0.999, 2.437]	102.34	5.595	1.882 (0.166)	[1.563, 2.213]	106.520	10.321
Carbon (L2)	1.667 (0.32)	[1.068, 2.327]	43.406	4.426	1.416 (0.149)	[1.128, 1.714]	46.311	7.422
Carbon (L3)	1.329 (0.446)	[0.486, 2.250]	77.994	5.945	0.969 (0.197)	[0.587, 1.36]	78.008	10.819

Appendix L. Mean WTP values for water zone groups

Group-D1 includes all points found within 1-kilometer distance from the nearest water system while Group-D2 includes all points beyond the 1-kilometer zone

	Group-D1 (within 1km)				Group-D2 (outside 1km)			
Attribute	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error
Water (L2)	2.329 (0.212)	[1.926, 2.758]	73.339	5.695	2.487 (0.265)	[1.985, 3.028]	77.453	6.903
Water (L3)	2.765 (0.244)	[2.302, 3.26]	160.41	7.126	2.405 (0.283)	[1.869, 2.982]	152.34	8.441
Erosion (L2)	1.464 (0.213)	[1.053, 1.892]	46.098	5.616	1.620 (0.278)	[1.09, 2.184]	50.461	6.936
Erosion (L3)	1.776 (0.193)	[1.404, 2.162]	102.01	6.265	1.955 (0.239)	[1.499, 2.438]	111.36	7.953
Carbon (L2)	1.383 (0.174)	[1.05, 1.733]	43.552	4.929	1.638 (0.216)	[1.226, 2.074]	51.001	6.051
Carbon (L3)	1.072 (0.234)	[0.621, 1.539]	77.308	6.827	0.955 (0.282)	[0.411, 1.518]	80.747	8.112

Appendix M. Mean WTP values for forest zone groups.

Group-E1 includes all points found within 1.5-kilometer distance from the protected forest region, while Group-E2 includes all points beyond the 1.5-kilometer forest zone.

	Group-E1 (within 1.5-km)				Group-E2 (outside 1.5km)			
Attribute	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error	Est (Std Err)	CI [L95%, U95%]	MWTP	Std Error
Water (L2)	2.592 (0.376)	[1.891, 3.38]	78.257	9.630	2.362 (0.185)	[2.007, 2.736]	74.039	4.920
Water (L3)	3.103 (0.436)	[2.296, 4.023]	171.94	12.988	2.524 (0.206)	[2.129, 2.94]	153.15	5.924
Erosion (L2)	2.039 (0.416)	[1.259, 2.907]	61.560	9.013	1.406 (0.186)	[1.046, 1.778]	44.066	4.954
Erosion (L3)	1.929 (0.334)	[1.299, 2.611]	119.81	11.055	1.845 (0.17)	[1.517, 2.184]	101.90	5.444
Carbon (L2)	1.304 (0.278)	[0.778, 1.865]	39.383	7.803	1.516 (0.155)	[1.219, 1.827]	47.522	4.336
Carbon (L3)	1.881 (0.452)	[1.031, 2.817]	96.179	10.968	0.846 (0.197)	[0.464, 1.237]	74.055	5.852

Summary

Soil is an essential resource with diverse ecological functions and socio-economic contributions. But due to abuse and mismanagement, coupled with the increasing demands from conflicting usage, soil resources have been under threat from being substantially degraded. To promote soil conservation and sustainable use, there has been growing interest to integrate economics into environmental policy making. But without an agreed-upon measure to evaluate the economic aspect of conservation and ecology, people have been less-accepting of regulating sustainability, especially when brought against maximizing profit. It is therefore imperative that a credible and comprehensive soil valuation process would be constructed that would provide realistic and normative value estimates of soil contributions.

Determining the economic worth of soil is complex and multi-faceted. Soils are one of the most complex Earth systems that are intrinsically connected with biodiversity, climate change, and the health of the broader environment. Its ecological functions and environmental services are often unrecognized and not well understood. As an economic resource, soil performs a variety of roles and functions. Aside from the multiple soil amenities directly benefiting private individuals, soil provides a broad range of public service to the broader community. Due to non-excludability and non-subtractability attributes, soil has no developed markets to determine the benefits derived by each household resulting in the underestimation of the soil's actual worth. There have been a number of frameworks and non-market based valuation techniques that have been proposed to provide an assessment of the soil's economic value. But while there has been growing literature proposing and developing conceptual frameworks for soil value estimation, these have remained largely hypothetical, with sparse real soil valuation studies other than those that valued soil services being part of a larger ecosystem. It is therefore critical to understand how actual valuation of soil can be implemented, which would entail the use of non-market based approaches and how these approaches relate to soil valuation frameworks.

