A framework for integrating environmental modeling towards enhancing bundled environmental services: An example from an Andean watershed

By

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Contributions of Authors

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Authors	Statement of Contribution
Martha Otero	Wrote the manuscript, implemented model set-up, simulations and execution, and analyzed data.
Raja Sengupta	Guided research methodology, supervised data analysis and edited the manuscript.
Jorge Rubiano	Aided with model implementation and data analysis.

Integrating environmental modeling towards bundling of environmental services in an Andean watershed

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Martha Otero	Wrote the manuscript, conducted field work, implemented model set-up and simulations, and analyzed data.
Raja Sengupta	Guided research methodology, supervised data analysis and edited the manuscript.

Bundling of environmental services in an Andean watershed: valuation and tradeoffs

Authors	Statement of Contribution
Martha Otero	Wrote the manuscript, designed the environmental index and analyzed data.
Raja Sengupta	Guided research methodology, supervised data analysis and edited the manuscript.

Abstract

There is an increasing interest in environmental benefits provided by ecosystems. Among the wide range of environmental services, hydrological services are important not only because they affect all sectors of human behavior, but because they are closely tied to other services such as food supply and carbon sequestration. Investments in watershed management and the development of new market incentives are of increasing importance in both temperate and tropical regions. One of the mechanisms that have been strongly promoted is the adoption of Payment for Environmental Services (PES). In Latin America, most PES programs have been designed by estimating the provision of services using conventional knowledge about the direction and magnitude of the linkages between land use and hydrological variables. However, more often than not, we do not actually have measurements and data about the relationship between land use and the provision of many environmental services. Therefore, there is an increasing demand for estimating environmental services tradeoffs emerging from land use changes in order to enhance the bundle of ecosystem services available from a watershed, as well as methods to quantitatively evaluate these linkages.

This thesis presents a conceptual framework for evaluating ecosystem functions and the application of an environmental index that combines environmental assessments and economic valuation. Three environmental services were evaluated: erosion control, discharge regulation and carbon sequestration. The 258 km^2 of Las Ceibas watershed served as a case study, which is an agricultural basin located in Colombia, South America. Runoff and sediment processes were analyzed using the Soil and Water Assessment Tool (SWAT), and carbon sequestration potential using the CO2FIX model. I simulated three scenarios: (1) rotational grazing, (2) green manures and cover crops and (3) reforestation. The results indicate that reforestation, which mainly targeted the increase of carbon sequestration, also improved the provisioning of hydrological services through improvements in erosion control and discharge regulation. Synergies were found between environmental services in the

three conservation-oriented scenarios analyzed. Tradeoffs were found between annualized profits from agricultural activities with sediment control, and annualized profits from agricultural activities with carbon sequestration. Environmental modeling proved to be a useful tool in making the synergies in bundled environmental services evident; furthermore it suggested alternatives for the most feasible management practice.

Résumé

Il y a un intérêt augmentant pour les avantages de l'environnement fournis par les écosystèmes. Parmi la large gamme de services de l'environnement, les services hydrologiques sont importants non seulement parce qu'ils affectent tous les secteurs de conduite humaine, mais parce qu'ils sont de près attachés à d'autres services comme les réserves d'aliments et la mise sous séquestre de carbone. Les investissements dans l'administration de ligne de partage des eaux et le développement de nouveaux stimulants du marché ont de l'importance augmentante dans les régions tant tempérées que tropicales. Un des mécanismes qui ont été fortement promus est l'adoption de Paiement pour les Services de L'environnement (PSE). En Amérique latine, la plupart des programmes PSE ont été conçus en estimant la disposition de services, la connaissance conventionnelle de la direction et de l'étendue des liaisons entre l'utilisation de terrain et les variables hydrologiques. Pourtant, plus fréquemment que pas, nous n'avons pas vraiment de mesures et de données du rapport entre l'utilisation de terrain et la disposition de beaucoup de services de l'environnement. Donc, il y a une demande augmentante pour estimer des échanges de services de l'environnement émergeant des changements d'utilisation de terrain pour améliorer le paquet de services d'écosystème disponibles d'une ligne de partage des eaux, aussi bien que des méthodes de quantitativement évaluer ces liaisons.

Cette thèse présente un cadre conceptuel pour évaluer des fonctions d'écosystème et l'application d'un index de l'environnement qui combine des évaluations de l'environnement et une évaluation économique. Trois services de l'environnement ont été évalués : le contrôle d'érosion, le règlement de renvoi et la mise sous séquestre de carbone. 258 km2 de Las Ceibas la ligne de partage des eaux ont servi d'une étude de cas, qui est une cuvette agricole trouvée en Amérique du Sud, Colombie. J'ai analysé le ruissellement et les processus de sédiment en utilisant le Sol et l'Instrument d'Évaluation D'eau (SWAT) et le potentiel de mise sous séquestre de carbone l'utilisation du modèle de CO2FIX. J'ai simulé trois scénarios : le pâturage rotationnel, les fumiers verts et les récoltes de couverture et reforestation. Mes résultats indiquent que reforestation, qui a surtout visé l'augmentation de mise sous séquestre de carbone, a aussi amélioré un approvisionnement de services hydrologiques par l'amélioration de contrôle d'érosion et de règlement de renvoi. Les synergies ont été trouvées entre les services de l'environnement dans les trois scénarios donné la nature de conservation des scénarios analysées. Les échanges ont été trouvés entre des utilités per année des activités agricoles avec le contrôle de sédiment et des utilités per année des activités agricoles avec la mise sous séquestre de carbone. Le fait de modeler de l'environnement s'avér être un instrument utile dans la réalisation des synergies évidentes dans les services fourrés de l'environnement; en outre il améliore la recherche vers la pratique d'administration la plus réalisable.

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Chapter 1

1 INTRODUCTION

1.1 Thesis Objectives

The overall goal of my research is to develop a methodological framework to integrate environmental modeling for bundling ecosystem services available from a watershed. The central hypothesis is that environmental services are bundled in such a way that synergies between these services are possible in landscapes located in the Andean environment. This research will be focused on environmental services provided by Las Ceibas Watershed in the Andean region of Colombia, where water runoff from the upper-catchment is increasing the sediment loads to a downstream water supply system. The specific objectives are to: (1) assess quantitatively the current situation of hydrological services in terms of erosion control and runoff through hydrological modeling, (2) estimate the potential to enhance hydrological services under different land use change scenarios, (3) asses carbon sequestration in current land uses and those land uses selected to improve hydrological services, (4) estimate the costs and benefits of potential land use change scenarios, and (5) evaluate land use change scenarios based on environmental and economic considerations.

The development of this methodological framework will initially be focused on determining the current land cover/land use in the area of concern, i.e. forests, agroforestry, agricultural, crops or grasslands, or mixtures of the above. Subsequently possible scenarios of changes would be developed and used to generate landscapes that were likely occur if economic and policy incentives that acted as drivers of land use change were implemented. The framework should be able to answer the following questions: (1) what is the potential of landscapes to increase hydrological services based on realistic future land use change scenarios? (2) what are the land use changes and management practices that could have a positive impact on producing “bundled” environmental services?, and (3) what are the best land use change scenarios from the environmental and economic perspectives?

1.2 Thesis Outline

The structure of the thesis is a manuscript-based thesis, in which chapters three, four and five correspond to manuscripts submitted to scientific journals. The chapter content is as follows:

Chapter 2 presents part of the research context from which this study has been developed, which was not addressed in the manuscripts. The research objectives have been described in this chapter, which starts with a definition of environmental services, followed by issues regarding environmental modeling, and socio-economic considerations regarding environmental services.

Chapter 3 is based on the first manuscript, where the scale effects of digital elevation models (DEM) on river network delineation and hydrological output is investigated. The main results from the hydrological modeling with the Soil and Water Assessment Tool (SWAT) are presented in this chapter together with a sensitivity analysis of model parameters and a validation approach. The effect of the 50m and 90m DEM on discharge and sediments are compared against the measured data.

Chapter 4 presents the second manuscript that elucidates the environmental modeling methodology used to quantify three environmental services in Las Ceibas watershed: erosion control, discharge regulation and carbon sequestration. Three potential scenarios of land use change and management practices are modeled and their effects on the environmental services are quantified and compared with the current conditions.

Chapter 5 contains the third manuscript, in which an environmental index is proposed to evaluate the services provided by Las ceibas for a particular land use change scenario. This index integrates the provisioning of the three services analyzed and the net present value of each alternative. The final index obtained for each of the three future scenarios is reviewed towards the bundling of environmental services in Las Ceibas watershed.

In Chapter 6 the methods used in this dissertation are reviewed and critiqued, leading to a set of avenues for potential future research, and identifying applications to other areas and challenges for bundling of environmental services.

Chapter 2

2 ENVIRONMENTAL SERVICES: PHYSICAL, SOCIAL AND ECONOMIC ISSUES

2.1 Environmental services of watersheds

According to the Millennium Ecosystem Assessment, ecosystem services are the benefits people obtain from ecosystems. They can be classified as (1) provisioning services, such as food, timber and water; (2) regulating services, such as water regulation and disease control; (3) supporting services, such as soil formation and nutrient cycling; and (4) cultural services, such as recreational, spiritual, religious and other non-material benefits (MEA 2005). There is no agreement regarding the concept of environmental services but in general the term refers to provisioning, regulating and supporting cultural ecosystem services as outlined in the Millennium Assessment's definition (Table 2.1). This research will be focused on hydrological services, relating to the provisioning and regulation of water, carbon sequestration and food production.

The hydrological services provided by watersheds, also called watershed functions, include regulation of water flow, maintenance of water quality, control of soil erosion and sedimentation, reduction of land salinization, regulation of ground water levels and maintenance of aquatic habitats. They can also be grouped in five general categories regarding the water cycle in a watershed: (1) collection of the water from rainfall and snowmelt, (2) storage of various amounts for different time periods, (3) discharge of water as runoff, (4) chemical reaction along pathways and detention sites, and (5) provision of habitat for the flora and fauna that constitute the biological elements of ecosystems (Black 1997). The concept of watershed functions is presented by Chow (1964) where he discusses the hydrologic functions of vegetative cover in terms of its beneficial effects. These effects include (1) build-up of organic matter in the soil; (2) organic material on the soil that protects against soil erosion; (3) slowing the runoff process; (4) increasing infiltration, and (5) shading that causes the reduction of snowmelt rates and evaporation. Since then watershed functions have been widely study and have captured the attention of researchers in the physical and social sciences as well as policy makers.

Table 2.1. Classification and examples of environmental services.
(Costanza et al. 1997; Daily 1999; de Groot et al. 2002; MEA 2005).

Production of goods
Food: terrestrial animal and plant products, forage, seafood, spice
Pharmaceuticals: medicines, precursors to synthetic drugs
Durable materials: natural fibre, timber
Energy: biomass fuels, low-sediment water for hydropower
Industrial products: waxes, oils, fragrances, dyes, rubber, precursors to synthetic products
Genetic resources: the basis for the production of other goods
Supporting services
Dispersal of seeds necessary for revegetation, pollination of crops and native vegetation
Soil formation and nutrient cycle
Regulating services
Carbon sequestration
Climate regulation through regulation of albedo, temperature and rainfall patterns
Regulation of the timing and volume of river and ground water flows
Protection against floods by coastal or riparian systems
Regulation of erosion and sedimentation
Regulation of species reproduction (nursery function)
Breakdown of excess nutrients and pollution
Regulation of pests and pathogens
Protection against storms
Protection against noise and dust
Biological nitrogen fixation
Cultural services
Nature and biodiversity (provision of a habitat for wild plant and animal species)
Provision of cultural, historical and religious heritage (e.g., a historical landscape)
Provision of scientific and educational information
Provision of opportunities for recreation and tourism
Provision of attractive landscape features enhancing housing and living conditions

However, despite decades of research and the wide range of hydrological models available, there is still a lack of information for policy questions about different watershed functions especially at large scales (Tomich et al. 2004). Studies of other watershed services, such as stream flow stabilization, water quality and quantity effects (particularly in the case of tropical settings) have seldom been done (Kramer et al. 1998). And although several studies have focused on soil erosion control, most of them were focused on on-site effects such as reduction of productivity, whereas off-sites effects such as siltation of reservoirs have been less explored (Lal 1998). Further, a vast majority of these studies are from the U.S., Canada, Australia, and Europe, and only a few from regions in the tropics and subtropics.

Besides hydrological services, carbon sequestration is another regulating environmental service that ecosystems can provide (MEA 2005). The atmospheric concentration of carbon dioxide (CO2) has increased by 31% since 1750. About three-quarters of the anthropogenic emissions of CO2 to the atmosphere during the past 20 years are due to fossil fuel burning. The rest is predominantly due to land-use change, especially deforestation (IPCC 2001). According to the Millennium Ecosystem Assessment (2005), the effect of changes in terrestrial ecosystems on the carbon cycle reversed during the last 50 years. From being a carbon 'sink' that stored carbon, they became a net source of CO2 during the nineteenth and early twentieth centuries, primarily due to deforestation, but also with contributions from degradation of agricultural, pasture, and forestlands. At present, carbon losses from land use change continue at high levels. Mitigating this grim scenario is the fact that properly managed terrestrial ecosystems can also act as carbon sinks through sequestration. Factors contributing to the growth of the role of ecosystems in carbon sequestration include afforestation, reforestation, forest management and changes in agriculture practices. However, according to the same assessment, the future contribution of terrestrial ecosystems to the regulation of climate is uncertain given the limited understanding of soil respiration processes.

This lacunae in knowledge about future contributions of land use change to the carbon cycle is a major impediment to the main objective of the United Nations Framework Convention on Climate Change (UNFCCC), which is "the stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system", since carbon sequestration is touted a way to mitigate greenhouse gas emissions. The Clean Development Mechanism (CDM) established under the UNFCCC provides financial support to developing countries in return for greenhouse gas reductions through payments for carbon sequestration in landscapes. In the first stages of the CDM, sinks were limited to afforestation and reforestation projects for the first commitment period. However, sinks resulting from cropland management, grazing land management, re-vegetation, and forest land management are now also recognized (Dumanski 2004).There are three general means by which agricultural and forestry practices can reduce greenhouse gases: (1) avoiding emissions by maintaining existing carbon storage in trees and soils; (2) increasing carbon storage by, for example, conversion from conventional to conservation tillage practices on agricultural lands; (3) substituting bio-based fuels and products for fossil fuels, such as coal and oil, and energy-intensive products that generate greater quantities of CO_2 when used.

Therefore, one can assume that CDM payments, including those designed to prevent tropical deforestation, will help reduce CO_2 emissions as this process is estimated to be responsible for about 20% of the world's annual emissions (IPCC 2001). Biomass and carbon content are generally high in tropical forests, reflecting their influence on the global carbon cycle. Tropical forests also have great potential for the mitigation of CO_2 through appropriate conservation and management due to their high rates of net primary production. According to a review by Silver et al. (2000) on reforestation of abandoned tropical agricultural and pasture lands, the above-ground biomass increases at a rate of 6.2 Ton C/ha/year during the first 20 years of succession, and at a rate of 2.9 Ton C/ha/year over the first 80 years of regrowth. During the first 20 years of regrowth, forests in wet zones have the fastest rate of carbon accumulation aboveground, followed by forests in the moist and dry zones. Tropical reforestation has the potential to serve as carbon offset mechanism both above and below-ground at least for 40 to 80 years. The review also indicates that forest growing on abandoned agricultural land accumulates biomass faster than other past land uses, while soil carbon accumulates faster on sites that were cleared but not developed, and on pasture sites. Another study proved that invasion of grasslands by shrublands increased carbon in vegetation to a much lower extent than usually expected, where soil carbon increased only on the drier sites and decreased in the wetter sites (Jackson et al. 2002). Regional estimates of the carbon sequestration potential of these practices are crucial if policy makers are to plan future land uses to reduce national CO_2 emissions and to participate in carbon trading markets.

Another program that emerged in 2007 after the UNFCCC is the The United Nations Collaborative Programme on Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (UN-REDD Programme), which is a collaboration between FAO, UNDP and UNEP. A multi-donor trust fund was established in July 2008 that allows donors to pool resources and provides funding to activities towards sustainable management of forests.

There is not yet an international agreement about a global REDD scheme. Based on current events, it is expected that governments of developing countries that choose to participate in REDD may develop their own strategies to reduce forest related green house emissions. To qualify for REDD payments, they would have to comply with a set of requirements such as the establishment of a base line of carbon stock fluctuations over time, and the establishment of methodologies to measure and monitor fluctuations in carbon stocks to assess the effects of their strategies on emissions (Kaimowitz 2008).

During the negotiations on how REDD should be implemented, several methodological concerns have been raised, specifically regarding additionality, leakage and non-permanence. The concept of additionality addresses the question of whether emission reductions and/or carbon sequestration would also have happened without payments for carbon credits. Only carbon credits from projects that are "additional to" the business-as-usual scenario represent a net environmental benefit. Leakage occurs when there is an increase in carbon dioxide emissions in one region or country as a result of an emissions reduction by a second region or country. Whereas international leakage is difficult to prevent by national governments, at least within the countries or within certain regions countries will have to demonstrate that reducing deforestation in one area does not cause additional deforestation in another area. Non-permanence involves the risk that emission removals by sinks are reversed, because forests are cut down or destroyed by natural disaster (Angelsen & Wertz-Kanounnikoff 2008).

In February 2009, the Readiness Plan Idea Notes (R-PINs) of 25 countries have been approved. The 25 accepted countries are eligible to receive funding to develop a

Readiness Plan (R-Plan), which will elaborate on the R-PIN and present a more detailed strategy for realizing REDD at the national level. The emphasis of the Readiness Mechanism is to assist developing countries determine a national reference scenario of deforestation, develop a monitoring system for REDD, and adopt a national strategy for reducing deforestation and forest degradation. The World Resources Institute analyzed the R-PINs and found that (Davis et al. 2009):

- Law enforcement challenges require greater attention,
- Unclear tenure is a major challenge in most countries, and responding to this challenge will require much more effort,
- Measure to increase policy coherence between sectors, particularly with regards to land use planning, need more attention,
- The adequacy of existing revenue distribution and benefit-sharing mechanisms should inform the development of a payment system under REDD,
- Transparency and accountability in forest monitoring systems for REDD need to be emphasized.

2.2 Hydrological Impacts of Land Use Change

Disturbance of the ecosystems can take many different forms, from conversion of forest into agricultural land to the development of industrial and urban centers. Each type of intervention will have its particular impacts on the existing hydrological cycle. These hydrological impacts can be grouped according to whether they are related primarily to water quality or water quantity. Usually impacts such as erosion, sedimentation and nutrient outflow are grouped together under water quality issues; while changes in water yield, seasonal flow, storm flow response, groundwater recharge and precipitation are considered as water quantity issues. The nature of these impacts on the economy can be summarized depending on the spatial context of the effect, either a reduction in on-site production (soils) or downstream off-site production or consumption (stream flow quality and quantity).

A review of several studies conducted by Aylward (2002) summarizes the hydrological impacts of changes in land use and conversion of tropical forests:

Water Quality

- Erosion increases with forest disturbance depending on the type and duration of the intervention.
- Increase in sedimentation rates are likely as a result of changes in vegetative cover and land use, and the amount is determined by the kind of processes supplying and removing sediment prior to disturbance.
- Nutrient and chemical outflows following conversion generally increase as leaching of nutrients and chemicals is increased.

Water Quantity

- Water yield is inversely related to forest cover, with the exception of upper mountain cloud forests where horizontal precipitation may compensate for losses due to evapotranspiration.
- Seasonal flows, in particular dry season base flow, may increase or decrease, depending on the net effect of changes in evapotranspiration and infiltration.
- Peak flow may increase if hill-slope hydrological conditions lead to a shift from subsurface to overland flows, although the effect is of decreasing importance as the distance from the site and the number of contributing tributaries in a river basin increase.
- Groundwater recharge is generally affected in a similar way to seasonal flows.
- Local precipitation is probably not significantly affected by changes in forest cover (at least up to a scale of 10 Km). Loss of cloud forests represents an exception because it is associated with loss of horizontal precipitation.

A review conducted by Brujinzel (2004), mainly in the Southeast Asian region, concluded that although reforestation and soil conservation measures are capable of reducing the enhanced peak flows and stormflows associated with soil degradation, no well-documented case exists where these measures have also produced a corresponding increase in low flows. This effect on peak flows can be associated with newly planted trees due to higher water use. With the removal of forest biomass, the total annual water yield increases, and rainfall infiltration opportunities are usually reduced to the extent that groundwater reserves are replenished insufficiently during the rainy season, resulting in a strong decline in dry season flows. According to Calder (2002), competing processes such as transpiration and infiltration may result in either a decrease or increase in dry season flow after reforestation or afforestation. Regarding water quality and erosion, Brujinzel (2004) concluded that plant cover is generally capable of preventing surface erosion and, in the case of a well-developed tree cover, shallow land slides as well, but more ‘deep-seated’ slides are determined rather by geological and climatic factors. Calder (2002) pointed out that it is a conventional fallacy that plantation forest can necessarily achieve the same erosion benefits as natural forests.

Given the accelerated attention paid to afforestation for carbon sequestration, Jackson et al. (2005) combined field research and 600 catchment observations to document changes in stream flow and water quality due to tree plantations. They concluded that substantial losses in stream flow, and increased soil salinization and acidification can be expected with afforestation. Plantations decreased stream flow by 227 millimeters per year globally (52%), with 13% of streams drying completely for at least 1 year. Further research is needed to determine the time lag between upland soil conservation measures and any resulting changes in sediment yield at large distances downstream, as significant variations can occur as a result of erosion from afforestation activities and underlying geological characteristic of watershed.

2.3 Environmental Modeling

The first step for understanding and quantifying environmental services provided by a watershed is to select appropriate models that can capture current conditions regarding climate, topography, soils and land use variability at the specific scale of interest. The selection of the model usually depends on the needs and use of the final outputs, the spatial and temporal scales, and data availability. In this case, the purpose is to estimate the hydrological services and carbon sequestration, and identify risk areas and hot spots where future changes can have an impact on the hydrological services provided to on and off site users. Biophysical interdependencies exist between land uses and water flow responses; therefore the selected models should provide enough information regarding sediments and water yields, which can be associated with specific land uses in the watershed.

2.3.1 Hydrological modeling

In general terms, hydrological models can be grouped in three categories: empirical, conceptual and physically-based models, depending on the physical processes simulated, the model algorithms describing these processes, and the data dependency of the model (Saavedra 2005). Empirical models are based on the analysis of field experiments and estimates are done using statistical inference. The limitation of applying empirical models at catchment level is the assumption of stationary, which assumes that underlying conditions do not change during the simulation period (Kandel et al. 2004). Conceptual models use empirical relationships and represent the catchment as a series of linear storages (Sivapalan et al. 2002). Parameter values for this type of model are usually obtained through calibration against observed data, which could represent a problem for ungauged watersheds (Zhou & Liu 2002). Physically based models are based on equations that represent physical processes related to transfer of mass, momentum and energy (Kandel et al. 2004; Merritt et al. 2003). Equations are used with continuous temporal and spatial data, although in practice the data used are often point source data that represent a unit of area. This is one criticism to these models as physical significance of these equations may be lost when using parameters at small scale (Saavedra 2005).

2.3.2 Model Uncertainty

Among the sources of information available to policy makers are predictive models capable of simulating the behavior of hydrological systems over a broad range of spatial and temporal scales. Models attempt to represent the complex, spatially-distributed interactions of water, energy and vegetation by means of mathematical equations (Cao et al. 2006). Physically based, distributed hydrological models are increasingly being used to solve complex problems in water resources applications (Bobba et al. 2000; Ewen et al. 2006), including environmental impacts of land-use changes, effects of climate change on water resources, forecast flood events and water planning and management in a catchment. Models are being developed to support decisions related to assessment of current conditions from previous activities, and to estimate the impact of alternative actions (Marnicio & Rubin 1988). However, problems with hydrological models include a lack of sufficient data to fully characterize spatial variability, scale problems for integrating field measurements with model parameter elements, and imperfect representations of real processes in models (Beven 1989; Beven & Freer 2001; Grayson et al. 1992). These factors result in a requirement for estimating model uncertainty and model calibration and validation (Anderton et al. 2002; Refsgaard 1997).

A definition of uncertainty analysis is 'the means of calculating and representing the certainty with which the model results represent reality' (McIntyre et al. 2002). Good modeling requires that the input data, structural and other uncertainties be propagated into the model predictions and communicated in an appropriate manner to the decision maker or stakeholder, and thus allowing an appropriate degree of confidence that can be attributed to the model results. There has been a surge in the attention given to methods for dealing with model uncertainty as: (a) decision makers begin to press for better quantification of the accuracy and precision of hydrological model predictions, (b) interest increases in methods for properly merging data with models and for reducing predictive uncertainty and (c) scientists begin to search for better ways to represent what is, and what is not, well represent about the hydrological systems they study (Wagener & Gupta 2005).

The implementation of models in policy making should reflect their intended use and the manner in which the results will be incorporated into the policy making process. In an ideal world, the model developer should know who will be using the model, what type of questions they are likely to address and how they will want information to be detailed or aggregated. Based on these considerations, the modeler can select the appropriate software and hardware systems, user's interfaces and model input and output formats. In reality, these considerations are very difficult to define and are constrained by the context of policy analysis. However, these considerations can increase the probability that the model is used effectively in supporting policy making (van Asselt & Rotmans 2002). The sources of model uncertainty are:

Model structural uncertainty

Model structural uncertainty is introduced through simplifications, inadequacies and/or ambiguity in the description of real-world processes (Wagener & Gupta 2005). For example, the structure of a conceptual rainfall-runoff model aggregates, in space and time, the hydrological processes occurring in a catchment into a number of responses represented by storage components and their interactions or fluxes (Ford 1999). This conception of reality might be poor, particularly regarding subsurface characteristics, and therefore uncertain. Consequently the conceptual representation of the watershed that is translated into a mathematical form in the model is carrying this initial uncertainty.

Structural uncertainty is also related to the scale issue since some of the mathematical descriptions of hydrological processes were derived in the laboratory and then applied to the catchment scale, such as Darcy's law that describes fluid permeability or sheet flow assumptions for surface runoff (Beven 1993). Since the scale that the hydrological modeling represents is different from the laboratory scale, and the diversity and heterogeneity found in natural environments must be modeled approximately using lumped state variables. This means that relationships and parameter values identified at laboratory scale should be only used as a starting point for model design, rather than as a definitive end result (McIntyre et al. 2002).

In the process of conceptualization of reality, the modeler uses a set of parameters to describe these aggregated processes and they may cover a large number of sub-processes that cannot be represented separately or explicitly (Van Straten & Keesman 1991). Aspects such as the size of the storage components, the location of outlets or the distribution of storages within the catchment can be described by the model parameters. The uncertainty introduced at this stage is through the assumption that these parameters are constants and representative of natural properties of the system (Wagener & Gupta 2005). Butts et al. (2004) grouped the differences between model structures due to the initial assumptions in:

- description and coupling of processes,
- representations of the spatial variability (e.g. zones, grids, sub-catchments),
- sub-grid process representations including distribution functions,
- degrees of lumping,
- interpretation and classification of variables (e.g. soil type, geology and land use cover).

The conceptual model describes the functioning of the basin and should be, in theory, based on experimental findings. However, these results are often missing and the understanding of the dominating processes at different scales is still incomplete (Uhlenbrook 2006).

Data structural uncertainty

This type of uncertainty is caused by errors in the measurement of input and output data, or by data processing. A large portion of prediction uncertainty is often due to problems and errors of input data, such as low quality, lack of long-term data, too few measuring stations, or difficulties in regionalizing point measurements to catchment scale (Blöschl & Grayson 2000). Input data is used to drive the model, which will only be an approximation of the real world and therefore imply problems of detection and measurement. A common example is the uncertainty in precipitation data that result from inadequate spatio-temporal sampling densities (Wagener et al. 2003). Additional uncertainty is introduced if long-term predictions are made, for example in the case of climate change scenarios for which per definition no observations are available.

Data processing uncertainty is also introduced when a model is required to interpret actual measurements. For example, radar rainfall data are measurements of reflectivity, which have to be transformed to rainfall estimates using an empirical model with a functional relationship and calibrated parameters, both of which carry a degree of uncertainty. Finally, a hydrological model might also be integrated with other type of models, such as socio-economic models, to assess for example impacts of water resources changes on economic behavior. Data to constrain these integrated models is rarely available (Letcher et al. 2004).

Parameter estimation uncertainty

Parameters are frequently not measurable in the field or they are not clearly related to catchment properties. Based on this ambiguity, Beven and Binley (1992) defined the equifinality problem, meaning that different parameter sets or different models can yield equally good simulation. In other words, the uncertainty arises when it is not possible to 'uniquely' locate a 'best' parameter set based on the available information. The lack of a unique correlation between conceptual model parameters and physical watershed characteristics will commonly result in significant prediction uncertainty if the model is extrapolated to predict the system behavior under changed conditions such as land use change or urbanization (Wagener & Gupta 2005).

Parameter estimation uncertainty is also related to the application of hydrological models in ungauged basins, where the behavior of a similar, but geographically different watershed is predicted for which no observations of the variable of interest are available. Therefore, adjustments of the model parameters (or even the model structure) have to be done in order to consider the characteristics of the represented system. In such cases the degree of adjustment is difficult to determine without measurements of the system response.

2.3.3 Watershed-scale models

Some of the commonly used watershed-scale hydrologic and nonpoint-source pollution models include: Agricultural NonPoint Source pollution model or AGNPS (Young et al. 1989), KINematic runoff and EROSion model or KINEROS (Woolhiser et al. 1990), the European Hydrological System model or MIKE SHE (Refsgaard & Storm. 1995), Precipitation-Runoff Modeling System or PRMS (Leavesley & Stannard 1995), Soil and Water Assessment Tool or SWAT (Arnold et al. 1999), Areal Nonpoint Source Watershed Environment Response Simulation or ANSWERS-Continuous (Bouraoui & Dillaha. 1996) and CASC2D (Ogden & Julien 2002).

AGNPS, ANSWERS-Continuous, and SWAT are continuous simulation models and are useful for analyzing long-term effects of hydrological changes and watershed management practices, especially agricultural practices. KINEROS (and also AGNPS) is single rainfall event model useful for analyzing severe actual or design single-event storms and evaluating watershed management practices, especially structural practices. CASC2D, MIKE SHE, and PRMS have long-term and single-event simulation capabilities.

Some of the models are based on empirical relations having robust algorithms, and others use physically based equations having computationally intensive numerical solutions (Borah & Bera 2003) The simple models are sometime incapable of giving desirable detailed results, and the detailed models are inefficient and could be prohibitive for large watersheds. Therefore, finding an appropriate model for an application and for a certain watershed is a challenging task. For certain applications, it is desirable to have a balance between the simple, approximate models and the detailed, computationally intensive models. Most of the commonly used models were formulated in the 1970s and 1980s, and lately most modeling research has focused on integrating modeling interfaces with geographic information systems and remote sensing data.

SWAT was developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods (Neitsch et al. 2002b). The SWAT model represents the heterogeneity of the studied area by dividing the watershed into subwatersheds, which are delineated based on an automatic procedure using Digital Elevation Model (DEM) data. Each subwatershed is parameterized using a series of hydrologic response units (HRUs). A HRU is a particular combination of land use, soil type and slope range. Subdividing the watershed into areas of unique land use, soil and slope range combinations enables the model to reflect differences in evapotranspiration and other hydrologic conditions for various crops and soils. With the SWAT model, runoff is predicted separately for each HRU and routed to obtain total watershed runoff. Soil water content, surface runoff, nutrient cycles, sediment yield, crop growth and management practices are simulated for each HRU and then aggregated for the sub-watershed by a weighted average.

The hydrology component is based on the water balance equation. The model predicts the surface runoff volume is predicted from daily rainfall using the soil conservation service (SCS) curve number (CN) equation:

$$Q_{surf} = \frac{(R_{day} - I_a)^2}{(R_{day} - I_a + S)}, \text{ Equation 2.1.}$$

where Q_{surf} is the accumulated runoff or rainfall excess (mm), R_{day} is the rainfall depth for the day (mm), I_a is the initial abstraction, which includes surface storage, interception, and infiltration prior to runoff (mm), and S is a retention parameter (mm) that varies among sub-watersheds according to soil type, land use, management and slope. S also varies with time because of changes in antecedent soil water content and is related to CN by the SCS equation below:

$$S = 25.4\left(\frac{1000}{CN} - 10\right), \text{ Equation 2.2.}$$

Peak runoff rate in the SWAT model is estimated by using the modified rational formula. Flow is routed through the channel using the Muskingum routing method. The watershed concentration time is estimated using Manning's formula, considering both overland and channel flow.

Regarding erosion and sediments transport, SWAT uses the Modified Universal Soil Loss Equation (MUSLE), which was derived from the Universal Soil Loss Equation (USLE) and is one of the most widely used models to study water soil erosion (Wischmeier & Smith 1978). It is designed to predict long-term average annual soil loss from field slopes under a specific land use and management system:

$$Sed = 11.8\left(Q_{surf}\, q_{peak}\, A_{hru}\right)^{0.56} K_{USLE} C_{USLE} P_{USLE} L_{USLE} F_{cfrg}, \text{ Equation 2.3.}$$

where Sed is the sediment yield (t) on a given day, Q_{surf} is the surface runoff volume (mm/ha), qpeak is the peak runoff rate, A_{hru} is the area of the HRUs (ha), K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the USLE cover and management factor, P_{USLE}

is the USLE support practice factor, L_{USLE} is the USLE topographic factor, and F_{cfrg} is the coarse fragment factor. The K factor reflects the susceptibility of soil particles to detachment by rainfall splash or surface flow, and is related to the integrated effect of rainfall, runoff, and infiltration. The LS factor accounts for the effect of slope length and gradient. In a slope, the length factor (L) is defined as the horizontal distance from the origin of overland flow to the point where deposition starts or runoff goes into a channel (Renard et al. 1997). The C factor reflects the effects of cover and management variables, while the P factor represents the effects of support practices such as contouring, strip cropping, terracing and subsurface drainage.

Major sources of uncertainty in SWAT model

One of the main source of uncertainty investigated using SWAT model is the one associated with model structure and model parameters (Huang & Liang 2006; Krysanova et al. 2007). It is related mainly to hydrological processes and the way these processes are parameterized. Therefore, uncertainty reduction relies on improved understanding of the physics processes and their effective representation. This can be achieved through improvements in model structure and model parameter estimations. An 'optimal set' of parameters can be found through calibration techniques and historical data for a given watershed. Studies (Beven & Binley 1992; Yapo et al. 1996) suggest that a single optimal parameter set for a hydrologic model may not exist and the uncertainties associated with the optimal parameter sets could be large. A model with the optimal parameter set may have the best fit over the period of the calibration data, but there may exist multiple parameter sets that are as good as the 'optimal' set. In addition, using different performance evaluation criteria could result in different optimal parameter sets. These multiple parameter sets are referred to as, for example, 'equifinality' (Beven and Binley, 1992) and 'equally probable parameter sets' (Van Straten & Keesman 1991). These limitations can be addressed by proving predictions within a range so that an optimal parameter set alone is not enough to represent the possible uncertainty associated with the model predictions. Thus, the parameter space needs to be sampled to generate realizations of the model simulations so that the prediction range can be estimated based on the model simulations. In this, a probabilistic approach based on Monte Carlo simulations is used to evaluate model performance and uncertainties associated with model parameters (see section 3.6.3).

2.3.4 Carbon Sequestration modeling

Till date, several models have been developed to analyze carbon levels and fluxes in ecosystems. These models range from very detailed ecophysiological models used in climate impact assessment, to very general empirical, descriptive models of ecosystem carbon budgets. None of these models has been accepted as a possible standard for carbon crediting projects (Table 2.2).

The selected model for this research was the CO2Fix simulation model. This is a tool which quantifies C stocks and fluxes in the forest biomass, the soil organic matter and the wood products chain. The model calculates the carbon balance with a time-step of one year (Masera et al. 2003). Basic input is stem volume growth and allocation pattern to the other tree compartments (foliage, branches and roots). Carbon stocks in living biomass are calculated as the balance between growth and turnover, mortality and harvest. The input for the soil module comes from turnover litter, mortality processes and logging slash. The organic matter decomposes and transforms into soil organic matter. The bioenergy module calculates the benefits for greenhouse gas emissions of the use of biomass instead of fossil fuels. Fuel sources for bioenergy can be either logging slash or industrial residues. In the financial module, costs and revenues can be specified to get an estimation of the project's profitability. In this research, the bioenergy and financial modules were not used given that there is no commercial activity from forests management.

Table 2.2. Carbon Sequestration models

Name	Description
Yasso07	Widely applicable soil carbon model with output characterized by uncertainty estimates. It was developed in particular focussing on climate and land use issues as well as greenhouse gas inventory systems.
Century	General model of plant-soil nutrient cycling which can be used to simulate carbon and nutrient dynamics for different types of ecosystems including grasslands and agricultural lands.
DNDC	DeNitrification-DeComposition simulation model of carbon and nitrogen biogeochemistry in agro-ecosystems.
Biome-BGC	Estimates fluxes and storage of energy, water, carbon, and nitrogen for the vegetation and soil components of terrestrial ecosystems.
Rothamsted Carbon Model	Medium to long-term soil organic matter turnover model.
EX-ACT	Ex-ante appraisal carbon-balance tool 6.4
CO2Fix Model	Quantifies the C stocks and fluxes in the forest biomass, the soil organic matter and the wood products chain with a time-step of one year.

Major sources of uncertainty in CO2Fix model

Errors in forest resource projections and C balances have two main sources: (1). the stochastic character of the estimated model coefficients and (2) measurement errors in the data, or lack of data used for model construction (Kangas 1997). Variability exists within one clearly defined forest type and is the result of different factors such as growth variation between years caused by weather circumstances, intra-species genetic differences, and site quality variation. This natural variability is not captured by CO2FIX because it very much relies on fixed input data from yield tables that can be seen as some sort of complete, and perfectly managed forests. Other stochastic events are management irregularity and risks such as storm and fire. Furthermore, natural variability occurs in carbon content of dry matter, basic wood density, litter and humus decomposition rates.

CO2FIX relies heavily on net annual increment data from yield tables. These tables are based on long-term measurement series in permanent plots and/or forest inventories. In these measurement series, errors and/or bias can occur. This type of uncertainty especially exists in the soil pools.

Nabuurs & Mohren (Nabuurs et al. 2008) carried out an uncertainty analysis of CO2FIX V 1.0. They specified input uncertainties in the form of simultaneous input distributions for an even-aged forest type. The 100 simulations with randomly chosen values of input gave an average total carbon stock of 316 Mg C/ha. The standard deviation was 12% and the 95% confidence interval was 254 - 403 Mg C/ha. They concluded that it was mainly the litter and humus coefficients and the carbon content that determined this uncertainty, but in general it was mainly the natural variability rather than a lack of data that determined the overall uncertainty.

2.4 Socio-economic considerations

In order to understand land use change process in a watershed and the related reduction in environmental services, it is necessary to identify the underlying causes by integrating the physical dimension with the economic and social dimensions. The Millennium Ecosystems Assessment (2005) named socio-economic drivers as indirect drivers of change. These drivers are divided into demographics, globalization, markets, consumption choices or beliefs. Ananda and Herath (2003) identify three major theoretical socio-economic frameworks concerning land use change and soil erosion: population pressure and poverty, past policies such as subsidies promoting intensive land use, and lack of institutions such as property rights and high transaction costs (Figure 2.1).

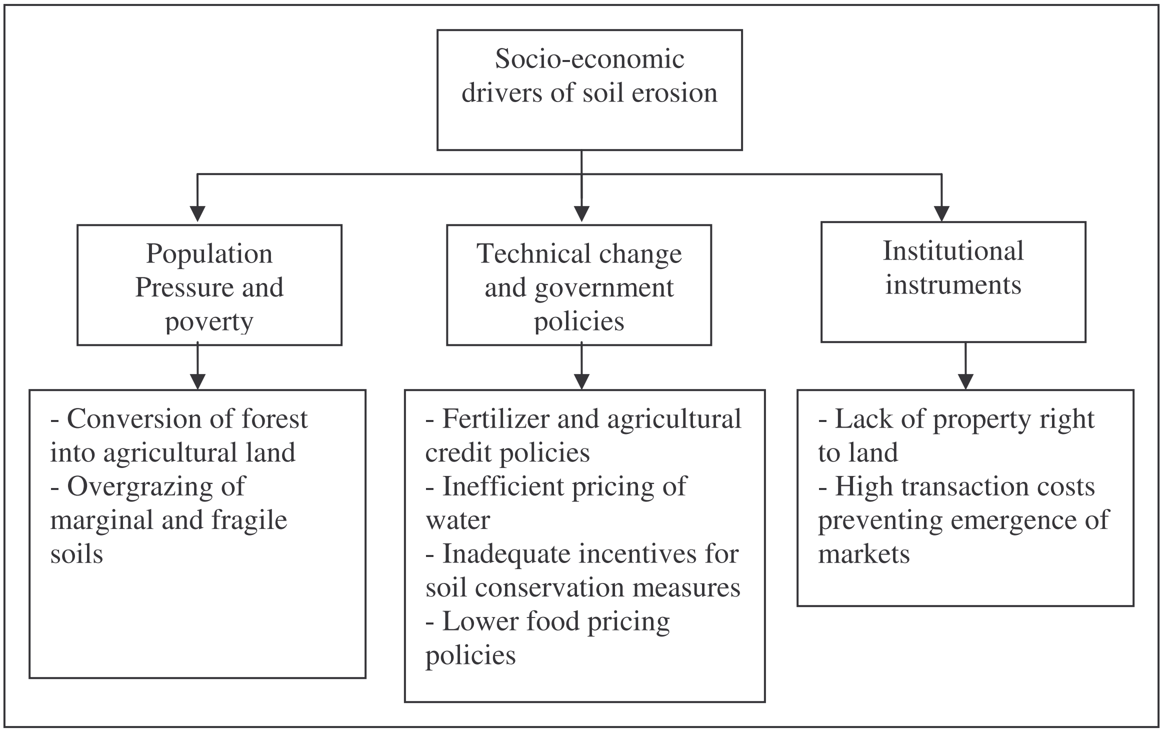

Figure 2.1. Socio-economic drivers of soil erosion (Ananda & Herath 2003).

There is a common hypothesis that rural poor are dependent on natural resources for survival and hence poverty is a major driver of degradation. Farmers overgraze in fragile soils such as those in the hillsides practicing shifting cultivation. According to Boserup's theory (1990), population growth may not necessarily lead to land degradation if there are certain technologies adopted and investments made while intensifying agriculture. These strategies refer to 'capital led' approaches that encourage conservation infrastructure such as terraces and planting perennials such as coffee and fruits trees. A study from Turner et al. (1998) in ten African countries and Barbier (1998) in West Africa supports this situation, where African farmers make considerable capital investments in protecting their land. Boserup (1993) argues that some of these conservation practices have been developed within the communities as traditional practices, without external interference. However, this path has not been followed in many Asian, Latin American and African countries, where the socio-economic conditions have constrained the implementation of conservation measurements (Kates & Haarmann 1992). These socio-economic conditions can describe vicious cycles where poor rural farmers with low returns cannot invest in the land hence driving degradation of natural resources (Barrett & Swallow 2005; Grey & Sadoff 2002). Other elements needed to close the cycle are the absence of proper incentives and lack of property rights.

Past policies such as fertilizer and agricultural credit policies during the Green Revolution supported intensive irrigation development and fertilizer use, which led to soil erosion. Another policy that has caused severe effects is controlling of food prices since it translates in lower returns for the farmers and consequently lowers investments in the land. On the other hand, import controls and high prices can also enhance degradation since farmers would prefer to intensify the production either by increasing the area (e.g. clearing of forest) or by using fertilizers (Ananda & Herath 2003). Ananda and Herath (2003) argue that poor institutional instruments such as weak property rights of land and water, and high transactions costs have constrained the emergence of markets that could mitigate soil erosion. In developing countries, insecure land tenure and property rights have undermined farmers' investment incentives in land.

In recent years, there has been an increasing emphasis on integrated watershed management (IWM) in many developing countries. Integration refers to shifting from technical interventions to more integrated approaches, with particular attention to social issues. Some of the concerns related to environmental services and the downstream – upstream relations on IWM are: (a) environmental degradation, particularly soil erosion and loss of hydrologic buffering in upstream areas, which have negative consequences in downstream areas; (b) changes in settlement patterns that expose greater numbers of downstream residents to fluctuations in stream flow; (c) increasing demands for limited supplies of water, in both upstream and downstream areas; and (d) alleviating poverty, which tends to be clustered in particular locations within watersheds (Swallow et al. 2006).

IWM requires a high degree of collective action, where individuals or groups working together toward common objectives. Although some components of watershed management such as cover crops, vegetative barriers or contour plowing can be applied by individuals on single farms, most operate across individual units, requiring some form of collective action (Swallow et al. 2005). This can include small groups of households coming together to manage water points, share stream flows, control localized erosion, and manage fertilizers and pesticides. IWM also calls for collective action among different actors located in different areas of the watershed given that actions of one group of users may have consequences for other users living in different zones, typically downstream. Coordination is necessary among different entities of the state that have policymaking, planning and authority functions. Many facets of IWM also call for cooperation between farmers and outside governmental or nongovernmental organizations.

2.4.1 Financing Mechanism to Enhance Environmental Services

Given the global concerns regarding the reduction of environmental services provided by tropical forests due to deforestation and unsustainable land management practices, some innovative financing mechanisms (IFM) have emerged in the last decades. However, these services are rarely adequately captured in terms of real financial and economic transactions (Costanza et al. 1997; Pagiola et al. 2004). IFM are defined by Verweij (2002) as 'an institutional arrangement that results in the transfer of new or increased financial resources from those willing to pay for sustainably produced goods and/or forest ecological services, to those willing to provide these goods and services in turn'. Richards (1999) classifies innovative financial incentive mechanisms into four main categories:

- Transfer payments involving the transfer of costs or benefits between different stakeholders, including fiscal market-based instruments and international transfer payments.
- The promotion of market or trade-based approaches.
- Promoting and influencing private or public investment flows.
- A property rights approach in which property and utilization rights are created, clarified, or modified.

The mechanism of Payment for Environmental Services (PES) is contained within this general concept of IFM. PES is defined by Piagola (2002) as a method of internalizing the positive externalities associated with a specific land use option. The concept of externality is introduced as an effect of particular land uses to off-site stakeholders. An environmental externality is defined as the non-marketed costs or benefits of land use actions which normally occur outside the forest (Pagiola et al. 2004; Richards 1999; van Noordwijk et al. 2004). There is also the case in which the one who generates a negative externality pays through pollution charges (Pagiola et al. 2005b). Another important concept in PES is internalization of externalities, i.e., making those who cause the externality aware of what they are doing and providing them with incentives to change their behavior (Bowers & Young 2000).

Environmental protection and poverty reduction have been promoted as the rationale for establishing PES in developing countries. Figure 2 presents a framework adapted from cases reviewed by Pagiola (2005a) and Landell-Mills (2002). This framework requires a multidisciplinary approach in order to achieve the environmental, economic and social assessments needed for designing a PES scheme. Other considerations needed that are not included in this framework are a legal and institutional analysis and market analysis.

According to the literature review on PES, there are several challenges when implementing this type of schemes (Burstein et al. 2002; Kiersch et al. 2004; Landell-Mills 2002; Llerena 2005; Pagiola et al. 2005a; Pagiola et al. 2005b):

- Biophysical identification and quantification of services within heterogeneous landscapes.
- Inclusion of unpredictable natural events such as floods and droughts in policy development.
- Estimation of tradeoffs between the services provided by a particular ecosystem
- Valuation of externalities and their internalization.
- Lack of legal and institutional frameworks.
- Variable willingness to participate.
- Uneven access to markets.