This research has been focused on understanding soil value and the process of soil valuation, particularly on the growing role of stakeholder participation, spatial data, physical modelling, and pedometric attributes in estimating value. Different non-market valuation techniques were used to estimate soil value and to better understand significant attributes affecting economic worth. Various economic frameworks and value types relevant to soil were reviewed, and a critical discussion on the use and limitations of soil economic value was also provided. Aside from providing estimates of soil value, this research has yielded important methodological and policy-related implications, including (a) the emphasis on the objectives of the study rather than a push for a universal structure; (b) the importance of matching valuation techniques with the

intended value type and the contexts of the study area; (c) a proposal for new typology of soil value; (d) the critical role of stakeholder participation in the valuation process and soil use management; and (e) the expanding part of spatial data and modeling in the valuation process.

Deciding on which framework and approaches to implement is a crucial decision to be made at the beginning of any soil valuation study. Although a unified framework would indeed make a comparison of values more straightforward and greatly support benefit-transfer analyses, valuation specifications, including the framework, should conform to the intended goals of the study and to the study area. Similar to the valuation of other environmental public goods, soil valuation is complex and dynamic, and its applications are wide-ranging. Different soil valuation studies are used for a variety of purposes, which would entail the use of different types of economic values and derived using various valuation techniques. Promoting a single valuation framework to encompass all these variations might be counter-productive to the cause of soil conservation espoused in the valuation of soil resources.

As for the valuation methods, the six valuation approaches presented in this study provided reasonable value estimates for soil. While the research does not conclude which of the various valuation techniques is most suitable, what was evident was that suitability of valuation method is highly dependent on the study area, spatial scale, stakeholder cognition, and the specific objectives of the valuation. Based on the notion that the valuation framework and techniques are highly dependent on research objectives, this research is proposing new typologies of soil value, focusing on particular expenditure is more useful in soil use planning and policy development. Three value types that could be suitable for future soil-related valuation include: *conservation value*, *infrastructure value*, and *damage value*. Conservation value reflects the cost associated with the protection for the continued utility of soil services, which can be estimated using contingent valuation or choice experiments. Socio-demographic attributes of stakeholders, such as education, environmental consciousness, and income levels were found to have major influence on conservation value. Infrastructure value reflects the cost needed to execute a construction development that will prevent adverse effects of soil degradation, which can be estimated by analyzing the farmers' defense expenditure through production function. Aside from demographic characteristics, spatial features such as erosion risk levels and proximity to amenities could have significant effects on this estimation. Damage value reflects the cost of damage control, or substitution, resulting from soil degradation, which can be valued using normalized dredging operations costs and hypothetical rehabilitation costs from the replacement cost method.

The role of stakeholders was also proven to be a crucial component in the valuation process. Unlike conventional valuation methods where the whole process of decision-making is left to the discretion of experts, participation considers the stakeholders as collaborators, and they are given a significant say in the decision-making process. In integrating participation in the valuation framework, the first concern would be as to which conceptual approach would be most applicable and useful to which valuation framework. Since stakeholders come from varying background and have different levels of competence, the one-size-fits-all policy cannot be used to elicit public participation. Determining who the relevant participants and acknowledging which stakeholder groups were to be considered were critical in the planning stage of the research. The difference in perspective, behavior, and cognition between stakeholder groups was at times stark and divergent. Stakeholders do not have homogenous perspectives but instead exhibit multiple identities, with different preferences and mind-sets. Another important issue is how to properly engage the stakeholders. Gaining the trust of the stakeholders while establishing neutrality of the researcher is an essential component that would allow free flowing discussion, elicitation of meaning responses and correct estimation of soil value. Understanding the community dynamics and its organization can be a useful starting point for planning the survey design and initializing engagement.