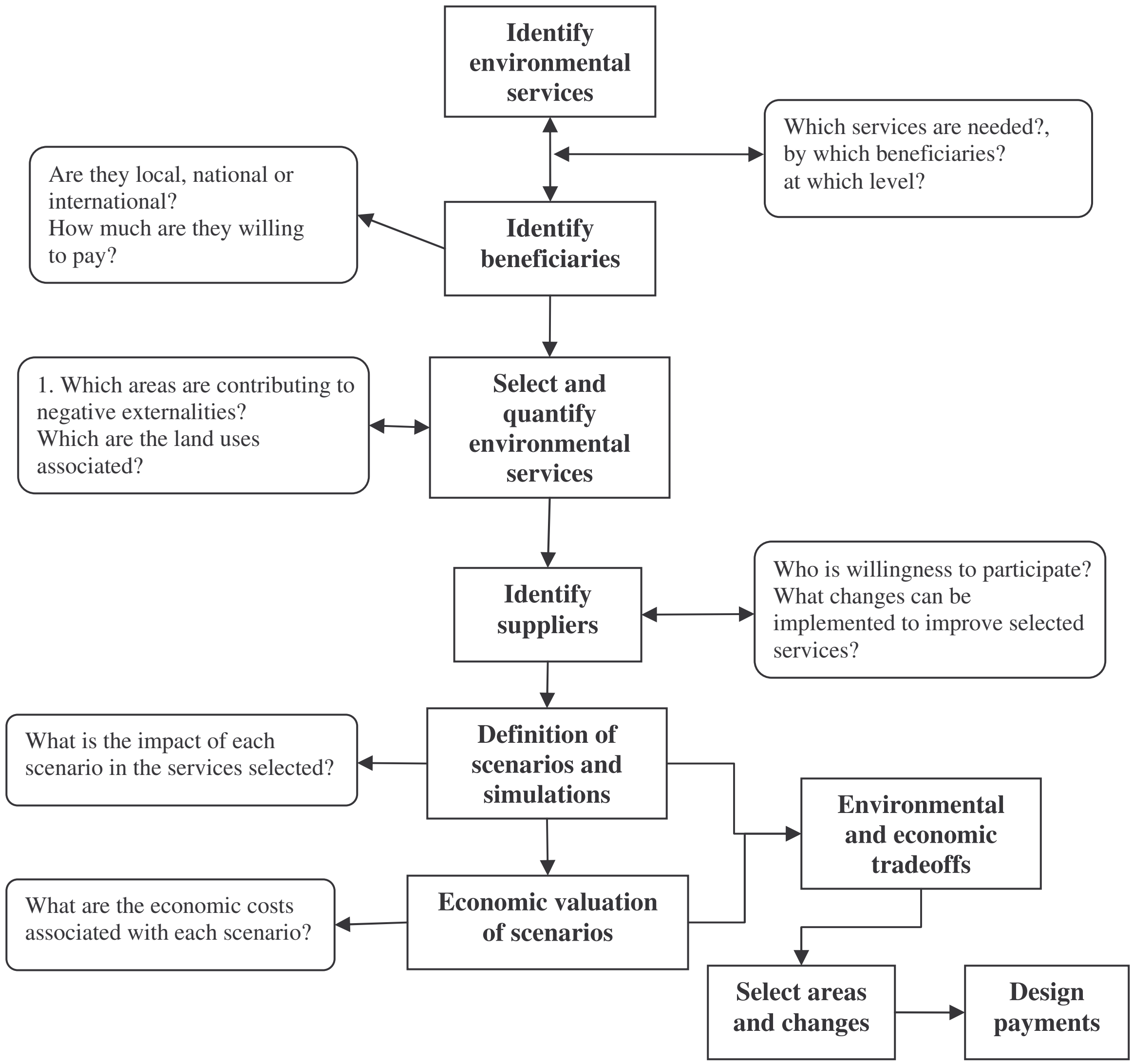

Figure 2.2. Framework for payment for environmental services.

2.4.2 Tradeoffs between environmental services

As mentioned above, there are different IFMs targeted at increasing the provisioning of environmental services and different scenarios of land use and management practices that enable this change. Each choice involves tradeoffs and will result in a different distribution of payments, therefore posing critical considerations for policy makers. For example, if the objectives of the mechanism is increasing income and providing environmental services, there will be a different geography of payments than if the only objective were the provision of environmental services. Environmental services tradeoffs can be classified in terms of the temporal and spatial scales, and the type of service targeted, and type of service ‘tradeoff’. Some examples of environmental services tradeoffs are: agricultural production and water quality, land use and biodiversity, water use and aquatic ecosystems.

The concept of tradeoffs has been traditionally associated with economics from the idea that resources are scarce (Stoorvogel et al. 2004). The traditional method used to estimate tradeoffs has been the cost benefit analysis. Given the large amounts of input data needed to conduct this type of analysis and the limitation to monetary terms, other methods such as multi-criteria analysis are being used. More recently tradeoff analyses have been developed not only based on monetary units but combined with physical or environmental units (MEA 2005).

Payments for environmental services in Colombia

In Colombia, there are two national programs called Certificates of Forest Incentives for Conservation and Reforestation, CIFc and CIFr respectively, which can be related to PES schemes. The incentive for conservation pays the owner of the land for protecting the natural forest ecosystems that have no or little previous intervention and the incentive for reforestation pays to owners or tenants for afforestation, and reforestation in areas without natural forests in the last five years. The conservation program had initially a budget of USD$600,000 in 1999. The funds were never executed because of the lack of operation guidelines that ensured the efficiency and transparency of funding allocation. The critical aspects that have been inhibiting the implementation of this program are the financial uncertainty to secure resources for the ten years period of the contractual terms, the willingness to allocate the funding from the Environmental Ministry to local authorities with different political interests, and the omission of the social heterogeneity implicit in a program of national scale.

Regionally, however, a pilot project from the Global Environmental Facility (GEF) entitled “Regional integrated silvopastoral approaches to ecosystem management project” has been implemented in Nicaragua, Costa Rica and Colombia. The aim of the project was to demonstrate and measure the effects of the introduction of payment incentives for environmental services to farmers on their adoption of integrated silvopastoral farming systems in degraded pasture lands, and the improvements in eco-systems functioning. The project in Colombia was implemented in the La Vieja watershed located in the coffee zone. After the coffee crisis, the main agricultural activities changed from coffee production to extensive cattle rising. The five years project started in 2003, covering an area of 63,831 ha, involving 80 farmers and with a total funding of USD$ 4.5 millions from the World Bank and USD$ 3.9 millions of matching funds. The area was divided in twenty eight land use classes which were ranked according to their potential of contributing to biodiversity and carbon sequestration. The ranking was basically made based on expert knowledge and multi-criteria analysis. The farms are visited annually to verify the changes in the land use before the payments are done. The weakness of this project is related to the extension and persistence of changes after the end of the project. The current source of funding is interested in global externalities, such as biodiversity and carbon sequestration, which are not easy to secure in the long term. There is still not enough information to measure the final impact of the project.

Another important case to cite is located in the Cauca River Valley, Colombia where water users were supported by the environmental authority and a private entity to form associations to protect and improve the hydrological services provided by upper catchments. The first association was created in 1987 and to date there are fifteen associations covering 602,000 ha and 3,825 water users, who represent 90% of the demand of the watersheds. The funding comes from voluntary payments depending on the water used and it is invested in different conservation practices such as reforestation, establishment of protected areas and incentives for changes in agricultural management practices. Although it can not be considered a 'true' example of a payment for environmental services (PES) scheme because there are no clear targeted services and providers, this case study is a valuable example of the sustainability of a project when local beneficiaries are involved.

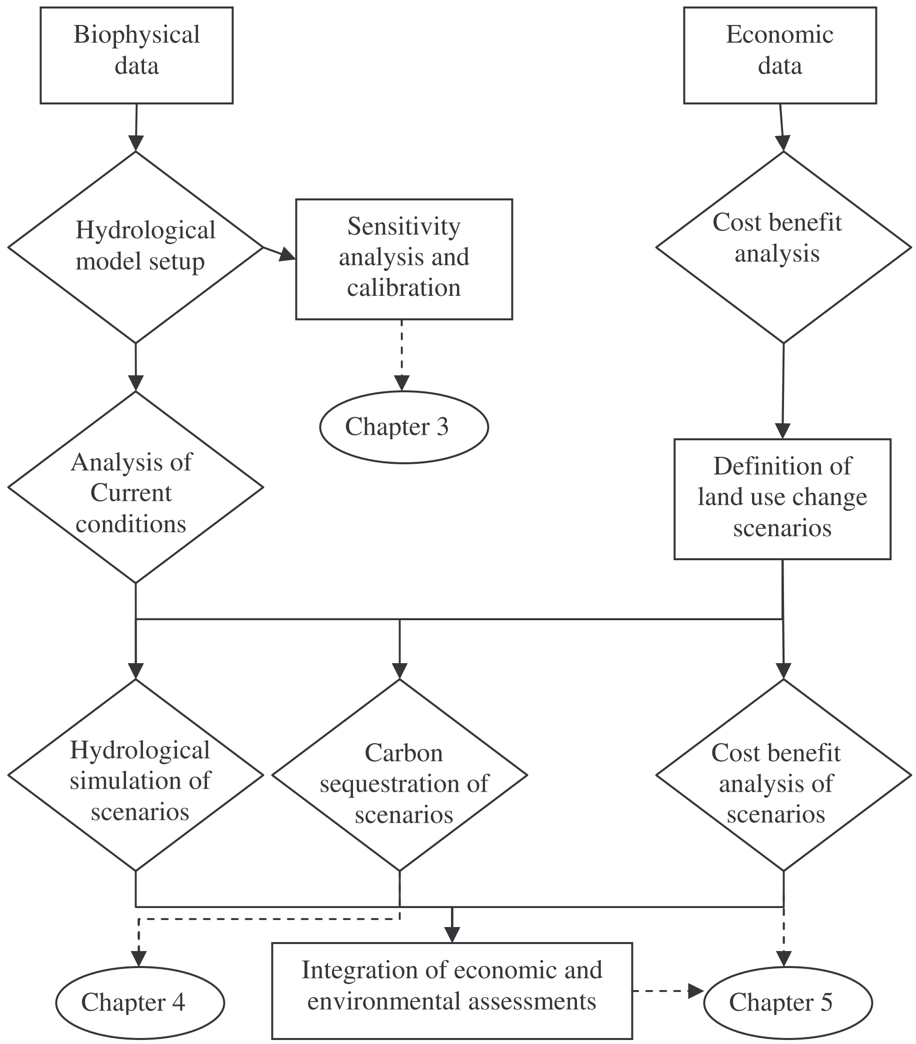

Figure 2.3. Flow of research activities.

Figure 2.3. indicates the research flow and the results of the different activities presented in the following chapters. The next chapter presents the results from the hydrological modeling for current conditions. Given the availability of three Digital Elevation Models, it was possible to analyze scale related-issues affecting the hydrological modeling process and how they affect the watershed delineation, flow and sediments yields. Results of quantifying hydrological services and carbon sequestration for potential land use scenarios are presented in Chapter 4. Consecutively, Chapter 5 presents the results of the economic analysis and its integration with biophysical analysis.

Chapter 3

3 EFFECT OF DEM RESOLUTION ON RIVER NETWORK DELINEATION AND HYDROLOGICAL MODEL OUTPUT

3.1 Abstract

Social and natural scientists acknowledge the importance of scale and how relationships and processes vary at different scales. Moreover, scale related-issues are of great concern in hydrological modeling for watershed management since hydrological processes occur at a wide range of spatial and temporal scales. Outputs from physical models depend on the quality and accuracy of the input data such as elevation data stored as a Digital Elevation Model (DEM). The resolution of the DEM is expected to affect the delineation of the river network and the watershed, which at the same time will affect the model's outputs such runoff, sediment and nutrient yields. Even though DEMs are increasingly available at a variety of resolutions, DEMs with resolutions finer than 30m are still costly especially in the developing countries. This paper addresses methods to measure the impact of DEM pixel size on river networks, and subsequently on simulated runoff. Using the Las Ceibas watershed in Colombia as a case study, the buffer-overlay-statistics (BOS) method is used to estimate the accuracy and displacement of river channel networks derived from DEMs with pixel sizes of 30m, 50m and 90m vis-à-vis those digitized from 1:25,000 scale cartographic maps. The corresponding runoff and sediment estimates are obtained using the Soil and Water Assessment Tool (SWAT) hydrological model. This study demonstrates that very high resolution DEMs are not always necessary to obtain accurate hydrological analysis and acceptable results can be obtained with the widely available 90m DEM.

Keywords: scales, GIS, hydrological modeling, SWAT, DEM.

3.2 Introduction

Social and natural scientists acknowledge the importance of scale and how relationships and processes can vary at different scales, including hydrological features across landscapes (Blöschl & Sivapalan 1995; Montgomery & Dietrich 1992). Literature addressing these issues and relating to GIS can be found in biology (Bian 1997), geomorphology (Walsh et al. 1998), social sciences (Evans et al. 2003; Gibson et al. 2000), hydrology (Blöschl & Sivapalan 1995) and landscape ecology (Jelinski & Wu 1996). Given that interaction between disciplines is increasing and due to the fact that larger and more detailed datasets are available at different levels of scales (due mainly to the increasing number of remote sensors and platforms), it is important that modeling processes deal with data at multiple scales and produce results that suit the decision making processes at different scales. (Atkinson & Tate 2000; Goodchild & Quattrochi 1997; Walsh et al. 1998).

In the field of hydrological modeling, there have been several attempts to develop predictive models of physical processes that vary across scales and that integrate datasets at different scales. In the 1990s there was increased interest in the development of hydrological models coupled with Geographical Information Systems (GIS) that integrated many processes such as infiltration, evaporation, overland flow, sediment transport and subsurface water movement, that were traditionally studied empirically at the local level (Arnold et al. 1995; Di Luzio et al. 2004; Maidment 1993; Young et al. 1989). This integration and availability of the hydrological models and GIS, coupled with the wide spread availability of data at multiple resolutions, posed new challenges in researching hydrological processes. In particular, the resolution of a Digital Elevation Model (DEM) can significantly affect the results of a hydrological simulation (Clarke & Lee 2007). This paper addresses the impact of DEM resolution on river channel network delineation and simulations of flow and sediments using the Soil and Water Assessment Tool (SWAT).

3.2.1 The Concept of scale

The term scale is used differently across disciplines and it is often used to refer to the extend of a study area, to the degree of detail, to the thematic scale or to the temporal scale. Gibson et al. (2000) define scale as the spatial, temporal, quantitative and analytical dimensions used to measure, observe and study any phenomenon. Cao & Lam (1997) list four meanings of scale within the spatial domain:

1. *Cartographic or map scale* is the ratio of a distance on a map to the corresponding distance on the ground. A large scale map covers a small area and vice versa. For example a map with a scale of 1:10,000 is more detailed than a map with a scale of 1:250,000. Since the scale in this case is expressed as a ratio it is said that 1:10,000 scale is larger than 1:250,000 scale.
2. O*bservational scale* is the spatial extent of the study area.
3. *Operational scale* refers to physical size or range over which different processes happen.
4. *Measurement scale* is the scale of the collected data (e.g. grain and extent).

Temporal scale is an issue still unsolved in GIS. Studies in the field of spatio-temporal GIS applications are numerous, but progress related to the inclusion of temporal scales has been lagging behind with respect to spatial scale issues (Frank et al. 2000). Most of the phenomena have space and time interactions that are inseparable and cannot be reduced to a set of spatial data independent of time or vice versa. Models using different spatial databases describe scenarios that often require different data to be compatible and comparable in specific temporal dimensions.

Blöschl and Sivapalan (1995) defined three characteristic temporal scales in hydrological processes: lifetime or duration (e.g. for intermittent processes such as a flood), period or cycle (e.g. for a periodic process such as snowmelt) and correlation length (e.g. for a stochastic process exhibiting some sort of correlation). Although some hydrological processes exhibit preferred scales, these definitions for temporal scale are often used interchangeably and sometimes it is not clear which one is used.

One of the main questions that a researcher or policy maker might need to answer before starting any project is what are the appropriate scales to study a particular phenomenon and draw relevant conclusions for planning and decision making? (Atkinson & Tate 2000; Marceau 1999). Answers regarding the selection of a particular scale will depend upon the scale of the system under study and the overall aims of the research. Other questions related to scales posed by Gibson et al. (2000) are: (1) How does scale affects the identification of patterns and problems; (2) How do diverse levels of a scale affect the explanation (causality) of observed patterns; and (3) How may propositions at one level of a scale be generalized at another level.

3.2.2 Scales and hydrological modeling

Outputs from physical hydrological models depend on the quality and accuracy of the input data. While the increasing availability of DEMs at multiple resolutions has revolutionized hydrology, it presents several scales problems (Sivapalan & Kalma 1995). Such problems are related to hydraulic attributes of surfaces across the landscape (Beven 1995; Viney & Sivapalan 2004), optimal thresholds to extract channel network (Heine et al. 2004), and significant topographic variability and errors with respect to the DEM resolution (Carlisle 2005). For example, the resolution of the DEM will affect the delineation of the river channel network and watershed boundaries, which consequently will affect the model's outputs.

Some references exist addressing the question of accuracy of topographic datasets and hydrological model outputs (Blöschl & Sivapalan 1995; Chaplot 2005; Chaubey et al. 2005; Cho 2000; Cotter et al. 2003). Blöschl and Sivapalan (1995) suggested that high resolution DEMs are required when local processes are dominant. Chaubey (2005) found that the watershed delineation, stream network and sub-basin classification obtained with a specific hydrological model (i.e., Soil and Water Assessment Tool -SWAT) greatly varied by changing the DEM grid sizes. In the same study, a decrease in DEM resolution resulted in decreased stream flow and nitrate load predictions. Chaplot (2005) contradicts the common agreement that DEM accuracy affects hydrological simulations. Chaplot's study showed that the computed runoff is not substantially affected if the DEM resolution is increased beyond 50 m. Cotter, et al. (2003) used six sets of DEM and soil maps of different scales from 50m to 500m. They found that the SWAT model was most affected by the DEM data resolution. Cho (2000) carried out a sensitivity analysis of runoff results using DEMs with two different resolutions, 1:24,000 and 1:250,000, released by the U.S Geological Survey. The runoff volume obtained with SWAT was higher for the 1:24,000 DEM; whereas the runoff volume obtained with 1:250,000 DEM was lower compared to the measured values. The literature indicates that the scale and accuracy of the input data such as DEMs affect the results of spatially distributed models. However, studies indicate that the vertical and horizontal accuracy of the DEMs should be enough to capture significant topographic variability within the watershed to get enough detail of sub-basins delimitation and to estimate runoff volumes.

In particular, the worldwide availability of the 90m DEM (Jarvis et al. 2004) obtained from the Shuttle Radar Topography Mission (SRTM) raises the question: is it sufficient for most hydrological modeling exercises, particularly in the developing world where spatial data are scarce? This dataset has a stated vertical accuracy of $\pm$ 16m at the 90% confidence level, although other studies have shown that the accuracy can be much better, e.g., ranging from 7.58 ± 0.60 m in Phuket, Thailand, to 4.07 ± 0.47 m in the Catskills mountains of New York, USA (Gorokhovich & Voustianiouk 2006). Further, relatively simple methods can be used to effectively delineate river networks that correlate well with real topographic features, regardless of landform type (Matsunga et al. 2009). Given these studies, our goal was to compare the accuracy and displacement of the river network, as well as discharge estimates, obtained from using a widely-available 90m DEM (CIAT 2004) to estimates obtained from finer resolution DEMs (50m) or large cartographic scale (1:25,000) maps.

3.3 The Soil and Water Assessment Tool (SWAT)

In general terms, hydrological models can be grouped in three categories: empirical, conceptual and physically based models, depending on the physical processes simulated, the model algorithms describing these processes and the data dependency of the model (Saavedra 2005). Empirical models are based on the analysis of field experiments and estimates are done using statistical inference. The limitation of applying empirical models at catchment level is the assumption of stationarity, which assumes that underlying conditions do not change during the simulation period (Kandel et al. 2004). Conceptual models use empirical relationships and represent the catchment as a series of linear storages (Sivapalan et al. 2002). Parameter values for this type of model are usually obtained through calibration against observed data, which could represent a problem for ungauged watersheds (Zhou & Liu 2002).

The Soil and Water Assessment Tool (SWAT) is a physical distributed model coupled with GIS that has been developed by USDA Agricultural Research Service (Di Luzio et al. 2004). Physically based models are based on equations that represent physical processes related to transfer of mass, momentum and energy (Kandel et al. 2004; Merritt et al. 2003). Equations are used with continuous temporal and spatial data, although in practice the data used are often point source data that represent a unit of area. This is one undeniable criticism of these models, as physical significance of these equations may be lost when using parameters at small scale (Saavedra 2005). This issue is related to the aggregation and scale effects problem.

SWAT is used to predict effects of different land management practices in water and soils resources over large watersheds of more than 100 km^2 (Neitsch et al. 2002a). SWAT requires information about weather, soil properties, topography, vegetation, and land management practices occurring in the watershed, and can consider long-term effects. SWAT has been developed since the early 1990s and it is widely used by environmental agencies and research institutes. The model simulates the hydrological cycle by coupling the land and water phases. The former determines the amount of water and nutrients that reach the river network of each sub-basin depending on the soils and land use. The latter, which is also called the routing phase, determines the movement of water within the channels down to the outlet of the watershed.

Several sources of modeling uncertainties result in the fact that model predictions are not a certain value, but should be represented with a confidence range of values ((Wagener & Gupta 2005;Beven, 1993). These sources of uncertainty are often categorized as model structure/model hypothesis uncertainties (uncertainties caused by inappropriateness of the model to reflect reality or the inability to identify the model parameters), data structural uncertainties (such as errors in rainfall or pollutant sources inputs), and parameter estimation uncertainties (see Section 2.3.2). Uncertainties regarding input data in SWAT model can be divided in (1) spatial inputs and (2) environmental observations. Spatial inputs such as land use map, soil map and topographic map are sources for errors on the spatial discretisation of land use, soil, and topography. They are also sources of errors on parameters for land use, soil, and topography. Environmental observation such as rainfall series and other weather related variables are sources of errors on observed values (e.g. errors regarding how they are measure, how they are recorded and how often).

SWAT utilizes single-threshold flow-accumulation values to represent stream networks (Arnold et al. 1993). The model uses the D8 algorithm, which takes into account the elevation of the adjacent cells and the distance between the centers of cells to calculate the flow path and accumulation. D8 has been widely used to calculate sub-watershed delineation in semi-distributed modeling (Tarboton et al. 1991). The single threshold area method assumes that channel sources represent the transition between the convex profile of the hillslope (sheet flow dominated) and the concave profiles of the channel slope (channel discharge dominated).

3.4 The study site

The 258 km^2 of Las Ceibas watershed served as a case study, which is an agricultural basin located in South America, Colombia, in the Andes Mountains (Figure 3.1). The nine main land cover and vegetation types of the watershed are (CAM 2005): grasslands and shrublands (27.6%), managed grasslands (1.9%), natural forests (7.3%), natural grasslands (31.0%), natural shrublands (12.8%), perennial crops (8.2%), seasonal croplands (0.5%), secondary forests (9.5%) and uncultivated land (1.2%). The ecosystem of the upper catchments is cloud forest, which has been primarily disturbed by agricultural development. There are advanced erosion processes going on in the watershed, from sheet erosion, rill and gully erosion. The causes of these processes are a combination of natural and anthropogenic factors. The watershed has gone through a change of land uses from natural forests and shrublands to agriculture and extensive cattle raising. In the last 20 years forest cover has been reduced by 1,000 ha approximately and the annual flow has been reduced from 5.5 m^3/s in the 1980s to 4.1 m^3/s in the 1990s. The erosion processes in the watershed are causing major sedimentation problems in water supply systems downstream, where in the rainy season the sediments have reached 127,000 Ton/year. An accurate hydrological model would help to evaluate possible alternatives to improve the current conditions in the watershed.

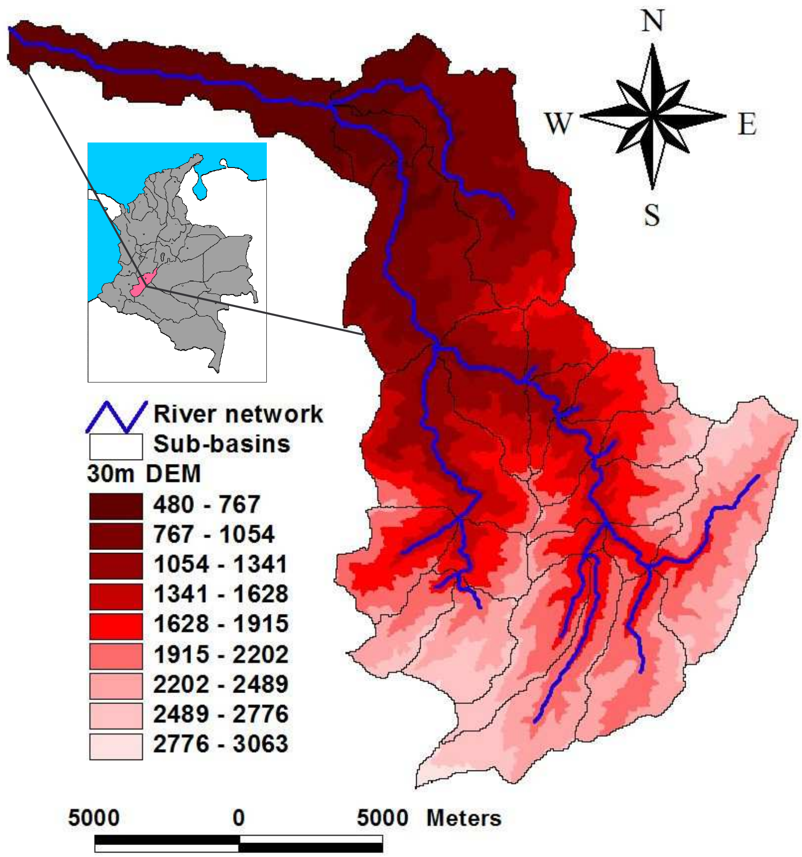

Figure 3.1. Las Ceibas watershed.

3.5 BOS Method

The buffering method BOS (buffer-overlay-statistics) was used to estimate the accuracy and displacement of the river network delineation (Tveite & Langaas 1999). The BOS method is based on a comparison of a line dataset of unknown quality to another of known or higher quality. This method was also used to compare four river networks that were calculated with different datasets and models for the Baltic Sea region (Miranda 2001). In this case, river networks obtained with the 90m DEM (CIAT 2004), 50m DEM (CAM 2006) and 30m DEM (ERSDAC 2009) are compared with a river network digitized from 1:25,000 scale cartographic maps (CAM 1984). The 90m DEM was obtained from the Shuttle Radar Topography Mission (STRM) and processed by the Land Use Team at International Center for Tropical Agriculture (CIAT). The 50m DEM was processed by the institution Coorporacion del Alto Magdalena (CAM), which was obtained from digitalizing 1:100,000 cartographic maps. These maps were obtained from Instituto Geográfico Agustín Codazzi (IGAC). The 30m DEM was obtained from the Advanced Spaceborne Thermal Emission and Reflection Radiometer (Aster) provided by the Earth Remote Sensing Data Analysis Center (ERSDAC) processed by CIAT. The 30m DEM was also aggregated to 50m and 90m resolutions in order to compare (i) results obtained with DEMs from the same origin in this case 30m, 50m and 90m DEMs from ERSDAC and (ii) with the DEMs with same resolution but different sources as 90m SRTM or 50m cartographic maps.

For the Las Ceibas region, the assumption was that the river network obtained from the 1:25,000 scale topographic maps are most accurate. First, buffers with different widths, varying from 10 m to 400 m, were generated around both linear datasets for the river networks obtained with the 90m, 50 m and 30 m DEMs. The river network obtained with 90m, 50 m and 30m DEMs are denominated as Y, X and Z respectively. The buffer around Y is YB_i, the buffer around X is XB_i and the buffer around Z is ZB_i . The buffer size will be called bs_i. Then the buffers are overlaid and the result is the area within buffers ($ZB_i \cap YB_i$) and ($ZB_i \cap XB_i$) with respect to the buffer area of the river network that is considered more accurate (ZB_i). The area for each of the buffers YB_i XB_i, and ZB_i are calculated together with the corresponding overlapped area ($ZB_i \cap YB_i$) and ($ZB_i \cap XB_i$). The accuracy curves are obtained by plotting the percentage of the overlapped areas.

Another result that can help to complement the relative differences between DEM is the average displacement. This curve represents the average displacement of the line of unknown quality relative to the line data of higher accuracy. On the x-axis the buffer size is plotted versus the average displacement, in the same length units as the buffer size.

The formula for the 50m and 90m DEM is as follows:

$$D_i = \frac{\pi}{2} 2bs_i \frac{Area\left(\overline{YB}_i \cap XB_i\right)}{Area\left(YB_i\right)} = \pi bs_i \frac{Area\left(\overline{YB}_i \cap XB_i\right)}{Area\left(YB_i\right)} \quad \text{Equation 3.1}$$

Where, D_i is the average displacement in meters. The term $\pi / 2$ is a correction term to account for underestimation, assuming an even distribution of line directions and errors equal to $2 / \pi$. The term $2bs_i$ is the total width of the buffer, taking into account that the buffer will be generated on both sides of the line data. Equation 3.1 was also applied to find the average displacement of the 90m DEM compared with the 30m DEM.

3.6 **Results**

3.6.1 Effect of the DEM resolution on SWAT watershed delineation

According to the DEMs, the watershed is stretched out in a south-north direction and showed a flatter terrain in its north-west part, whereas the south-east area exhibited steeper slopes. The area of the watershed estimated with the 50m-CAM DEM and 90m-SRTM DEM was 1.2% and 2.4% lower than the area estimated with the 30m-ASTER DEM (Table 3.1). Regarding the results obtained with the ASTER DEMs, the differences between resolutions were of 2.3% and 1.2% of the 90m and 50m DEMs with respect to the 30m DEM. Two sub-basins were selected to investigate if the difference of areas obtained with the two DEMs was consistent for watersheds with different ranges of slope. Although there is a degree of bias on the selection of these sub-basins, the selection was done between upper and lower catchments to minimize this bias. The area of the upper sub-basin calculated with the 50m DEM and 90m DEM was 0.9% and 1.9% lower than the one calculated with the 30m DEM. For the lower sub-basin, the area estimated with the 50m DEM and 90m DEM was lower 2.8% and 21.5% respectively than the one estimated with the 30m DEM (Table 3.2).

Table 3.1. Catchment characteristics estimated with five sets of DEMs.

	30 m (ASTER)	50 m (CAM)	50 m (ASTER)	90 m (SRTM)	90 m (ASTER)
Altitude					
Min	440	440	440	486	440
Max	3,080	3,085	3,080	2,970	3,080
Mean	1,300	1,300	1,300	1,318	1,300
Slope (%)					
Min	0	0	0	0	0
Max	72.84	72.63	71.09	48.20	57.30
Mean	42.02	41.59	40.03	38.02	39.67
Reaches length(Km)	92.0	91.1	90.2	83.7	87.5
Area (Km^2)	261.5	258.4	255.3	255.3	256.4

Table 3.2. Catchment characteristics estimated with ASTER 30m, CAM 50m and SRTM 90m DEMs.

	Entire watershed			Upper watershed			Lower watershed		
	30 m	50 m	90 m	30 m	50 m	90 m	30 m	50 m	90 m
Altitude									
Min	440	440	486	1,785	1,783	1,675	440	440	486
Max	3,080	3,085	2,970	2,763	2,760	2,800	925	920	878
Mean	1,300	1,300	1,318	2,445	2,432	2,416	582	580	600
Slope (%)									
Min	0	0	0	1.02	1.07	3.1	0	0	0
Max	72.84	72.63	48.20	52.74	52.52	48.37	50.87	50.11	45.44
Mean	42.02	41.59	38.02	49.07	48.93	41.86	9.65	9.75	7.48
Reaches length km	92.0	91.1	83.7	6.93	6.90	6.43	27.96	28.15	15.20
Area (Km^2)	261.5	258.4	255.3	10.1	10	9.9	27.1	26.32	21.23

3.6.2 Effect of the DEM resolution on SWAT river network delineation

Through visual inspection of the three river networks it was possible to detect a higher number of reaches which were not correctly placed in the lower area of the watershed compared to those that were estimated with the 30m-ASTER DEM. The length of the reaches estimated with the 30m-ASTER DEM was 27% higher than the length estimated with the 90m DEM. The length of the reaches with the 50m-CAM DEM was 25% higher than the one obtained with the 90m-SRTM DEM. The higher the resolution of the DEMs, the more heterogeneity is captured in the elevation, which is translated into the estimation of the reaches. In effect, if the resolution of the DEM is higher, then the lengths of the reaches are longer.

The BOS method was used to calculate the accuracy curve for the 90m-SRTM DEM, 50 m-CAM DEM and 30m-ASTER DEM river networks compared to that from cartographic maps with an scale of 1:25,000 (Figure 3.2). From the slope of the curve, on the left side, it is possible to infer the degree of accuracy. According to Tveite (1999) the reason for this consideration is that a higher percentage of area in both buffers, with buffer sizes closer to zero, means a higher similarity of both data sets. The method did not specify a range for the accuracy degree. In this case the slope of the trend line is higher than 45 degrees, which can be interpreted as adequate. With this model it is not possible to compare only those reaches that are drawn in a similar way in both DEMs, and exclude those reaches that are completely misplaced due to the lack of topographic heterogeneity. In other words the model takes into account errors such as those drawn in lower areas, where the variation in altitude is small and the elevation resolution is not enough to draw the correct flow path.

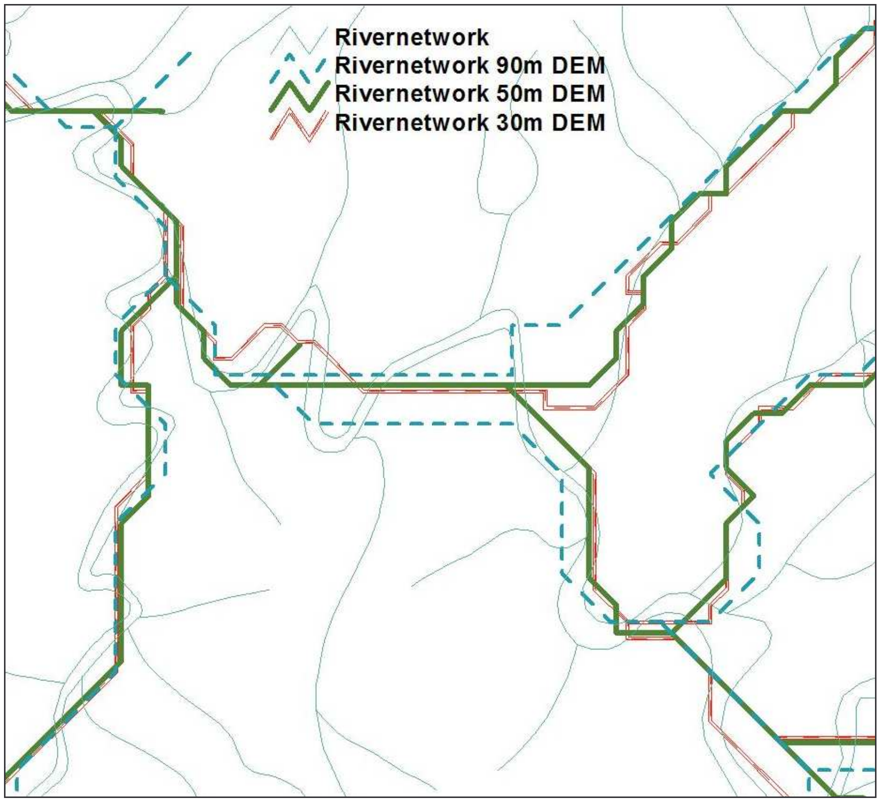

Figure 3.2. Comparison of river network delineations obtained from DEM of 90m-SRTM, 50m-CAM and 30m-ASTER with cartographic river network.

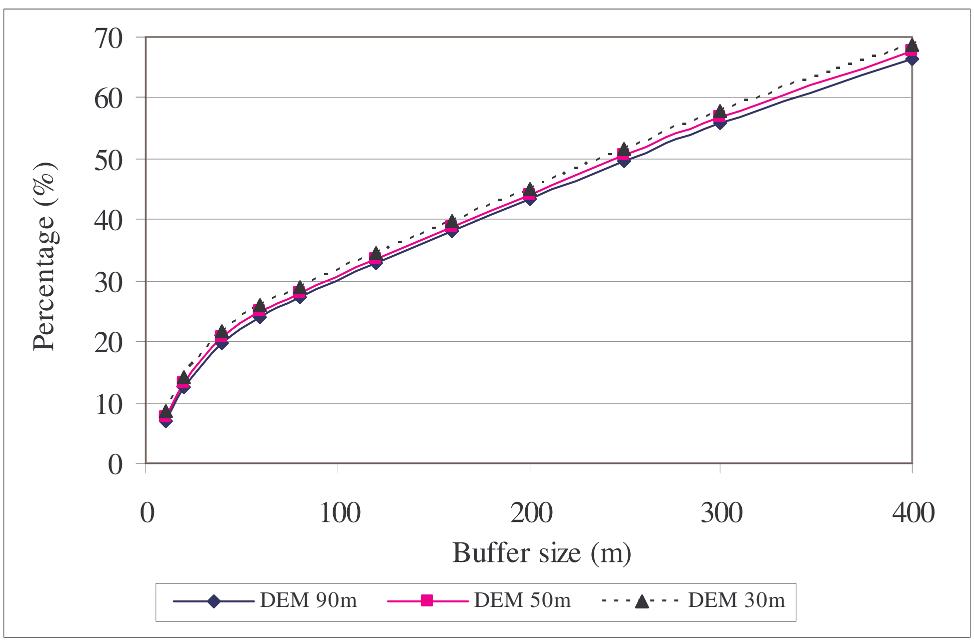

Figure 3.3. Accuracy curve for the river network delineation with the 90m-SRTM, 50m-CAM and 30m-ASTER DEMs compared to river network from 1:25,000 cartographic maps.

Figure 3.3 shows that the accuracy of the river delineation obtained with the 90m-SRTM, 50m-CAM DEM and 30m-ASTER DEM compared with the one obtained with cartographic maps. The average displacement (Figure 3.4) shows the same tendency as the accuracy curve but not in the last section, between the buffer size of 60 m and 400 m, where the slope is negative. This change indicates that for buffer sizes bigger than 60 m the displacement of the river network obtained with the three DEMs decreases.

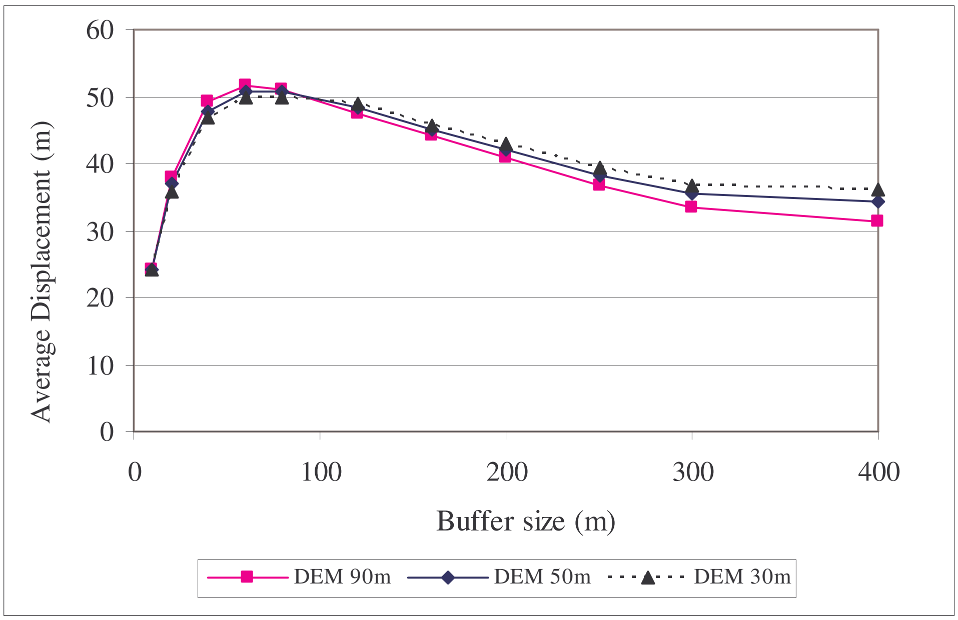

Figure 3.4. Average displacement curve for the river network estimated with the 90m-SRTM, 50m-CAM and 30m-ASTER DEMs and the river network from cartographic map.

3.6.3 Sensitivity of the Model Parameters and validation approach

The sensitivity of the model parameters to different DEMs was tested using the SWAT2005 sensitivity analysis tool. This tool combines the One Factor at a Time (OAT) design and Latin Hypercube sampling (Van Griensven 2005). The Latin-Hypercube simulation is based on the Monte Carlo Simulation but applies a stratified sampling approach requiring less number of simulations. The sensitivity analysis tool determines the relative ranking of which parameters most affect the output variance due to input variability. The model results are analyzed with multi-variate linear regression or correlation statistic method. This tool also contains an auto-calibration procedure that is used to obtain an optimal fit of process parameters and it is based on a multi-objective calibration, which incorporates the Shuffled Complex Evolution Method algorithms. Multiple output parameters can be simultaneously evaluated through a global optimization criterion that allows the aggregation of objective functions for individual variables. A statistical method uses the fit of the observed series to its related calculated series and translates the normalized values of the objective functions per variable. These objective functions are then aggregated to a single global criterion determined by optimal fit of the maximum Nash and Sutcliffe value, which considers all of the participating variables rather than by means of a weighted sum.

The method required 369 simulations of the model to determine the sensitivity for flow and sediment loads for each DEM. The results indicate that sensitivity of the parameters did not change between DEMs. The most sensitive parameters for flow were: curve number, available water capacity, soil evaporation compensation factor, surface runoff lag coefficient and baseflow alpha factor. For sediments load estimation, the most sensitive parameters were: curve number, maximum canopy storage, and baseflow alpha factor (Table 3.3).

Table 3.3. Auto-calibrated parameters selected for discharge and sediment of the SWAT model for the Las Ceibas watershed

Parameter	Description	Range
CN2	Initial SCS runoff curve number to moisture condition II	31-93
SOL_AWC	Available water capacity of the soil layer (mm/mm)	0.05-0.50
FFCB	Soil evaporation compensation factor	0.01-1.0
SURLAG	Surface runoff lag coefficient (days)	0-4
ALPHA_BF	Baseflow alpha factor (days)	0-1
CANMX	Maximum canopy storage (mm H_2O)	

The model was calibrated and evaluated for flow and sediment loads using the 50m-CAM DEM and 30m-ASTER DEM. The observation data were divided in two periods, with data from 1984 to 1993 used for calibration and data from 1994 to 1999 used for evaluation. The Nash and Sutcliffe Efficiency (NSE) was used to quantitatively describe the accuracy of model outputs and it corresponds to a perfect match of modeled discharge to the observed data. In this model, the NSE varied from 0.61 to 0.82 for the daily flow, and from 0.69 to 0.89 for monthly flow. For the sediments, Nash and Sutcliffe efficiency varied from 0.58 to 0.76 for daily loads, and from 0.63 to 0.80 for monthly loads (Table 3.4). The efficiency value of 0.7 is usually considered as sufficiently good for hydrological evaluation, and 0.6 as satisfactory (Krysanova et al. 2007). Although higher NSEs were obtain with the 30m-ASTER DEM, between 0.01 and 0.02, the NSEs obtained with the 50m-CAM are still within acceptable ranges. Following the calibration, the model was run with the 30m, 50m and 90m DEMs to estimate scale effect due to lower DEM resolutions.

Table 3.4. Nash and Sutcliffe Efficiency using the 30m-ASTER DEM and 50m-CAM.

Flow	NSE	Sediments	NSE
Daily Flow 30m-ASTER	0.63-0.82	Daily loads 30m-ASTER	0.62-0.76
Daily Flow 50m-CAM	0.61-0.82	Daily loads 50m-CAM	0.58-0.74
Mean Flow 30m-ASTER	0.70-0.89	Mean loads 30m-ASTER	0.63-0.80
Mean Flow 50m-CAM	0.69-0.88	Mean loads 50m-CAM	0.63-0.79

3.6.4 Effect of the DEM resolution on SWAT flow

The flow obtained with SWAT simulations over the evaluation period from 1994 to 1999 using the 30m, 50m and 90m DEMs was compared with the measured daily and monthly flows. Figure 3.5 presents a comparison of the daily flow for the four datasets from 1994 to 1996. The %Bias is defined as the relative percentage difference between the average simulation and measured data time series over n time steps and is given by (Tolson & Shoemaker 2007):

$$\%Bias = \frac{100 \times \left(\sum_{j=1}^{n} Simulated_j - \sum_{j=1}^{n} Measured_j \right)}{\sum_{j=1}^{n} Measured_j} \quad \text{Equation 3.2}$$

The %bias obtained was higher with the 90m DEM than the 30m and 50m DEMs (Table 3.5). This indicates that the results obtained using the 30m-ASTER and 50m-CAM DEMs were closer to the measured data. The %Bias obtained after aggregating the 30m-ASTER DEM to 50m and 90m were better compared to the 50m-CAM and 90m-SRTM respectively. These results could be explained due to the vertical accuracy and characteristics of the datasets.

Table 3.5. Mean measured and simulated flow at Guyabo station for the calibration period from 1994-1999.

Scenario	Mean Flow (m^3/s)	%Bias
Mean Measured Flow at Guayabo station	4.68	-
Mean simulated Flow with 30m-ASTER DEM	4.58	2.1
Mean simulated Flow with 50m-ASTER DEM	4.55	2.8
Mean simulated Flow with 90m-ASTER DEM	4.48	4.3
Mean simulated Flow with 50m-CAM DEM	4.53	3.2
Mean simulated Flow with 90m-SRTM DEM	4.41	5.8

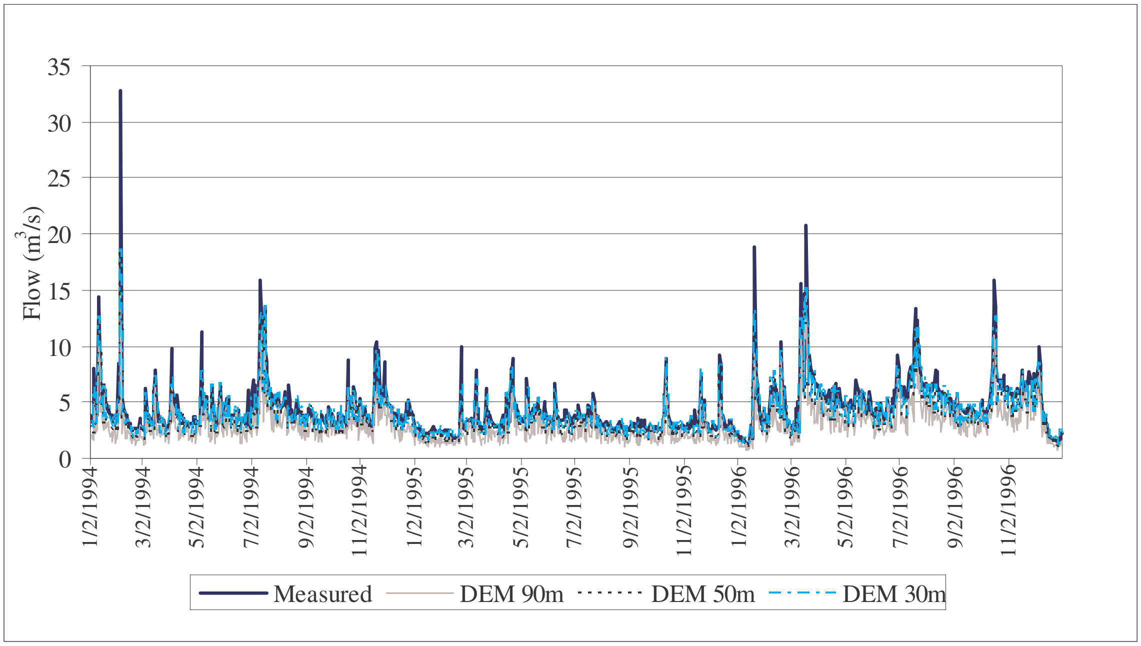

Figure 3.5. Daily flow measured at Guayabo station and the estimated flow with the 90m-SRTM, 50m-CAM and 30m-ASTER DEMs.