The development of spatial technologies has provided a valuable advantage in environmental assessment and monitoring, given that environmental attributes and economic value commonly exhibit spatial dependency. Spatial attributes and environmental risk factors were shown to significantly impact the formation of stakeholder cognition and preference. The inclusion of spatial factors in economic valuation was especially critical in aggregating individual values and in explaining preference heterogeneity. Another group of parameters that were found to be very useful in soil valuation was the attributes relating to soil quality. However, data limitations, particularly the absence of up-to-date soil maps, is one of the biggest limitations hindering the further use of soil parameters in valuation. Upgrading the soil map inventory would therefore be crucial in conducting comprehensive valuation of soil resources, and in assessing land use management and policy alternatives. Soil resource audits should also be updated to match growing applications specifically for soil valuation. Since assessment of soil value is undertaken at varying scales, soil data inventories would need to be modernized at the local, regional and national scales.

Samenvatting

De bodem is een essentiële hulpbron en draagt bij aan diverse ecologische aan sociaal economische processen. Door verkeerd management gekoppeld aan een toenemende vraag voor bodem in conflicterende belangen, is deze natuurlijk hulpbron bedreigd en op veel plekken onderhevig aan degradatie. Om bodem conservering en duurzaam gebruik te stimuleren is er een groeiend interesse in het benaderen van de bodem vanuit een economisch perspectief bij het formuleren van milieubeleid. Echter, mede omdat er geen duidelijk afgesproken methodes zijn om de economische aspecten van bodem conservering te evalueren, wint de maximalisering van winst uit het gebruik van bodems het van duurzaam gebruik. Het is daarom belangrijk een methode te maken zodat het waarderen van de bodem op een geloofwaardige en duidelijke tot stand komt, waarbij dit bodemwaarderingsproces leidt tot realistische normatieve schattingen van de bijdrage van bodem aan duurzaamheid.

Het vaststellen van de economische waarde van bodem is complex en heeft vele facetten. Bodems vormen een van de meest complexe natuurlijke systemen die op intrinsieke wijze verbonden zijn met biodiversiteit, klimaat en klimaat verandering en de gezondheid van het milieu in bredere zin. De ecologische functies en 'diensten' worden vaak niet herkend of zijn slecht onderzocht. Als economische hulpbron heeft de bodem een veelzijdige rol. Behalve de directe voorzieningen waar een individuele gebruiker van profiteert, heeft de samenleving in bredere zin ook voordelen van bodems. Maar vanwege het feit een bepaald gebruik van bodem nooit exclusief is en het gebruik van de bodem de waarde niet verminderd voor een ander (in tegenstelling tot private goederen), heeft bodem geen goed ontwikkelde markt om de voordelen uit te drukken, bijv. per huishouden, zodat de werkelijke waarde onderschat wordt. Er bestaan diverse systemen die niet op een markt gebaseerd zijn om de economische waarde van de bodems te bepalen. Echter, dit zijn voornamelijk conceptuele raamwerken met maar een paar specifieke studies om de bodem zelf een waarde te geven, anders dan als onderdeel van een groter ecosysteem. Het is daarom belangrijk om te begrijpen hoe de eigenlijke waardering van bodems kan woorden geïmplementeerd, waarbij gebruik wordt gemaakt van een aanpak die niet op markt principes is gebaseerd.

Dit onderzoek richt zich op het begrijpen de waarde en het proces van waardering van bodem, in het bijzonder de rol van stakeholders, ruimtelijke informatie, fysisch modelleren en bodemkundige attributen in het schatten van de waarde. Verschillende, niet-markt gebaseerde, methoden zijn gebruikt om beter te kunnen begrijpen welke bodemeigenschappen bijdragen tot de economische waarde. De meest gebruikte methodes met hun voordelen en beperkingen zijn vergeleken, om te kijken welke het meest geschikt zou zijn.

De belangrijkste methodologische aspecten beleid gerelateerde bevindingen zijn: (a) het is niet mogelijk of gewenst een enkele universele methode te gebruiken, (b) het is belangrijk de methode aan te passen aan het type waarde dat men voor ogen heeft en aan een specifiek gebied, (c) er wordt een voorstel gedaan voor een nieuwe typologie van de waarde van bodem, (d) stakeholders hebben een essentiële rol in het proces van waardering, en (e) ruimtelijke informatie en modellering hebben een toegevoegde waarde in het proces van waardering.