With respect to monthly flow, the biggest differences were obtained during the months of March and September (Figure 3.6). In these months, flow estimations with the 30m-ASTER DEM were 7.00% lower in March and 10.2% higher in September compared with the measured flow. The flow estimations with the 50m-CAM DEM were 7.90% lower in March and 10.0% higher in September against the measured flow. Flow estimations with the 90m-SRTM DEM were 11.2% lower in March and 12.3% higher in September. The flow pattern over the year obtained with the three DEMs was smoother than the measured flow. One of explanation of the smoother flow pattern during the year could be the aggregation effect. SWAT aggregates results from physical equations at sub-basin level. Although the simulations were done with the maximum number of sub-basins allowed by the model, during that aggregation process, some degree of heterogeneity could have been lost.

Runoff volume was higher for 30m-ASTER DEM, probably due to the finer resolution and slope which increased the estimated runoff from the watershed. That is, as the slope of the watershed flattens (90m DEM), the response of stream flow was delayed and resulted in reduced runoff volume. The 30m DEM resulted in higher runoff discharges, whereas a flatter DEM resulted in lower rise and lower runoff discharge. These results were somewhat consistent with findings made by Chaubey (2005) and Chaplot (2005).

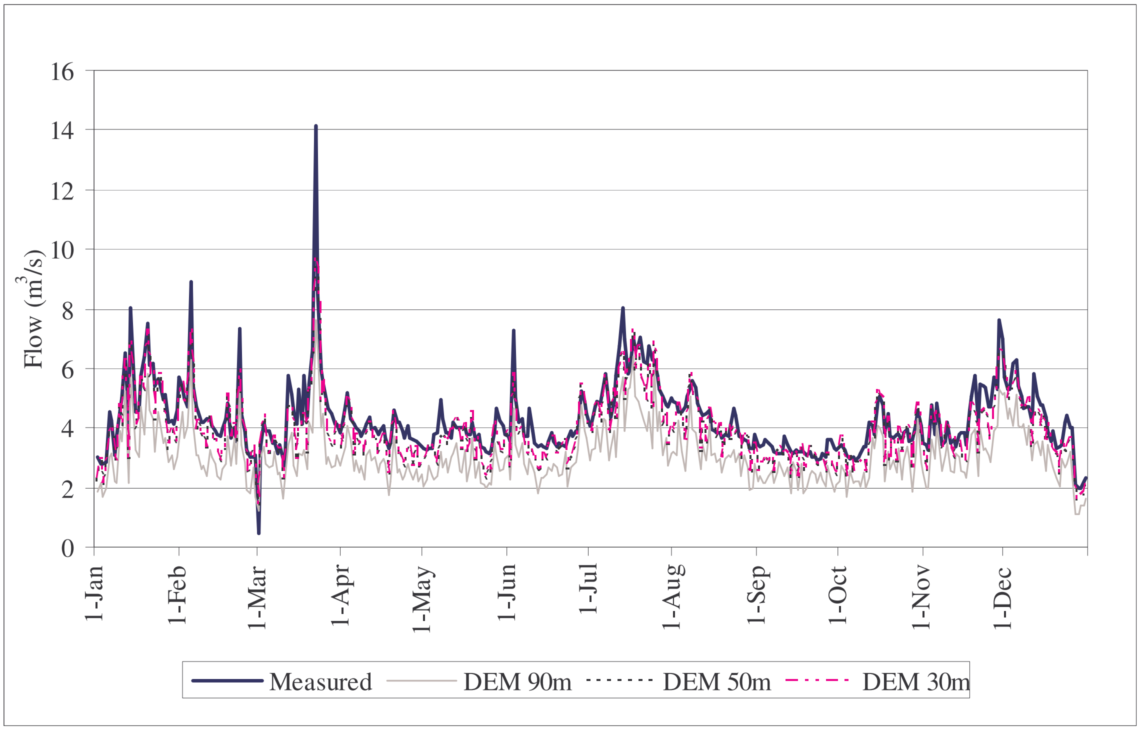

Figure 3.6. Mean daily flow at Guayabo Station and the estimated flow with the 90m-SRTM, 50m-CAM and 30m-ASTER DEMs.

3.6.5 Effect of the DEM mesh size on SWAT sediments

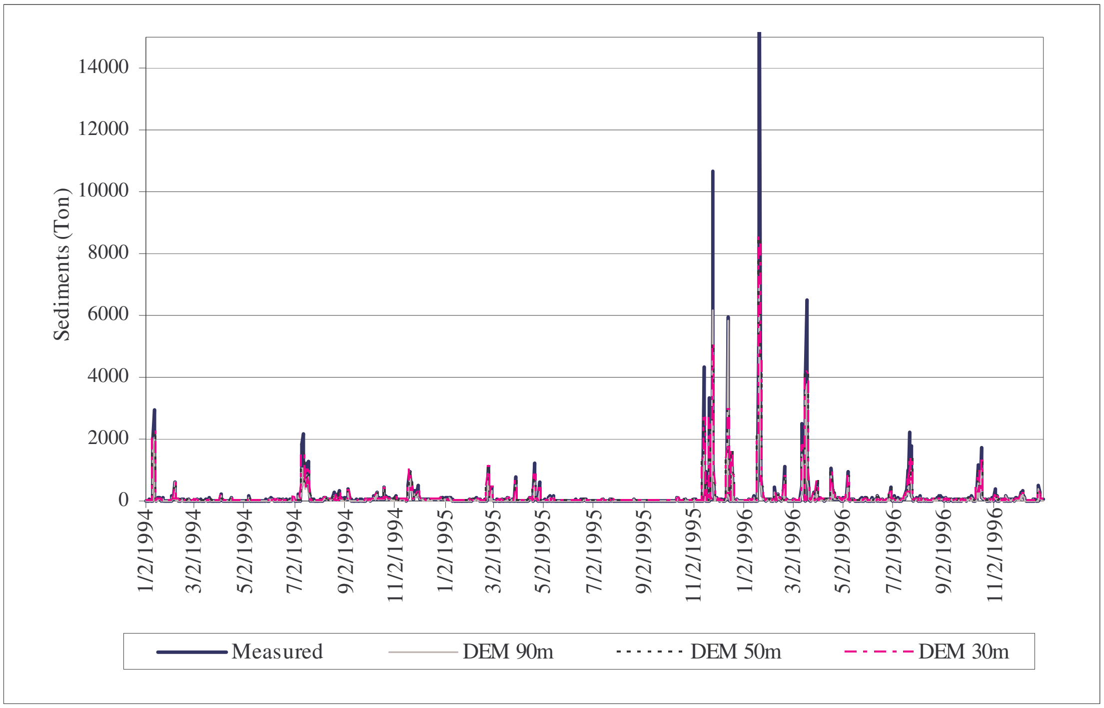

Figure 3.7. Daily sediment load at Guayabo Station and the estimated sediment load with the 90m-SRTM, 50m-CAM and 30mASTER DEMs in the evaluation period 1994 - 1996.

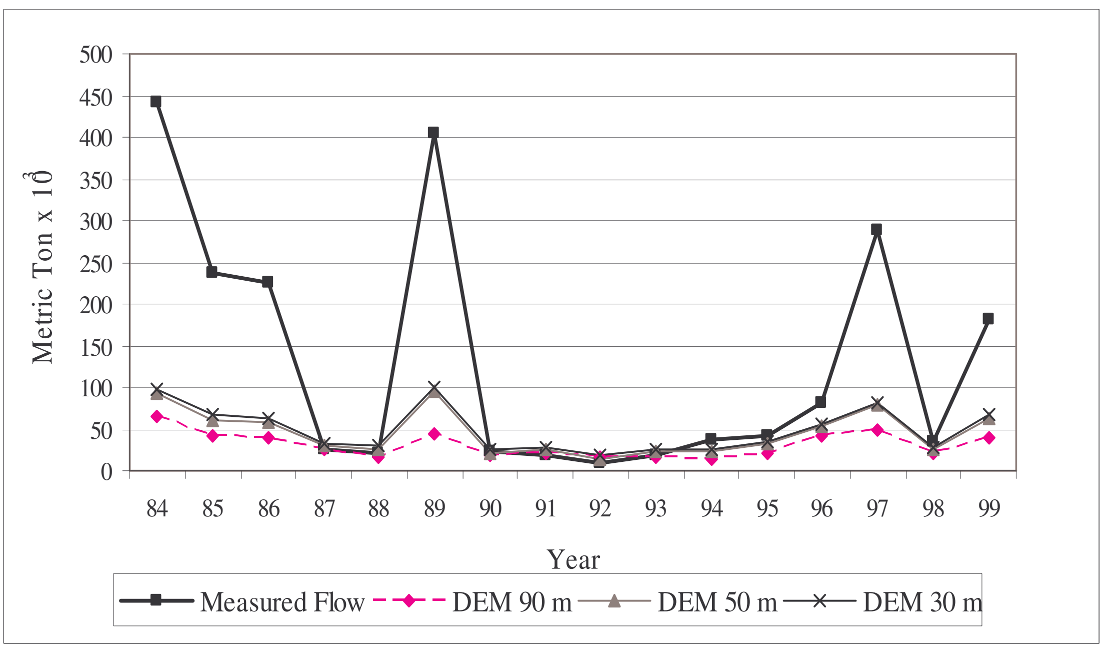

Figure 3.8. Annual sediment load measured at Guayabo Station and the estimated load with the 90m-SRTM, 50m-CAM and 30m-ASTER DEMs from 1984 to 1999.

The mean annual sediments yields measured at the Guayabo station from 1984 to 1999 was 5.04 Ton/ha, while the estimations obtained with SWAT were 1.89 Ton/ha, 1.75 Ton/ha, and 1.18 Ton/ha with the 30m-ASTER, 50m-CAM and 90m-SRTM DEMs respectively (Figure 3.7). As shown in Figure 3.8, the peak events were not well simulated by SWAT.

3.7 Conclusions

Distributed hydrological models like SWAT are widely used to evaluate flow and water quality response of a watershed, therefore quality of the input data should be crucial to reduce model uncertainty. The DEMs used were adequate to capture significant topographic variability for watershed and river network delineation for the Las Ceibas watershed. Although the delineation obtained with the 90m-SRTM and 50m-CAM DEM was less accurate than the results obtained with the 30m-ASTER DEM, these resolutions reflected the real topographic features in the landscape. Obviously, the usefulness of these results will depend on both the final purpose of the model and the accuracy needed for development, implementation or project design. In this case, where the watershed has a great slope variation, both DEMs were able to capture the water flow patterns. Through visual inspection and as can be expected, it was possible to detect fewer errors in the river network delineation at steeper terrains than in lowlands. Similar results were obtained when comparing the watershed areas for sub-basins in the upper and lower part of the catchment.

Simulation results over a 27 year period showed that flow predictions at the outlet of the 258 km^2 Las Ceibas watershed were overestimated during seasonal peak flow, but were adequate regarding the mean annual flow. The difference of bias regarding the mean flow obtained with the 30m-ASTER and 50m-CAM DEMs was just 1.2%, indicating that the output was not significantly affected by the resolution variation. Given that the model variables remained the same for the three simulations, with 30m-ASTER, 50m-CAM and 90m-SRTM DEM, it is possible to conclude that there is a negligible effect on watershed delineation and flow due to the DEM mesh sizes. The results showed that aggregating the

30m-ASTER DEM to 50-CAM and 90-SRTM resulted in an decreased contributing area and decreased slopes. DEM resolution also affected hydrologic response significantly, and with the increasing DEM grid size, the simulated peak discharge and sediment loads decreased. However, at no point was bias from the measured flow greater than 5.8% for the 90m DEM.

Although it is desirable to have as much detail as possible when using distributed hydrological models, the cost of the DEM determines the resolution of the data used. Given scarce data availability in the developing world and the high costs associated with acquiring high resolution data, it is cost effective to use 90m and 50m DEMs and pay more attention to other sources of variation in the hydrological model, such as land cover and soil variables.

The results obtained in this study demonstrated that very high resolution DEMs are not always necessary to obtain accurate hydrological analysis for basins like Las Ceibas, which has great slope variation. Moreover, improved DEM resolution does not improve estimations of sediment loads during peak flows. However, if necessary, more efforts should be made to collect data at finer resolution for watersheds located in lowlands. Further research is needed to investigate the sensitivity of the model to different DEM resolutions for watersheds with low slope variation.

Chapter 4

4 Integrating environmental modeling towards bundling of environmental services in an Andean watershed

4.1 Abstract

Investments in watershed management and ecosystem conservation are of increasing importance in both temperate and tropical regions due to the accelerated reduction in environmental services. There is a critical need for estimating environmental services synergies and tradeoffs emerging from land use changes in order to enhance bundled ecosystem services available from a watershed, as well as methods to quantitatively evaluate these linkages that could guide its management, policy-making and investments. The SWAT model and the CO2Fix model were used to determine the biophysical provisioning of erosion control, discharge regulation and carbon sequestration in an Andean tropical watershed. Maps of the change in the provisioning of services for a period of fifty years were obtained at the basin level and the hydrological response unit level. Three scenarios were simulated: rotational grazing and reforestation of grasslands, green manures and cover crops of grassland and perennial crops. Results indicate that reforestation, which mainly targets increased carbon sequestration, also improved the provisioning of hydrological services, through improved erosion control and discharge regulation. Discharge regulation improved more with cover crops and green manures than with rotational grazing due to improvement in water infiltration. Given the conservation nature of the scenarios analyzed, positive correlations were found between environmental services in the three scenarios. Environmental modeling proved to be a useful tool in making evident synergies resulting from the bundling of environmental services, and help to explore the feasibility of the implementation of a particular scenario.

Keywords: environmental modeling, environmental services, hydrological services, carbon sequestration, Andes, SWAT.

4.2 Introduction

According to the Millennium Ecosystem Assessment, ecosystem services are the benefits people obtain from ecosystems. They can be classified as (1) provisioning services, such as food, timber and water; (2) regulating services, such as water regulation and disease control; (3) supporting services, such as soil formation and nutrient cycling; and (4) cultural services, such as recreational, spiritual, religious and other non-material benefits (MEA 2005). There is no agreement regarding the concept of environmental services but in general the term refers to provisioning, regulating and supporting cultural ecosystem services as outlined in the Millennium Assessment's definition. This research will be focused on hydrological services, related to the provisioning and regulation of water, carbon sequestration and food production.

There is an increasing interest in environmental benefits provided by ecosystems under threat. According to the Millennium Ecosystem Assessment (2005), environmental services have reduced 60% globally. For example, during the 1990s, it is estimated that approximately 15 million hectares of forest globally were cleared and converted to other land uses every year (FAO 2005), and around 1,035 million hectares were affected by human-induced soil degradation (GEF & IFAD 2002). In the Andean Region of South America, conditions are not different from the global tendency. Erosion is a major issue that has been affecting upland farmers across the region, reducing on-site agricultural production, and affecting major water supply systems and off-site hydropower generation downstream.

There is an underlying chain of causes and consequences that explains the reduction of environmental services. The drivers of change of the services provided by ecosystems could be either direct drivers (e.g., land use land cover (LULC) change and resource consumption), or indirect drivers (e.g., globalization, markets, consumption choices or beliefs) (MEA 2005; WRI 2005). Hydrological services provided by watersheds (e.g., regulation of water flow and control of soil erosion) are directly related to land use change and land management practices, which are linked with socio-economic drives (e.g., poverty, pressure on land or economic incentives) (Boardman et al. 2003).

Given the accelerated reduction in environmental services, investments in watershed management and ecosystems conservation and the development of new regulations and market incentives are of increasing importance in both temperate and tropical regions. One of the mechanisms that have been strongly promoted, particularly in the developing world, is the adoption of Payment for Environmental Services (PES). Given the recognized value of water by agricultural, human and industrial sectors, hydrological services may provide the most opportunities for PES approaches. In the Andes, the environmental services with the highest probability of being economically compensated are those related to watershed functions such as sediment retention, water availability in the dry season, and carbon sequestration. Although some PES projects have been established in several watersheds of the world, there is an increasing demand for understanding the relationships between land use and the impact on watershed hydrology, upstream and downstream relationships, as well as methods to quantitatively evaluate these linkages that could guide watershed management, policy-making and investments (Aylward 2002; Bruijnzeel 2004).

In Latin America, most PES programs have been designed by estimating the services using general data or conventional knowledge about the direction and magnitude of the linkages between land use and hydrological variables (Landell-Mills & Porras 2002; Pagiola 2002; Rojas & Aylward 2003). Biophysical models have not been successful in meeting the real needs of decision-makers, either because they do not take into account the linkages between land use change and environmental services or the upstream and downstream interactions (FAO 2004; Kosoy et al. 2005; Rosa et al. 2004b; Sierra & Russman 2006).

In the few cases where research has explored options to maximize individual services, such as crop or timber production, there is limited research into trade-offs with other environmental services such as water resources, carbon sequestration or biodiversity. For example, afforestation for carbon sequestration can highly impact the stream flow and water quality (Jackson et al. 2005). Understanding environmental impacts of alternative management approaches for the range of environmental services is essential (Carpenter et al. 2006; Nelson et al. 2008). Quantification of these causal relationships will enable a reliable assessment of the services provided by a watershed or a region (Nelson et al. 2009). Therefore, there is a critical need for estimating environmental services tradeoffs emerging from land use changes in order to enhance the bundle of ecosystem services available from a watershed, as well as methods to quantitatively evaluate these linkages that could guide its management, policy-making and investments.

However, despite decades of research and the wide range of hydrological models available, there is still a lack of information for policy questions about different environmental services from watersheds (also termed as hydrological services or watershed functions), especially at large (i.e., coarser) scales (Tomich et al. 2004). Studies of other hydrological services, such as stream flow stabilization, water quality and quantity effects (particularly in the case of tropical settings) have seldom been done (Kramer et al. 1998), especially long term studies in agricultural watersheds (Santelmann et al. 2004). Although most analyses of hydrological services have focused on soil erosion control, a number of these studies are focused on on-site effects such as reduction of productivity, whereas off-site effects such as siltation of reservoirs have been less explored (Lal 1998). Even more, a vast majority of these studies are from the U.S., Canada, Australia, and Europe, and only a few the tropics and subtropics.

Besides hydrological services, carbon sequestration is another regulating environmental service that ecosystems can provide (MEA 2005). The atmospheric concentration of carbon dioxide (CO_2) has increased by 31% since 1750. About three-quarters of the anthropogenic emissions of CO_2 to the atmosphere during the past 20 years is due to fossil fuel burning. The rest is predominantly due to land-use change, especially deforestation (IPCC 2001). According to the Millennium Ecosystem Assessment (2005) the effect of changes in terrestrial ecosystems on the carbon cycle reversed during the last 50 years. They were on average a net source of CO_2 during the nineteenth and early twentieth centuries, primarily due to deforestation, but with contributions from degradation of agricultural, pasture, and forestlands. Terrestrial ecosystems became a net sink around the middle of the last (i.e., 20th) century, although carbon losses from land use change continue at high levels. Factors contributing to the growth of the role of ecosystems in carbon sequestration include afforestation, reforestation, forest management, and changes in agriculture practices. However, according to the same assessment, the future contribution of terrestrial ecosystems to the regulation of climate is uncertain given the limited understanding of soil respiration processes.

The main objective of the United Nations Framework Convention on Climate Change (UNFCCC) is "the stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system". Carbon sequestration is a way to reduce greenhouse gas emissions. The Clean Development Mechanism (CDM), established under the UNFCCC provides financial support to developing countries in return for greenhouse gas reductions through payments for carbon sequestration in forests. In the first stages of the CDM, sinks were limited to afforestation and reforestation projects for the first commitment period. However, sinks resulting from cropland management, grazing land management, re-vegetation, and forest land management are now also recognized (Dumanski 2004).There are three general means by which agricultural and forestry practices can reduce greenhouse gases: (1) avoiding emissions by maintaining existing carbon storage in trees and soils; (2) increasing carbon storage by, e.g., tree planting, conversion from conventional to conservation tillage practices on agricultural lands; (3) substituting bio-based fuels and products for fossil fuels, such as coal and oil, and energy-intensive products that generate greater quantities of CO_2 when used.

Tropical deforestation is responsible for about 20% of the world's annual CO_2 emissions (IPCC 2001). Biomass and carbon content are generally high in tropical forests, reflecting their influence on the global carbon cycle. Tropical forests also have great potential for the mitigation of CO_2 through appropriate conservation and management due to their high rates of net primary production. According to a review by Silver et al. (2000) on reforestation of abandoned tropical agricultural and pasture lands, the aboveground biomass increases at a rate of 6.2 Ton C/ha/yr during the first twenty years of succession, and at a rate of 2.9 Ton C/ha/yr over the first 80 years of regrowth. During the first twenty years of regrowth, forest in wet zones have the fastest rate of carbon accumulation aboveground, followed by forest in the moist and dry zones. Tropical reforestation has the potential to serve as carbon offset mechanism both above and belowground for at least 40 to 80 years. The review also indicates that forest growing on abandoned agricultural land accumulates biomass fasters than other past land uses, while soil carbon accumulates faster on sites that were cleared but not developed, and on pasture sites. Another study proved that invasion of grasslands by shrublands increased carbon in vegetation to a much lower extent than usually expected, where soil carbon increased only on the drier sites, and decreased in the wetter sites (Jackson et al. 2002). Regional estimates of the carbon sequestration potential of these practices are crucial if policy makers are to plan future land uses to reduce national CO2 emissions and to participate in carbon trading markets.

4.3 **Study Area**

The Andean mountains contribute to the quality of life and ecosystems in the nearby areas by providing environmental services. But despite a wealth of available natural resources, the welfare of the rural population has declined significantly over the last decade (CAM 2005; López 1999; Rosa et al. 2004a; Suarez 1999). The region contains high levels of unemployment due, in part, to decreases in agricultural product prices and rural sector investments. Difficult economic conditions have forced many rural communities to over-exploit the natural resources. Such land use management strategies not only jeopardize the productivity of their own private lands (De Janvry & Glickman 1991) but also cause detriment to surrounding areas. In Colombia, for example, sediment deposition and increased flooding cause damages of approximately USD 1 billion per year (Estrada et al. 2003).

In the Andes, many environmental tradeoffs exist. For example, private agricultural production of upper catchment farmers conflict with public water conservation for hydro electrical power and urban uses. Although studies using experimental economic methods demonstrate that many users are willing to accept reduced short term income in order to maintain the long term ecosystems benefits (Cardenas 2003), there is a recognized need for policy development and implementation to foster more sustainable practices (e.g. minimum tillage).

The Andes Mountains Region is one of the main hot spots of erosion induced soil degradation in the world. Severe soil erosion in the Andes Mountains of South America constrains rural development and exacerbates poverty by decreasing the productive capacity of highland agriculture and livestock raising (Amézquita et al. 1998; Ruppenthal et al. 1996; Veneklaas & Vanek 1990). This process is considered the major form of soil degradation in the Colombian Andes, and has been related to overgrazing and inadequate agricultural practices such as frequent burning, tillage and lack of cover crops (Lal 1998; Muller-Samann 1999; Oldeman et al. 1991). Romero (2005) argued that there is a great variation in soil loss in the Andean region and that the conditions under which measurements were taken are generally not properly reported. Suarez de Castro and Rodríguez (1962) reported a variation between 1 to 800 Ton/ha/yr for the same soil in the Colombian Andes and Ruppenthal et al. (1996) found a maximum of 222 Ton/ha/yr. Stroosnijder (1997) conducted a study in five watersheds in Ecuador on sediment measurements in rivers, and found that the average was 7.3 Ton/ha/yr depending more on land use than on soil type. Studies in Colombia, Venezuela and Indonesia (Table 4.1) on runoff plots measured soil losses ranging from 0.2 to 8.9 Ton/ha/yr in established coffee plantations (Ataroff & Monasterio 1997; Iijima et al. 2003; Suarez de Castro & Rodríguez 1962).

Table 4.1 Soil losses from studies on runoff plots under similar conditions. Adapted from Hoyos (2005).

			Plot characteristics			Average
Land Use	Plantation	Location	Slope (%)	Measured Time (Years)	Area (m^2)	Soil Loss (Ton/ha/year)
Shade	Established	Colombia	53	8	90	0.1-1.1 [a]
coffee	Established	Colombia	10-60	8	6000	10.4 [b]
	Established	Venezuela	60	2	12	0.6 [c]
	Recent	Colombia	45	8	120	0.6-4.8 [a]
Sun Coffee	Established	Venezuela	60	2	12	1.2 [c]
	Recent	Indonesia	60	1	12	3.2 [c]
	Recent	Venezuela	27	4	108	2.0-8.9 [d]
		Colombia	-	2	2500	0.5 [b]
Pasture		Colombia	21	8	10-40	3.4-61.4 [a]
Pasture		Colombia	21	8	30	514.0-873.3 [a]
Rotation						
Bare soil						

(a) Suarez de Castro and Rodriguez (1962), (b) Suarez de Castro (1953), (c) Ataroff and Monasterio (1997); (d) Ijima et al. (2003), (c) Under various treatments: tillage, no-tillage, alley cropping and no alley cropping.

The 258 km^2 Las Ceibas watershed serves as a case study (Figure 4.1). The ecosystem of the upper catchments is cloud forest, which has been primarily disturbed by agricultural development. There are advanced erosion processes, such as sheet erosion, rill and gully erosion, that are occurring in the watershed. The causes of these processes are a combination of natural and anthropogenic factors (Personal communication). The watershed has gone through a change of land uses from natural forests and shrublands to agriculture and extensive cattle rising. Conversions of landcover from forest to pasture have increased soil compaction and soil temperature, and has decreased relative humidity, water infiltration and organic matter. According to the environmental authority, during the last twenty years, forest cover has been reduced by approximately 1,000 ha. The erosion processes in the watershed are causing major sedimentation problems in the water supply system that brings water to Neiva City (Figure 4.1). During the rainy season, the sediments have reached 127,000 Ton/yr. The annual discharge has been reduced from 5.5 m^3/s in the 1980s to 4.1 m^3/s in the 1990s. Despite the vulnerability of this watershed with respect to its hydrological and economic value, its hydrology and ecosystem functions remain poorly understood.

Las Ceibas is divided in 22 political districts or divisions called veredas. It has a rural population of 2,283 habitants, where 31.2% are children between 0-10 years old, 13.7% are children between 10 -14 years old, 46.9% are adults between 15-59 years old and 8.2% are over 59 years old. Regarding gender, 56% of the population are men and 44% are women. There are 732 residential properties and 117 properties as open space, which are used for agricultural purposes as part of the productive system of each family. From those properties, 64% have electricity and 32% do not (the non-response rate was 4%). Regarding the water supply, 28.7% take water from natural water springs located on their farm, 31.3% of the people get water from a different property, 13.% get their water from the local water supply system, 6.8% get water from associated water supply systems, and an additional 9.8% from regional water supply systems (the non-response rate was 10.5%).

The rural communities are small, homogenous and follow family traditions. They are affected by the globally common phenomenon termed *urbanization*, where the media, access to urban centers, and poor returns from farming (combined with political instability in rural areas) tend to make the youth desire a future that is not tied to agriculture.

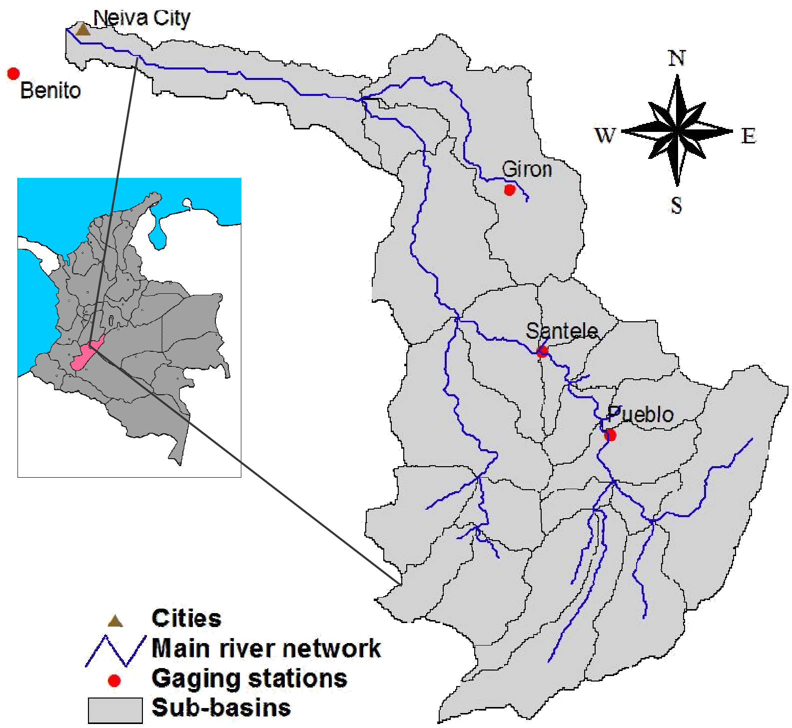

Figure 4.1 Location of Las Ceibas watershed, sub-basins and Neiva City.

4.4 Modeling of current hydrological services

The Soil and Water Assessment Tool (SWAT) was used to quantify potential changes and improvements in hydrological service, specifically soil erosion control, in Las Ceibas watershed. This model has been widely used to predict, with good confidence, the use of Best Management Practices (Arabi et al. 2007). The watershed was divided in twenty two subbasins, which were divided in seventy seven Hydrological Response Units (HRU). Each HRU is the total area within a subbasin with a particular set of land use, soil type and management (Neitsch et al. 2004).

The five main land cover and vegetation types of the watershed are (CAM 2005): grasslands and shrublands (27.6%), managed grasslands (1.9%), natural forests (7.3%), natural grasslands (31.0%), natural shrublands (12.8%), perennial crops (8.2%), seasonal croplands (0.5%), secondary forests (9.5%) and uncultivated land (1.2%). In the natural forests, the most representative species are: Alder (Alnus sp), Oak (Quercus sp), Encenillo (Weinmannia sp), Arrayan (Myrcia sp), Siete cueros (Tibouchina sp), Chusque (Chusque sp), Gaque (Clusia sp.), Laurel (Persea sp.), and ferns (Cyathea sp). In the secondary forest the most representative species are: Amarillo (Ocolea), Cedar (Cedrela sp), Cobre (Dalbergia sp), Guayacán (Bulnesia sp), Diomate (Astronium sp), Caracolí (Anacardium excelsum), Igúa (Pseudosamanea guachapele), Balso (Ochroma Lagopus), Algarrobo (Prosopis sp), and Palms (Syagrus sp). The soil types range from sandy to silt – clay, and mainly superficial especially in the steep terrains. Soils have a neutral reaction and low organic content in grasslands, and high and very high organic content in those areas with good vegetation cover such as forests (Figure 4.2). The soils located in the cold humid areas are well drained and acid. The soil profiles were classified in the subgroup level Typec Haplustepts and Lithic Haplustepts (IDEAM 2000).

Table 4.2 presents information related to the weather stations used for modeling. These data have been provided by the Instituto de Hidrología, Meteorología y Estudios Ambientales de Colombia (IDEAM). The land use map was provided by CAM, which was obtained from a combined study of remote sensing data analysis and field survey.

Table 4.2. Weather Input Data for SWAT.

Data provided	Name	Years	Elevation (m.a.s.l)	Location
Daily Precipitation	Pueblo Nuevo	1985-2005	1580	0249 N-7505W
Daily Discharge	Pueblo Nuevo	1983-1999	1280	0249N-7505W
Daily Levels	Pueblo Nuevo	1983-1999	1280	0249N-7505W
Daily Precipitation	Santa Helena	1983-2005	1160	0251N-7506W
Daily Precipitation	Hacienda La Gironda	1983-2005	1060	0245N-7507W
Daily Precipitation	Aeropuerto Benito Salaz	1978-2006	439	0258N-7518W
Daily Temperature	Aeropuerto Benito Salaz	1978-2006	439	0258N-7518W
Relative Humidity	Aeropuerto Benito Salaz	1978-2006	439	0258N-7518W
Sunshine duration	Aeropuerto Benito Salaz	1978-2006	439	0258N-7518W

Wind speed	Aeropuerto Benito Salaz	1978-2006	439	0258N-7518W
Daily Precipitation	Palacio Vega Largo	1978-2006	1100	0256N-7504W
Daily discharge	Guayabo	1980-2000	650	0255N-7509W
Daily Levels	Guayabo	1977-2003	650	0255N-7509W

Following a run of the model, the mean annual erosion intensity in the watershed was determined to be 2.32 Ton/ha and the maximum is 16.48 Ton/ha. Figure 4.3 shows the spatial distribution of sediment yields per hectare in the different hydrological units. Based on the calculation of potential annual soil erosion rates, it was found that areas with very low soil loss dominate with 40.1% of the total area (Table 4.3), while very high and high rates are found in 1.7% and 3.3 % of the area.

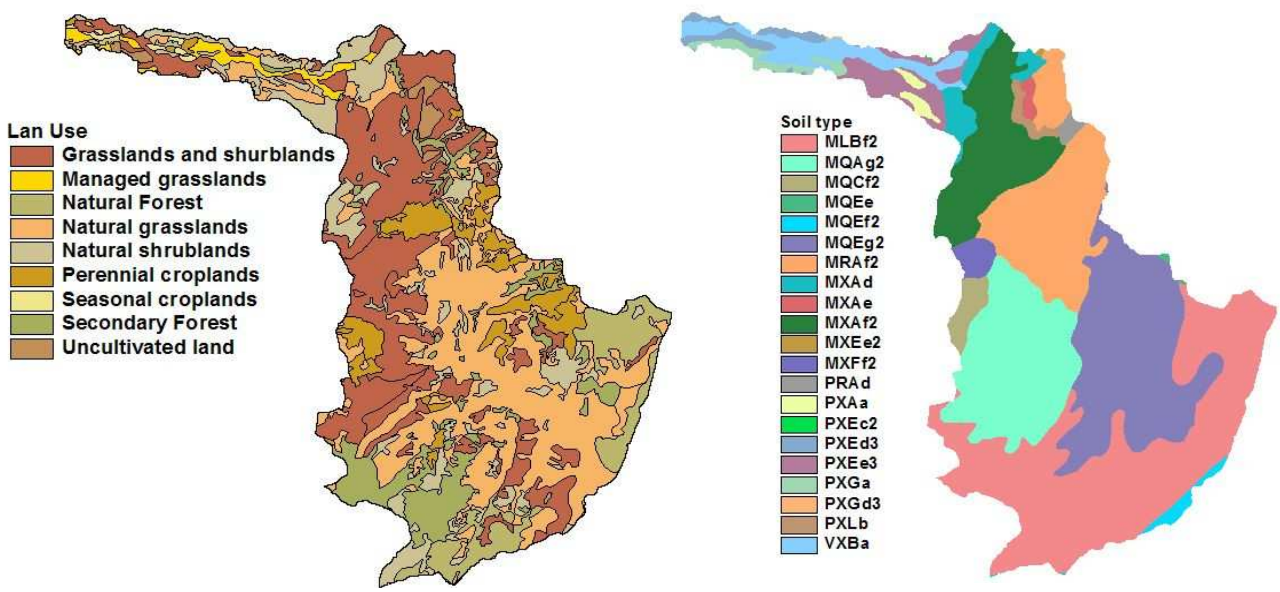

Figure 4.2 Land use and soil type distribution in Las Ceibas watershed.
M: Mountainous, P: hillside, V: Valley, L: cold humid, Q: Temperate humid, R: Temperate dry, X: Hot and very dry, pedology description (B , A, C, E, G, L), a: slope 0 – 3%, b: slope 3% - 7%, d: slope 12 – 25%, e: slope 25 – 50%, f: slope 50-75%, g: slope higher than 75%, 2: moderate erosion, 3: severe erosion.

Table 4.3 Distribution of soil loss in Las Ceibas watershed.

Soil loss category	Category Range (Ton/ha/year)	Percentage of watershed area
Very low	< 1	40.1
Low	1 -3	28.0
Medium	3 -5	26.9
High	5 – 9	3.3
Very high	> 9	1.7

By analyzing the results of the SWAT run, it can be inferred that the highest contribution to the annual sediment yields is from mixed grasslands with 56.4%, followed by the shrublands with 22.0% and forest with 17.8% (Table 4.4). The hydrological units that have the highest rate of sediment yields are located unevenly across the watershed, either in the lower, middle or upper subbasins. Although the highest contribution to annual sediment yields is from mixed grasslands, there are small hydrological units with high rates of erosion (Figure 4.2). As for the sediment distribution among soil types, the highest amount is generated from the soil type (MLBf2) that is located in a mountainous landscape with a cold humid climate and high slope, and also has the greatest areal extent in the watershed (Table 4.5).

When analyzing the sediments production per hectare by land use, the highest rate was 3.07 Ton/ha for shrublands, followed by forest with 1.88 Ton/ha, and mixed grasslands with 1.71 Ton/ha (Table 4.4). With respect to the soil erosion rate by soil, the highest production per hectare was found in the soil type coded PXGd3 with 10.33 ton/ha, which has an area of just five hectares and thus gives a total annual sediment yield of 54 tons (Table 4.5). The soil type with the second highest rate is the soil coded PXEc2 with a rate of 9.7 tons/ha, in which two hectares produce 23 tons annually.

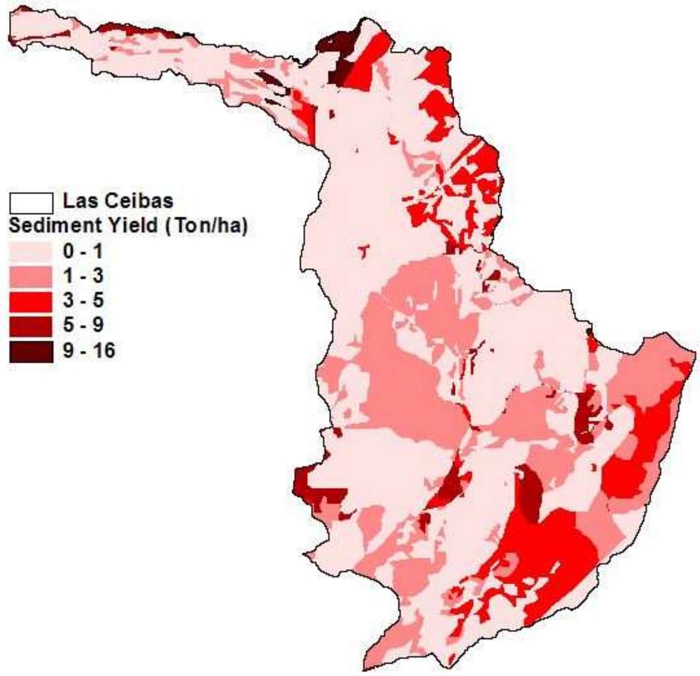

Figure 4.3 Risk erosion map of Las Ceibas watershed.

Table 4.4 Annual Sediment yields by land use in Las Ceibas Watershed.

Land Use	Annual sediment yield (Ton)	Contribution to annual sediment yields (%)	Annual sediment yields per ha (Ton/ha)
Mixed grasslands	26,639	55.7	1.71
Mixed shrublands	10,412	21.8	3.07
Perennial crops (Coffee)	5,810	12.1	2.99
Forest	3,488	7.3	0.78
Seasonal croplands	1,420	3.0	0.85
Uncultivated land	99	0.2	0.30

Table 4.5 Annual Sediment yields by soil type in Las Ceibas Watershed.

Soil Type	Annual sediment yield (Ton)	Contribution to annual sediment yields (%)	Annual sediment yields per hectare (Ton/ha)
MLBf2	19,370	41.00	2.69
MRAf2	7,772	16.45	2.30
MQAg2	6,517	13.79	1.81
MQEg2	4,047	8.57	0.86
PXEe3	3,077	6.51	3.85
MXAf2	2,427	5.14	0.98
PXEd3	1,302	2.76	3.77
MXAd	1,178	2.49	1.98
VXBa	558	1.18	0.57
VXBa	558	0.84	0.57
PXAa	395	0.83	2.69
MQCf2	390	0.19	1.10
PXGa	91	0.12	0.33
PXGd3	54	0.09	10.33
MQEf2	41	0.05	0.22
PXEc2	23	0.02	9.70
MQEe	9	0.00	0.43
MXAe	0	0.00	0.00
MXEe2	0	0.00	0.00
MXFf2	0	0.00	0.00
PRAd	0	0.00	0.00

4.4.1 Model calibration and evaluation

The model was calibrated and evaluated for flow and sediment loads using the 30m and 50m DEM. The observation data was divided in two periods, with data from 1984 to 1993 used for calibration and data from 1994 to 1999 used for evaluation. In this model, the NSE varied from 0.61 to 0.82 for the daily flow, and from 0.69 to 0.89 for monthly flow. For the sediments, Nash and Sutcliffe efficiency varied from 0.58 to 0.76 for daily loads, and from 0.63 to 0.80 for monthly loads. The efficiency value of 0.7 is usually considered as sufficiently good for hydrological evaluation, and 0.6 as satisfactory (Krysanova et al. 2007). A detailed description of the uncertainties associated with the model and sensitivity analysis are presented in section 2.3.1 and section 3.3.

4.5 Modeling of current carbon sequestration

The carbon fixed by the main land uses in Las Ceibas was estimated using the CO2Fix model (Masera et al. 2003; Schelhaas et al. 2004). The CO2Fix is an ecosystem model that quantifies the carbon stocks and fluxes in biomass and the soil organic matter with time step of one year. Carbon stocks in living biomass are calculated as the balance between growth and decay (turnover, mortality and harvest). The soil organic carbon is calculated from the decomposition of litter from turnover and mortality processes. The main land uses selected for simulation were: forests, mixed grasslands (without management), shrublands and coffee. Section 2.3.4, carbon sequestration modeling, presents a description of carbon sequestration models and the uncertainty associated with the selected model. The CO2Fix model for this case study did not require calibration, as earlier studies have shown it performs well if the model was parameterized with data that reflects local conditions. The model was run with data on table and forests inventories obtained from IDEAM (IDEAM 2000) .

The CO2FIX model was run for the four land uses independently for fifty years. The basic wood density data were obtained from Brown (1997) and initial humus and carbon contents were derived from local forest inventory data (IDEAM 2000). The forest was simulated with two cohorts: oaks and ferns. In the forest, the natural mortality was set to 1%, no logging was assumed and wood products were excluded from the carbon calculations. Foliage turnover was set at one per year (Kira & Shidei 1967). They suggest higher foliage turnover for tropical moist forests between 1.3 and 1.5 year, but CO2FIX V 3.1 does not allow foliage turnover values over 1 year. Branches turnover was set at 0.10 per year (Kira & Shidei 1967) and root turnover was set at 0.10 per year (Gill & Jackson 2000).

The grass is simulated as a tree with a small stem volume, a large amount of foliage and roots and no branches (Bailis 2009; Groen et al. 2006; Lasco et al. 2005; Schelhaas et al. 2004). The stem part is needed, since allocation to foliage and roots is driven by stem increment (Schelhaas et al. 2004). In the grasslands, 70% is harvested every year by grazing and turnover rates are 0.8 for foliage and 0.9 for roots. Coffee was simulated as an agroforestry system with plantain. Shrublands and coffee were simulated similarly with a basic wood density of 0.45 and 0.40 respectively, with turnover rates of 0.6 for foliage and 0.3 for roots.

Table 4.6 Main parameters used for simulating land use carbon sequestration in Las Ceibas watershed.

Land Use	Initial humus content of the soil (Ton C/ha)	Basic wood density (Ton DM/m^3)[a]	Carbon content (Ton C/Mg DM)	Initial Carbon Stock (Ton C/ha)
Forest				
Oak	125	0.50	0.50	115
Fern	125	0.48	0.45	76
Grasslands	9	1.00[b]	0.47	22
Shrublands	10	0.45	0.48	47
Coffee	12	0.40	0.50	65

a: (Brown 1997); b: (Schelhaas et al. 2004)

In the four land uses, the carbon in the biomass was more sensitive to changes in harvesting and mortality than carbon in the soil. The net carbon sequestration was estimated as the difference between long-term average carbon stocks and initial carbon stocks in each of the land uses. The initial carbon stocks in the forests and in the agroforestry system was high, while it was assumed to be low and moderate in the grasslands and shrublands respectively (Table 4.6). The net carbon sequestration in the forest was relatively low at 0.70 ton C/ha, due to the low rate of harvesting assumed. The simulations indicate that the long term total carbon storage ranges from 60 to 220 Ton C/ha. Carbon stored in living biomass ranged from 36 to 135 Ton C/ha and carbon in soils ranged from 24 to 85 Ton C/ha (Figure 4.4). The net carbon sequestration reached 35 Ton C/ha for forest, -13 Ton C/ha for the grasslands, 28 Ton C/ha shrublands and 46 Ton C/ ha for the perennial crops.

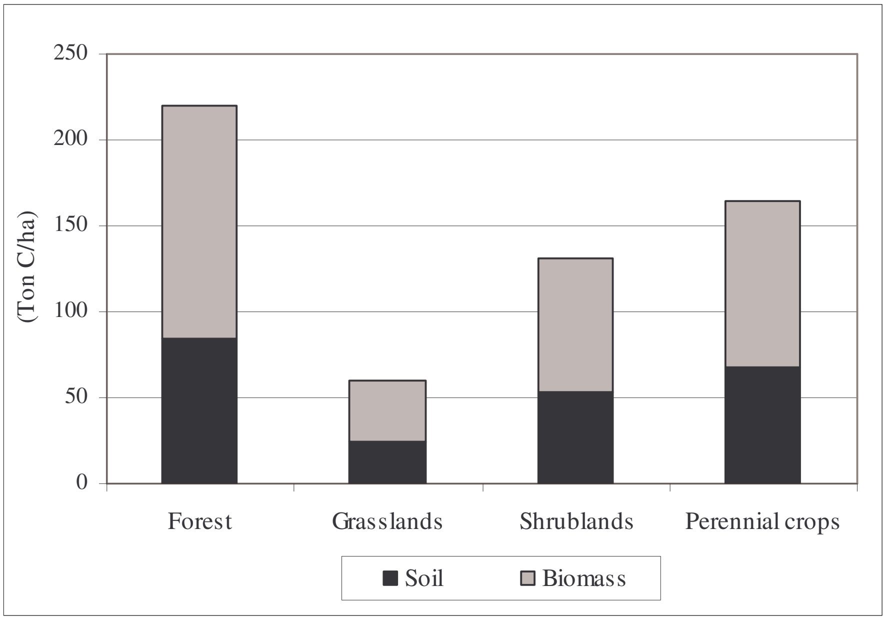

Figure 4.4 Comparative long-term carbon stocks in biomass and soil in five land uses.