De keuze van waarderingsmethode is belangrijk. Het bleek niet mogelijk een generiek best mogelijke methode te maken of kiezen. Een uniforme methode zou de onderlinge vergelijking van de waardering van bodems en gebieden vergemakkelijken, zijn de gebruikte methodes toch specifiek voor dit gebied en de context. Net als de analyse en waardering van ecosysteem functies en diensten is de waardering van bodems complex en dynamisch, met verrijkende toepassingen. Gezien de variatie en breedte in bodem waarderingsstudies, zou het risico bestaan dat een uniforme 'one size fits all' methode een verarming teweeg brengt en niet leidt tot het nodige niveau van bodemconservering.

Dit onderzoek richt zich op nieuwe typen voor bodemwaarde, waarbij er een direct verband gezocht wordt met beleidsontwikkeling voor duurzaam bodem gebruik en beheer. Zes methodes van waardering zijn vergeleken en allen geven een redelijke schatting van de waarde van de bodem. Geen van de methodes was duidelijk beter dan een andere, en het bleek dat allen erg afhankelijk zijn van het gebied, de schaal, de kennis van stakeholders en de specifieke doelen van de waardering. Drie types van bodem waardering zijn gevonden die het best functioneren: waarde van bodemconservering, infrastructuur waarde (zie hieronder)en schade door degradatie. Conserveringswaarde zijn de kosten die geassocieerd worden met bodembescherming en een onverminderd functioneren van de 'diensten' van bodems ('soil services'). Dit kan geschat worden met de methodes 'contigent valuation' of 'choice experiments'. Van belang bij de toepassing van deze technieken zijn vnl. het niveau van opleiding van de stakeholders, hun milieubewustzijn en inkomen. Infrastructuur waarde is hier gedefinieerd als de kosten van het aanleggen van infrastructuur om de negatieve effecten van bodemdegradatie op te vangen. Dit kan geanalyseerd worden met een 'production function' methode door te kijken naar de uitgaven van boeren voor bodemconservering. Dezelfde eigenschappen van stakeholders zoals hierboven genoemd zijn van belang, maar het maakt sterk uit of boeren ervaring hebben met erosie risico of de positieve aspecten van conservering ondervinden. Schade door degradatie zijn de kosten als gevolg van bodemdegradatie of van vervanging van infrastructuur, bijv. bagger- en schoonmaakkosten als gevolg van sedimentatie in waterbekkens en drainage systemen benedenstrooms.

De rol van stakeholders in de verschillende waarderingsmethoden is cruciaal. In tegenstelling tot methoden waarbij alle beslissingen door experts gedaan worden, is hier met stakeholders samengewerkt en hebben zij in de besluitvorming een belangrijke rol gekregen. Hierdoor was het belangrijk om de best toepasbare methode te kiezen. Omdat stakeholders verschillende achtergronden en niveaus van kennis hebben, bleek het niet mogelijk een enkele universele methode te gebruiken. Stakeholder groepen verschillen in gedrag, hebben een verschillende kijk op het probleem en denken daar anders over. Het is dus belangrijk om vertrouwen op te bouwen en neutraal te blijven, en geen oordeel te vellen over bepaalde uitkomsten, zodat er een vrije uitwisseling van ideeën kan plaatsvinden zodat de resultaten een goede reflectie zijn van de stakeholders.

De ontwikkeling van ruimtelijke technieken is een belangrijk voordeel gebleken omdat economische waarden en ecosysteem waarden ruimtelijk aan elkaar gerelateerd zijn. Het begrip en de voorkeuren van stakeholders blijkt sterk gerelateerd te zijn aan de ruimtelijke patronen van bodem erosie risico. Deze patronen laten zich vooral gebruiken om individuele stakeholder resultaten te aggregeren, en de heterogeniteit in voorkeuren te begrijpen. Ook bodem eigenschappen die direct gerelateerd zijn aan bodem kwaliteit zouden nuttig kunnen zijn, maar hier loopt men toch tegen de beschikbaarheid van bodem data aan, en het gebrek aan actuele bodemkaarten. Het verzamelen van aanvullende bodem informatie en het moderniseren van de bodemkaart kan belangrijk zijn voor verdere studies naar de waarde van bodems als natuurlijke hulpbron. Dit zou op verschillende schalen moeten gebeuren aangezien de waardering van bodems ook op verschillende schalen uitgevoerd kan worden.

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