4.6 Potential changes

The potential changes in Las Ceibas towards increasing the hydrological services are analyzed based on the risk erosion map obtained from SWAT simulations. The scenarios proposed target those HRUs capable of producing significant changes in reducing soil erosion (Table 4.7). The impact of implementing land-use and management alternatives can be determined through dynamic simulation of different scenarios from 1990 to 2040. The scenarios include changes in land cover and management practices such as use of rotational grazing, minimum tillage and green manures. The land use changes are mainly related to reforestation with native species. The impacts of these changes are analyzed regarding the environmental services of discharge regulation, erosion control and carbon sequestration.

Table 4.7. The ten highest rates of erosion per HRUs in Las Ceibas.

Land Use	Soil Type	Sediment Yields (Ton/ha)
Shrublands	PXEe3	16.48
Mixed grasslands	MLBf2	13.72
Mixed grasslands	PXEc2	10.85
Mixed grasslands	MQEg2	10.83
Shrublands	PXGd3	10.33
Shrublands	MXAd	10.17
Mixed grasslands	PXAa	9.88
Uncultivated land	PXEd3	9.48
Mixed grasslands	MLBf2	9.48
Shrublands	MRAf2	9.42

Scenario 1: rotational grazing

Most grasslands were cleared from forests since the mid eighties with a deforestation rate of 230 ha/yr, which were then gradually impoverished through burning at the end of every dry season to generate regrowth. Given that farming systems have to be developed to suit the ecological and economic characteristics of the region, the changes proposed take into account the economic importance of livestock production together with the need to improve the environmental conditions of the watershed. In the current livestock production, the animals are allowed to roam the entire pasture and have free access to stream corridors, which can lead to erosion, overgrazing and water pollution. In the proposed scenario, the extensive livestock system is changed to a rotational pasture livestock system, where cattle are moved into different areas to control how land is grazed. The rotation was applied to 18% of the grasslands that have an erosion rate higher than 3 Ton/ha and are located mainly in the middle and upper subbasins. A total area of forty six square kilometers was reforested in this scenario.

Scenario 2: green manures and cover crops

Green manures and cover crops are known to help soil and water conservation besides increasing the organic matter and improving the soil structure. Grasses and legumes are also used in a variety of erosion control measures such as vegetative hedges, strip cropping and cover cropping. Given the economic importance of livestock production, the aim of this scenario is to improve the current conditions of the grazing activity. Evidence from Colombia suggests that improved grasses may also help in carbon fixation by removing as much as 2 billion Tons of carbon dioxide annually from the atmosphere (Fisher et al. 1994). These improved grasses refer to deep-rooted grasses that have been introduced in the South American Savannas. The perennial grasses (Andropogon gayatus and Brachiaria humidicola) can convert as much as 53 Tons of CO_2 per hectare annually to organic matter. When sown with legumes, the grasses are able to fix even more CO_2. Green manures and cover crops were simulated in all of the area covered by perennial crops.

Scenario 3: reforestation of grasslands

Several studies have shown the positive effect of reforestation on erosion control and sometimes discharge regulation (Günter et al. 2009; Keesstra et al. 2009; Quintero et al. 2009; Zheng et al. 2008). Reforestation with native species in low productive lands with erosion rates higher than 3 Ton/ha was modeled as an option to reduce sediment yields. The area used in the simulation corresponds to forests that were converted to grasslands and shrublands and located 1,500 meters above sea level. Ten percent of the basin that is twenty five square kilometers were reforested in this scenario. The current land covered by forest was not modified and kept with a low harvesting rate.

4.7 Results

Discharge regulation: The mean annual flow obtained in the current and three scenarios was lower than the flow measured, as shown in Figure 4.5. The highest reduction of 4.9% was obtained with reforestation of native species, followed by green manures/cover crops with 3.4%, and rotational grazing with 2.6% (Table 4.8). Although the reforested area was small compared with the areas that were changed in the other two scenarios, the impact of reforestation on discharge was the highest amongst the three scenarios.

According to Heuvelmans et al. (2005) simulated hydrographs have a tendency to underestimate peak flows or may send a false signal during the recession periods when SWAT is applied to basins that had a short travel time of much less than a day. In the SWAT model, the surface runoff volume is predicted from daily rainfall using the soil conservation service (SCS) curve number (CN) equation (Section 2.3.1). Peak runoff rate in the SWAT model is estimated by using the modified rational formula. Flow is routed through the channel using a variable storage coefficient method or the Muskingum routing method. The watershed concentration time is estimated using Manning's formula, considering both overland and channel flow. This can explain the underestimation of the peak flows obtained for Las Ceibas watershed. The use of these empirical equations is the main criticism to SWAT model, Alternative routines to the channel routing module could be a channel reach continuity equation, in which the process are physically described (Shen et al. 2009).

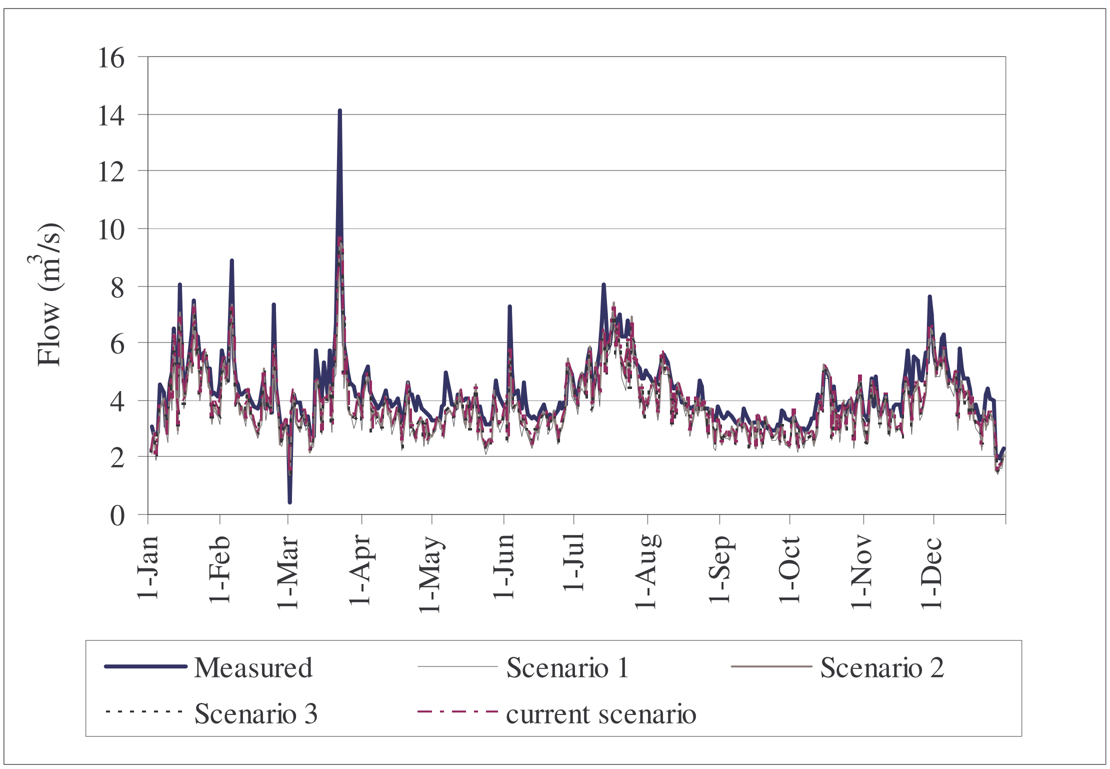

Figure 4.5 Mean daily flow at Guayabo Station and the estimated flow in the three scenarios.

Table 4.8 Mean measured and simulated flow and sediments loads in the three scenarios.

Scenario	Mean Flow (m^3/s)	Change in Flow (%)	Sediment Load (Ton x 10^3) Complete series	Change in sediment Load (%)	Sediment Load (Ton x 10^3) Without peaks	Change in sediment Load (%)
Measured	4.68	-	130	-	31	-
Scenario 1	4.52	3.4	44	66.1	26	14.5
Scenario 2	4.56	2.6	45	65.9	27	14.3
Scenario 3	4.45	4.9	41	68.8	24	21.6

Erosion control: Although sediments loads obtained with SWAT follow the measured trend, peak loads were under estimated in years 1985, 1986, 1989, 1997 and 1999 (Figure 4.6). Given that erosion in Las Ceibas watershed is caused by water, these peak events are associated with extreme events that caused large movements of soil, especially in the upper and middle catchments, which were not well simulated by SWAT. The mean annual sediment load measured at Guayabo station is 130 x 10^3 Ton and the load obtained in the scenarios was at least 60% lower (Table 4.8) due to the underestimation of the peak events. If the annual mean is calculated without peak events, the mean annual load measured is 31 Ton x10^3 and the modeled amount with current conditions was 27 x 10^3 Ton. If the peak loads are not included in the mean, the largest reduction in sediment load was obtained with reforestation, were the load was 21.6% lower than the load obtained during current conditions. The sediment loads obtained in the other two scenarios, rotational grazing and green manures, were 14% and 9% lower than the load obtained during current conditions.

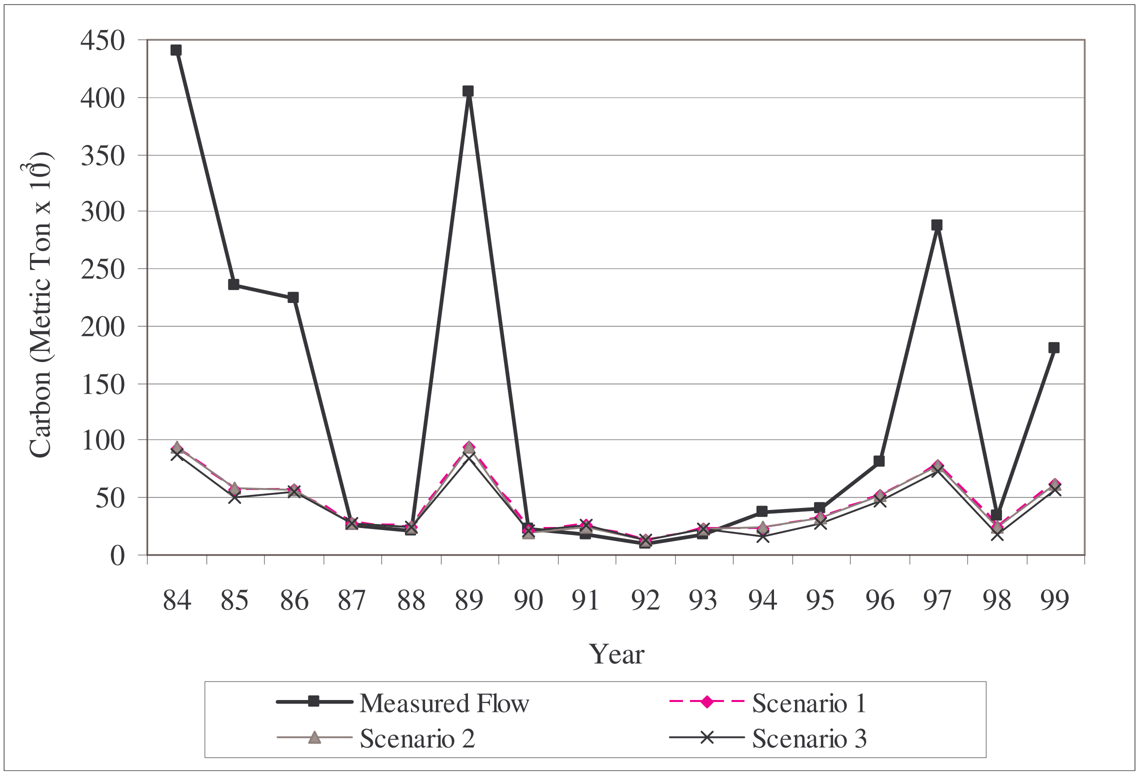

Figure 4.6 Annual sediment load measured at Guayabo Station and the estimated load in the three scenarios.

Carbon sequestration: The area currently covered by forest was not changed in any of the three scenarios and was kept constant in 35 Ton C/ha in the fifty years period. Therefore, all changes were solely due to introduction of rotational grazing, cover crops and green manure, or reforesting grasslands. The largest change was obtained in the reforestation of the grassland scenario, where the net carbon sequestration was 81% higher than at current conditions. Net carbon sequestration increased 63% with rotational grazing and 34% with green manures and cover crops (Figure 4.7).

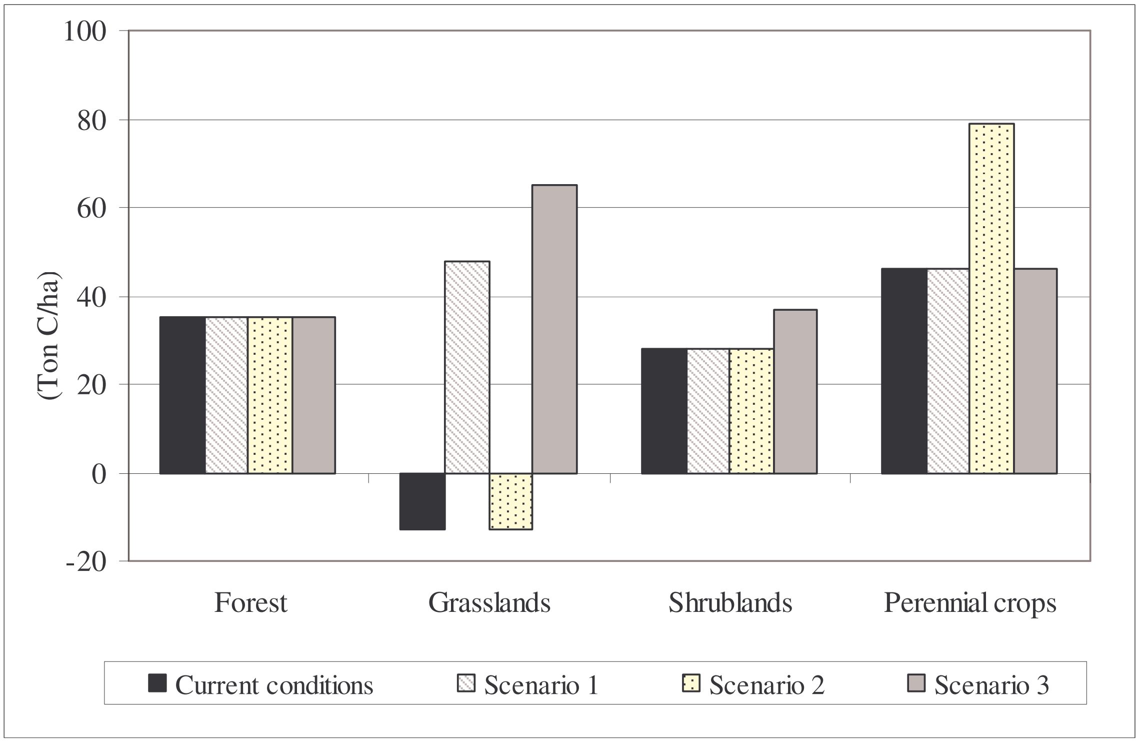

Figure 4.7 Net carbon sequestration from soil and biomass for the three scenarios.

Figure 4.8 compares the provisioning of environmental services in the three scenarios. Reforestation, which mainly targeted the increase of carbon sequestration, leads to decreases in peak river flows. Figure 4.9 shows the change in the provisioning of the ecosystem services in fifty years. At the basin level, the most positive change in the three ecosystems was obtained with reforestation followed by rotational grazing and green manures. This result is directly related with the fact that grasslands cover a large area of the Basin, and reforestation took place in 18% of the basin. At the HRU level, it was possible to identify small areas with the same land use and same soil type that have a great impact on the final output. This was the case for the perennial crops and some grasslands. The discharge regulation improved more with the cover crops and green manures than with the rotational grazing due to the improvement of water infiltration. Although the changes obtained in the scenario of green manures and cover crops were the lowest for carbon sequestration and erosion control, the impact of the change was high taking into account that the area modified was just 8% of the total area.

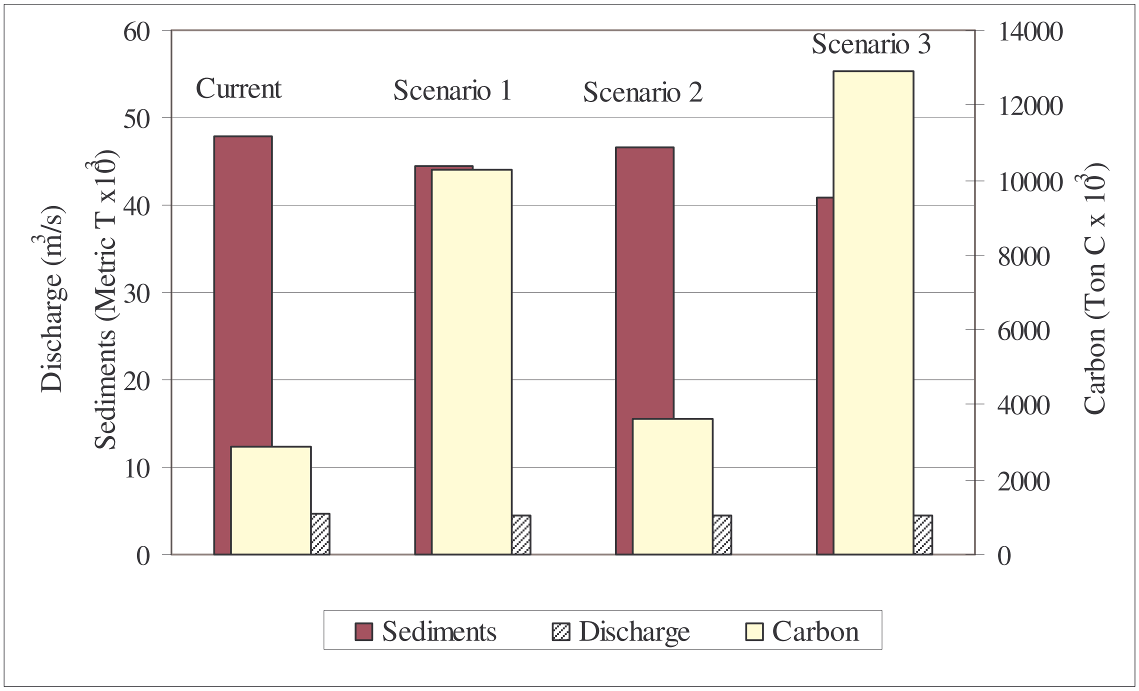

Figure 4.8 Current and potential provisioning of environmental services in Las Ceibas.

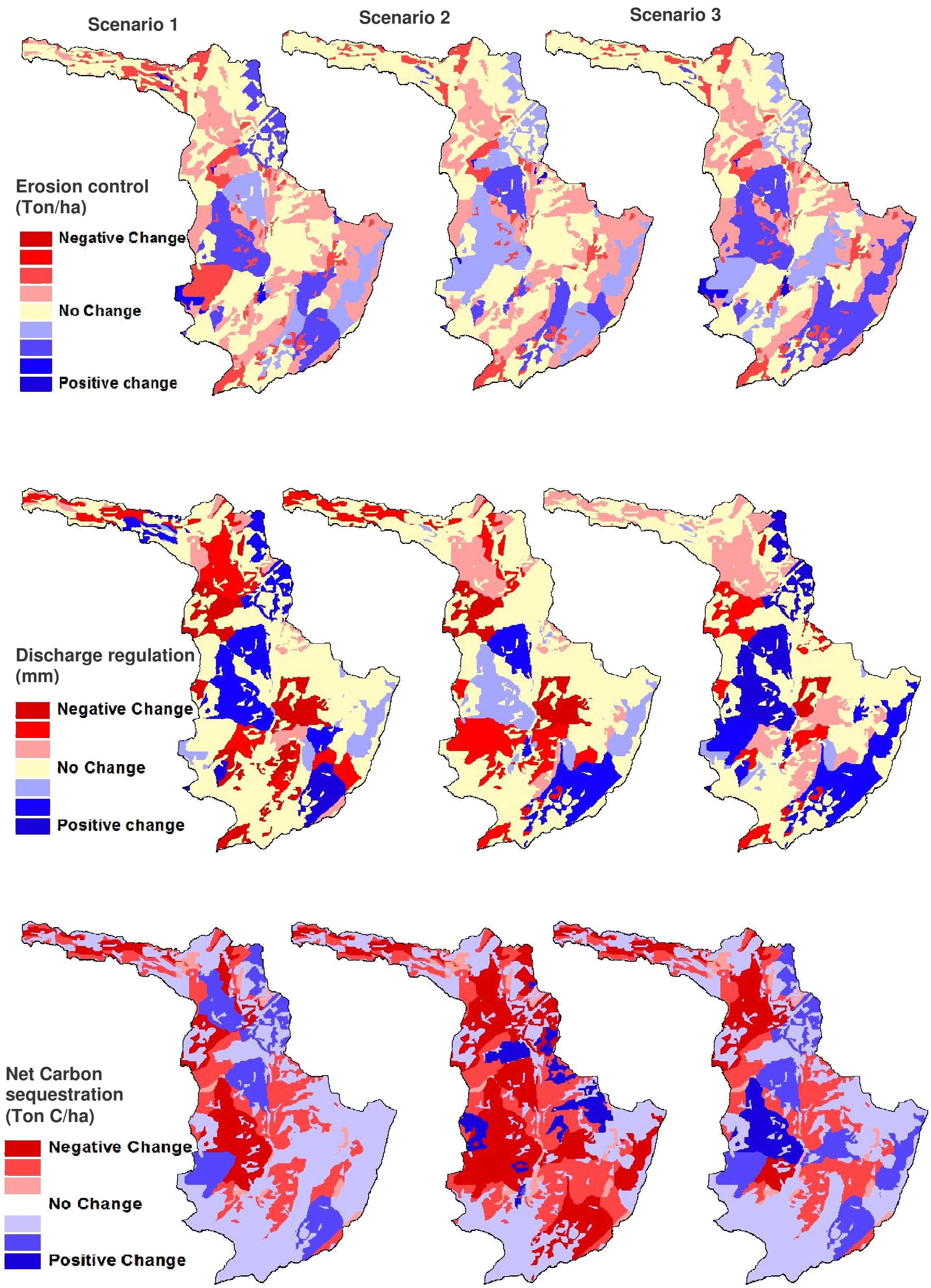

Figure 4.9 Maps of change of erosion control, discharge regulation and carbon sequestration for the three scenarios.

4.8 Conclusions

Environmental modeling can be a useful tool to analyze different scenarios for a bundle of ecosystem services. It was possible to find synergies between services for different scenarios at the basin level and the hydrological response unit level. No evidence of tradeoffs among services was found given the conservation nature of the scenarios analyzed. For example, reforestation largely increased carbon sequestration but also increased the provisioning of hydrological services. Moreover, scenarios that targeted the improvement of hydrological services also improved carbon sequestration to some degree.

The main land use targeted for change in this basin was grasslands. The hydrological model indicated that the HRUs that were contributing the most to sediments were the ones in the middle, covered by grasslands and therefore selected for reforestation and rotational grazing. Although grasslands offer extensive area for carbon sequestration and storage, more information is needed on annual carbon release and uptake rates in tropical grasslands (White et al. 2000). Basic input information for carbon sequestration models is also needed for other tropical ecosystems especially regarding soil carbon storage.

Making evident the linkages between environmental services can increase the feasibility of the implementation of a particular scenario. It is often hard to argue that national funds should pay for conservation incentives in watersheds. For example, the model suggests that reforestation can result in lower maximum stream flows but the differences will be modest. It appears unlikely that financial incentives to promote reforestation, based solely on improved soil and water management would be economically sound. To increase the potential profitability of reforestation, carbon sequestration should be bundled with hydrological services.

This paper presented the results of an environmental modeling exercise that investigated the biophysical provision of ecosystem services but did not include the economic evaluation. Before payments for these ecosystem services are instituted, tradeoffs between their biophysical provision and their value to people and the environment should be identified.

Chapter 5

5 Bundling of Environmental Services in an Andean watershed: valuation and tradeoffs

5.1 Abstract

Services provided by natural ecosystems sustain and fulfill human life. Given the importance and increasing degradation of ecosystem services, economic incentives such as Payment for Environmental Services (PES) have emerged as a way of transferring financial resources from those beneficiaries willing to pay for these services to those stakeholders willing to provide them. Decisions concerning both ecosystem management and economic incentive design are complex and imply environmental and social trade-offs. Ecosystem valuation has been used as a tool to guide decision making, and has evolved to integrate economic considerations and ecological understanding. Examples of bundled services valuation that apply valuation methods, such as neoclassical economic techniques, contingent valuation and environmental benefit indexes, demonstrate the need of an integrative economic and ecological assessment framework. This paper presents a conceptual framework for valuating ecosystem functions and the application of an environmental index that combines environmental assessments and economic valuation. Las Ceibas watershed in Colombia served as case study to determine the environmental index taking into account three ecosystem services: erosion control, discharge regulation and carbon sequestration. The analysis evaluated three land use change scenarios: rotational grazing, cover crops and green manures, and reforestation.

Keywords: environmental services, economic valuation, environmental index, bundling.

5.2 Introduction

The Millennium Ecosystem Assessment (MEA 2005) broadly defined ecosystem services as the benefits that people obtain from ecosystems. This, however, introduced the challenging question: how can we value both ecological and economic ecosystem benefits adequately. Proponents of ecosystem service valuation believe that valuations can: (1) improve understanding of problems and trade-offs; (2) be used directly to make decisions; (3) illustrate the distribution of benefits and thus facilitate cost-sharing for management initiatives, and (4) drive the creation of innovative institutional and market instruments that promote sustainable ecosystem management (Armsworth & Roughgarden 2001; Aylward & Barbier 1992; Daily 1997; Dasgupta et al. 2000; Pagiola et al. 2005b). On the other hand, some ecologists believe that economic valuation fails to arrive at an accurate value for ecosystem services, since it does not provide for an adequate methodology that measures the entire range of supporting, regulating and cultural services.

5.3 The theory behind environmental service valuation

Since the late 1960s, there has been a growing interest in the analysis and valuation of the benefits provided by ecosystems (Turner et al. 2003). This interest was initiated by an increasing awareness that benefits provided by natural ecosystems were often underestimated in decision making (Hein et al. 2006). Today, the debate on what is the value of natural ecosystems has received much attention in the scientific literature. Over the years economic science has not only been concerned with direct use value of ecosystems, focusing on quantifying and analyzing goods and services that produce tangible benefits (e.g. timber, food, energy) but also, in recent years, researchers have tried to broaden their scope in recognizing the indirect use, non-use, existence, bequest and option values of ecosystems. In pursuing this effort, economists have tried to develop techniques to extend monetary valuations to ecosystem services (Chee 2004).

Farber et al. (2002) provide a detailed description on the various economic concepts of ‘value’. According to them, the basic notion of value that guides economics is an anthropocentric notion, meaning a contribution to a human goal, objective, desired condition, etc. The model used by economists argues that value is based on satisfaction, pleasure or a utility goal. In this study, Farber et al. (2002) describe two core concepts in valuation techniques: the willingness to pay (WTP) for a service and the willingness to accept (WTA) compensation for a service loss. Once these two quantities are measured, costs/benefit scenarios allow the comparison of environmental impacts for environmental management. Then, an evaluation of the net social costs and benefits of each scenario for the different environmental issues can be performed (Pearce & Howarth 2000).

5.3.1 Ecological and economic valuation

The debate amongst ecologists and economists regarding valuation of ecosystems tries to answer the question of what is the value of nature (Heal 2000; Sagoff 1998; van der Straaten 2000; Wilson & Howarth 2002). Ecologists argue that economic valuation of ecosystem might not capture the full range of biological and sustainable issues that surround ecological resource management. Heal (2000) for example, argued that ‘‘the emphasis on valuing ecosystems and their services is probably misplaced” and emphasized that “valuation is neither necessary nor sufficient for conservation. We conserve much that we do not value, and do not conserve much that we value’’. Sagoff (1988) claimed that environmental systems are connected to core social values that cannot be reduced to monetary terms. Wilson & Howarth (2002) argued that ecological resource management involves questions of equity that are poorly addressed through the standard methods of environmental valuation. Moreover, the intrinsic values of natural system processes within the natural system itself may hold different functional value properties than their corresponding economic values. Turner et al. (2003) argued that the ability to value nature’s services is constrained by the complexity of nature itself, reliable and accurate estimates of all services are not possible given the complexity of the ‘‘production function’’ of nature, which is little understood most of the time.

There are two issues regarding the complexity of nature and its implication to valuation that are cited in the literature reviewed: (1) critical thresholds inherent to ecological systems, which are related to shifts in ecosystem state where a small change in a driver causes a marked change in ecosystem condition (Groffman et al. 2006), e.g. changes in water flow regimen due to changes in tree densities, and (2) the joint production of goods and services that is inherent to most of nature's processes, e.g. forests perform valuable hydrologic, nutrient cycling, and climate regulation functions, which is related to bundled ecosystem services. Slight alterations in ecosystem conditions can substantially change the economic value because human lives and communities might be at high risk. Therefore, traditional economic valuation may not be able to adequately capture the full impact of changes, e.g., in the proximity of a critical threshold (Limburg et al. 2002). Although some valuation methods, such as avoided costs or replacement costs, try to value services by determining the costs to replace them or restore them after they have been damaged (see next section), they cannot cover the full range of ecological costs that are caused (Heal 2000). From the conservation perspective, the standards to avoid such critical thresholds, like deforestation rates, should be ecologically-based and not based on economics (Farber et al. 2002).

The joint production of services provided by ecosystems is related to trade-offs that emerged when enhancing one or a group of services. Valuation techniques rarely take into account the linkages and effects that a particular strategy caused in other services (Carpenter et al. 2006). Researchers have developed models to try to understand these linkages and processes, but sometimes they can be data intensive and typically cannot capture the range of process. In some cases, where there is not enough data or resources for modeling, hypotheses related to these linkages are formulated by expert knowledge. Most of the time this step is not included in the economic valuation processes, not even when just one single service will be targeted.

5.3.2 Ecosystem Valuation Framework

Several studies have provided frameworks for the valuation of ecosystem services (Costanza et al. 1997; de Groot et al. 2002; MEA 2005; Santelmann et al. 2004; Turner et al. 1998). The framework presented in this review (Figure 5.1) was adapted from Hein et al. (2006) and De Groot et al. (2002). In this framework, the first step is to translate ecological processes into a limited number of ecosystem functions, which will provide ecosystem goods and services. In this sense, there is a distinction between functions and services, where services depend on functions and are valued by people. The gray boxes indicate the steps attached to each section of the framework. The ecological values are associated with the concept of sustainability, and supporting and regulating services. These are determined by ecological criteria (e.g., integrity, resilience, and resistance), and are related to non-anthropocentric values (Table 5.1). Social values are non-use values and are related to cultural services (e.g. historical landscape, recreation and tourism, educational information). Economic values are related to anthropocentric values, and can be estimated through the methods explained in the next section (Table 5.1). From an economic perspective, the first task is to identify how an environmental change affects well-being, and the second task is to estimate the value of these changes through a variety of direct and indirect valuation techniques (Pearce & Howarth 2000).

Table 5.1. Compilation of meanings of the word ‘value’ (Chee 2004; Pearce & Howarth 2000; Turner et al. 2003).

Applications of the word ‘value’

Anthropocentric Value:

Use value

- Direct use value: where individuals make actual use of a resource for commercial purpose, also called market value (e.g. timber from a forest).
- Indirect use value: the value of entities that may have little or no market value, but have use value, also called intrinsic value (e.g. watershed protection or carbon sequestration by forests)

Non-Use value

- Existence value: the value attached to the knowledge that species, natural environments and other ecosystem services exist, unrelated to current or future use.
- Bequest/vicarious value: a willingness to pay to preserve the environment for the benefit of other people in the future.
- Option value: a willingness to pay a certain sum today for the future use of an asset.
- Quasi-option value: the value of preserving options for future use assuming an expectation of increasing knowledge about the functioning of the natural environment.

Non-anthropocentric Value:

- Functional value: value based on the contribution one (ecological) entity makes to the existence of another. It also encompasses the good of collective entities.
- Intrinsic value: the value that an object possesses independently of its valuation by others.

Aggregation of values is perhaps the most challenging element in the framework. The traditional procedure of economic valuation is to establish individual-based values, and then aggregate them to have a total value. This is appropriate when the services provided are individually enjoyed, as is the case for private goods and services that are not shared, and where there are no positive or negative impacts or externalities on others. Indicators resulting from economic valuation, ecosystem assessment and social perceptions are usually not in the same units and thus difficult to compare. In this stage the participatory methods or group deliberations, described in the next section, are increasingly used as decision aid.

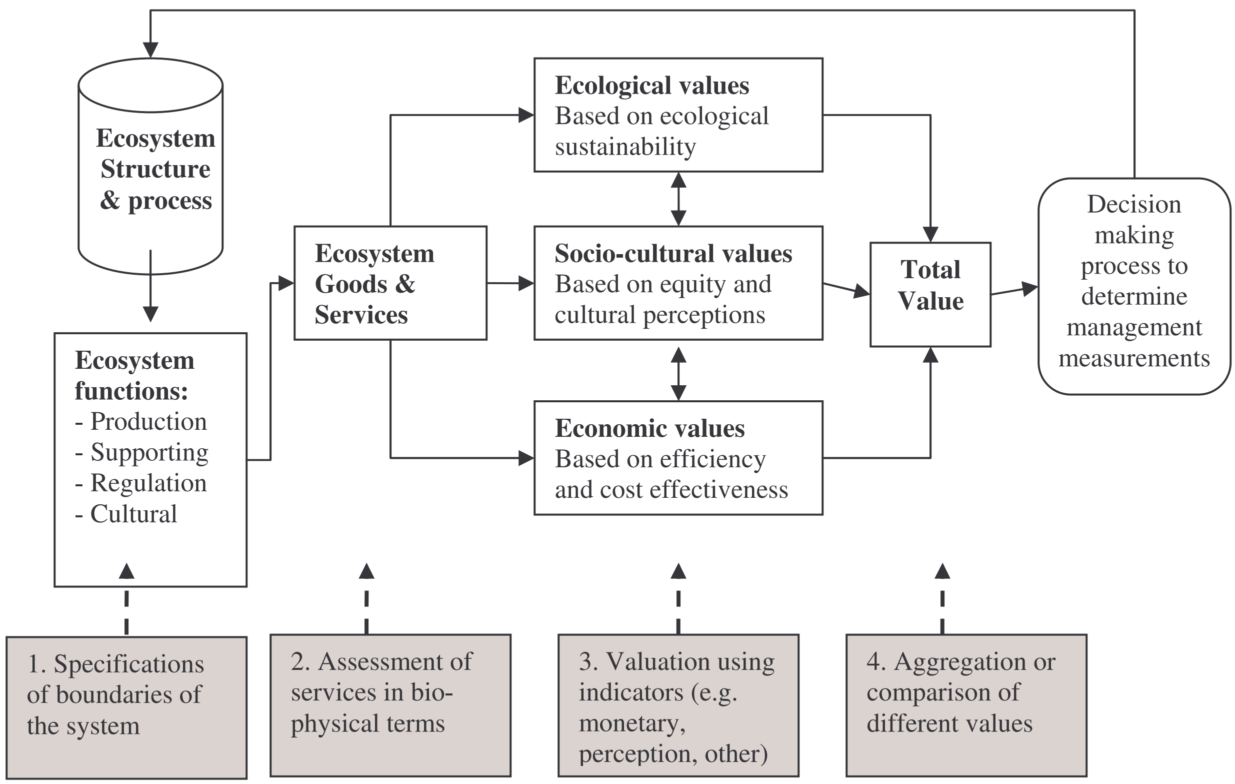

Figure 5.1. Framework for valuation of ecosystem services (de Groot et al. 2002; Hein et al. 2006).

5.4 Valuation Methods

There are two main groups of methods within ecosystem services valuation: neoclassical methods and participatory methods. Within the first group there are six major economic valuation techniques when existing markets do not adequately capture social values in the neoclassical literature (Arrow et al. 1993; Chee 2004; Farber et al. 2002; Garrod & Willis 1999; Pearce 1998; Pearce & Howarth 2000): avoided cost, factor income, replacement cost, travel cost, hedonic pricing and contingent valuation. Five methods form the second group (Aldred & Jacobs 2000; Cardenas & Carpenter 2008; Chee 2004; Jack 2009; Lienhoop & MacMillan 2007; Peterson et al. 2003; Proctor & Drechsler 2003): conjoint analysis, Multi-Criteria Analysis, citizens' juries, deliberate contingent valuation and scenario planning.

5.4.1 Ecosystem Services Valuation in PES

The basic principle behind PES is that land users living in areas that are providing environmental services should be compensated for the costs of their provision, and that those who benefit from these services should pay for them, thereby internalizing these benefits (Pagiola et al. 2005a; Wunder 2006). Payment schemes of bundled services are found where different services are sold from a single land area, reflecting the fact that ecosystems co-produce more than one service and that an investment in the production of one service results in simultaneous production of other services. Landell-Mills and Porras (2002) surveyed 28 cases of PES schemes selling bundled services, and distinguished two types of schemes: (1) merged bundles, where it is not possible to separate the services, and (2) shopping basket bundles, where specific services can be bought and land users sell different services to different buyers. The main advantage of merged bundles is that they are easier to manage and that they reduce transaction costs in the PES scheme. However, they could be less effective since merging services makes it impossible to target payments to individual beneficiaries. On the other hand, the shopping basket approach is better designed to maximize returns, however this approach is more complex to manage and results in higher cost.

In PES schemes the payment must be at least equal to the additional benefit that the land user receives from the alternative land use (otherwise land users would not change their behavior). Furthermore the payment must be less than the value of the benefit to downstream populations (otherwise beneficiaries would not be willing to pay for services). In this sense, valuation techniques can be divided to estimate (i) the value of the supply and the required compensation, and (ii) the value of the demand.

Valuation of the supply

According to FAO (2004), most valuation studies for the implementation of PES schemes are based on the estimation of the opportunity cost. The opportunity cost refers to the net income that providers can earn whenever a productive activity is avoided or transformed under the PES scheme. This value will indicate the approximate amount of the compensation required to offer an effective incentive in order to change or maintain the intended land use. The opportunity costs can be estimated using the valuation techniques described in the previous chapter. For example, the returns in sustainable forest management should compete with those of alternative land uses such as agriculture. The opportunity cost of conservation mainly depends on alternative land uses. In developing countries, for communities living in or around a forest, the main opportunity cost of forest conservation is the net predetermined opportunity to carry out farming on the land. Opportunity costs are estimated by considering the highest profitable alternative. Other costs incurred can also be added in the equation such as labor costs, costs related to conservation activities and transaction costs (Pagiola et al. 2005a).

Valuation of the demand

Most valuation studies for PES schemes estimate the willingness to pay for the service. The payment capacity of the beneficiaries is analyzed though their willingness to pay, usually estimated through contingent valuation, in which people are asked the value of a particular service based on a hypothetical market or scenarios. Another common technique is to use indirect methods such as Factor Income, e.g. by estimating the economic value of the water resource as an input to local economic processes.

Aggregation and comparison of costs and benefits

- *Cost-benefit analysis (CBA):* is based on the net present value of an alternative, and is the total value of benefits that an alternative is expected to yield minus the total expected decrease in stakeholder value or opportunity costs (Turner et al. 2003).

- *Cost-effectiveness analysis (CEA):* This method allows the selection among alternative strategies to achieve a given environmental objective by comparing the costs of each strategy (measured in monetary units) with its environmental impact (measured in physical units). This analysis allows ranking of policies based on cost-effectiveness ratios, with the assumption that all of these policies are worthy to be undertaken (Lu et al. 2003).

5.4.2 Tradeoff analysis between indicators or services

Aggregation methods have been used to analyze the trade-offs between economic and environmental indicators. An example of this type of assessment was performed to analyze the relationships between land use and hydrology in the Arenal basin of Costa Rica, where the marginal values of changes in flows of water and sedimentation for a downstream hydroelectric plant were estimated. In this case, it was found that, while sediments from pasture compared with forested areas did have a cost expressed in the loss of hydroelectric production, ranging from US$35 to US$75/ha respectively, this loss was exceeded by the benefits of increased water yield from pasture areas, which ranged from US$250 to US$1,100 (Aylward & Echeverria 2001). Depending on the type of the forest area cleared, the highest yield of water appeared to be associated with fragmented cloud forest areas which have the highest rates of interception of precipitation. One of the reasons behind this result is that the Arenal reservoir is an inter-annual regulation reservoir, in which hydroelectric production depended on total flows, and it is therefore largely independent of dry season flows. The results indicated that ranching produced higher net present values than what was offered by the government for reforestation. When the costs and benefits were examined making a distinction among various kinds of landholders, they found that the higher return per hectare depended in part on the location in the catchment, primarily to large landholders, and that the incentives that were being offered for conservation still appeared attractive to small landholders. In this study, the authors analyzed other potential benefits such as carbon sequestration under pasture, where estimations were based on potential biomass.

Another case that combines analysis of environmental processes and economic valuation, is the research conducted in the Fuquene Lake, Colombia, where linear programming was applied to measure the tradeoffs between the economic performance of different activities and environmental externalities, such as sediments and nutrients loads that arrived into the lagoon (Rubiano et al. 2007). An optimization model calculated the costs of land use changes and technology under different spatial and temporal scenarios. The researchers found that the opportunity cost of reducing one ton of sediments per hectare was US$1,578 for farmers located in the upper catchment, and US$1,255 for farmers located in the middle-catchment. The authors argued that this cost could be avoided if conservation farming practices were offered and adopted by Fuquene farmers because net income could be improved and negative externalities reduced.

An ecological assessment that evaluated the effect of alternative agricultural practices on wildlife habitat was conducted in Iowa, USA through the application of an habitat suitability score system (Santelmann et al. 2006). The scenarios targeted three different policy choices: (1) profitable agricultural production remained the dominant objective of landscape management, (2) land cover patterns evolved as landowners attempted to meet water quality standards and (3) cover patterns changed to increase habitat for indigenous wildlife and support biodiversity. The score system was developed by reviewing species' use of habitat, which was used to prepare a habitat map for each species in the past, present and for future scenarios. The results showed that the biodiversity scenario ranked highest in providing habitat, followed by the water quality and production scenarios. Although the results were intuitively expected, this type of assessment and landscape analysis can be a powerful tool for policy deliberation.

Few cases are cited in the peer-reviewed literature regarding payment of bundled services and most of the cases are found in institutional reviews. The bundled services approach was used by FONOFIFO in Costa Rica to sell to different beneficiaries services such as carbon sequestration, water quality and quantity, biodiversity and landscape beauty. It was also used by The Nature Conservancy in Belize, Bolivia, Costa Rica and the United States to bring additional revenues for biodiversity protection by promoting the sale of carbon credits and biodiversity services (Landell-Mills & Porras 2002).

The US Department of Agriculture's Conservation Reserve Program (CRP) developed and used environmental indicators to broaden the program's environmental benefits beyond erosion reduction, which was the primary focus of the program. The CRP has used an environmental benefits index (EBI) since the 1990s to rank applications for land enrollment. Even though the CRP cannot be classified as a PES scheme per se since the direct beneficiaries are not the ones paying for the services, it is an interesting case to mention because its focus on multiple services. The index calculates a score for applications for enrollment submitted by potential participants. In each enrollment period it is possible to change the relative weights assigned to each objective. Some of the factors that are usually included in the index are wildlife habitat, water quality from reduced water erosion, runoff, and leaching, reduced wind or water erosion, long-term benefits of certain practices such as hardwood trees, air quality, conservation priority and a cost factor (Ribaudo et al. 2001; Sullivan et al. 2004). Reviews of the CRP suggested that bidding has reduced costs, but that the full potential of bidding may not have been realized (Claassen et al. 2008). Even though the EBI has been adapted to be more flexible and to include more science oriented decisions regarding environmental impacts, criticism against CRP pointed out that the criteria to set up weights can still be manipulated to meet political and bureaucratic objectives. Another criticism is that the CRP should restrict enrollment based on high priority, environmentally sensitive areas to have more cost-effective results (Yang et al. 2005).

The Global Environmental Facility (GEF) in the project "Silvopastoral Ecosystem Management" adopted a similar approach to the EBI of the CRP (Pagiola et al. 2005a). In this case, the different land uses were associated to a ranking system, which was used afterwards to set up the payments. Separate indexes were developed for biodiversity and carbon sequestration for each land use. For example, in the biodiversity index the annual crops did not receive points and primary forests were the ceiling with 1.0 points. For the carbon sequestration index, degraded pastures did not receive points while primary and secondary forests and silvopastoral systems received the highest score of 1.0. Then, both indexes were added with the same weights to have a total environmental index. One of the limitations of this methodology is that biodiversity not only depends on the land use, as it was calculated, but on the extension, location and its relation to other land uses. As in the previous case, the incentives could turn perverse if no limitations are established beforehand to prohibit inappropriate adoptions.

Significant differences exist between PES schemes that are government-financed and those programs were funding comes from the users of the ecosystem service being provided (Engel et al. 2008; Wunder et al. 2008). In this review, the user-financed programs were more directly adapted to local conditions and needs, had been better monitored, and suggested less complexity due to fewer objectives than government-financed programs. On the other hand, the government-financed programs benefit from economies of scale due to their large size and tend to evolve to become more like user-financed programs.

Most of the cases reviewed focused first on enhancing one single service and it is usually the reason for establishing a PES. Afterwards, policy makers realize that there are synergies with other ecosystem services, such as the case of carbon sequestration in Australia and the Nature Conservancy initiative. Targeting a particular ecosystem service can emerge from a local problem or externality such as sediments going into a dam or water quality in a water supply system. In these cases, the relationship between supplier and beneficiary is clear and therefore a market can be identified. Afterwards, other services are attached to this emerging market, either as merged bundled or a shopping basket. In the FONOFIFO and CRP cases, although several services are considered within the index, there is no formal evaluation of trade-offs between the multiple services offered. The examples of bundled services pointed out that further research is needed to quantifying the real environmental impacts of changes in land use and management practices.

5.5 Integration of physical assessment and economic valuation

In the new methodology proposed in this paper, an environmental index is obtained for each land use type for three different scenarios. This index combines three environmental services: erosion control, discharge regulation and carbon sequestration. It also includes the net present value (NPV) or the discounted operational profits of each alternative obtained from the cost-benefit analysis. Simulation results from the Soil and Water Assessment Tool (SWAT) (Arnold et al. 1999) and the carbon model CO2FIX

(Schelhaas et al. 2004) are compared with the present state to obtain a change ratio. The three components have the same weight and are added to obtain the total index.

The spatial scale of the environmental index (EI) is the Hydrological Response Unit (HRU) *i*, for which a land use change scenario *j* is assigned and is equal to j = 1, 2…N, depending on the number of selected scenarios. The erosion control is denoted as *Sij*, the discharge regulation is denoted as *Dij*, the amount of carbon retention is denoted as *Cij* and the net present value is denoted as *Vij*. Equation 5.1 determines the index and is equal to:

$$EI_{ij} = \frac{(S_{i1} - S_{ij})}{|S_{i1}|} + \frac{(D_{i1} - D_{ij})}{|D_{i1}|} + \frac{(C_{ij} - C_{i1})}{|C_{i1}|} + \frac{(V_{ij} - V_{i1})}{|V_{i1}|}$$

$$EI_{ij} = -\frac{\Delta S}{|S_{i1}|} - \frac{\Delta D}{|D_{i1}|} + \frac{\Delta C}{|C_{i1}|} + \frac{\Delta V}{|V_{i1}|}, \text{ Equation 5.1}$$

$$\text{where } S_{i1}, D_{i1}, C_{i1}, V_{i1} \neq 0$$

The optimal environmental index for each hydrological unit would be the highest possible. The index can be adapted to enphasize any of the four components by adding a weighting factor. This methodology was applied to three land use change scenarios in the Las Ceibas, (Figure 5.2).

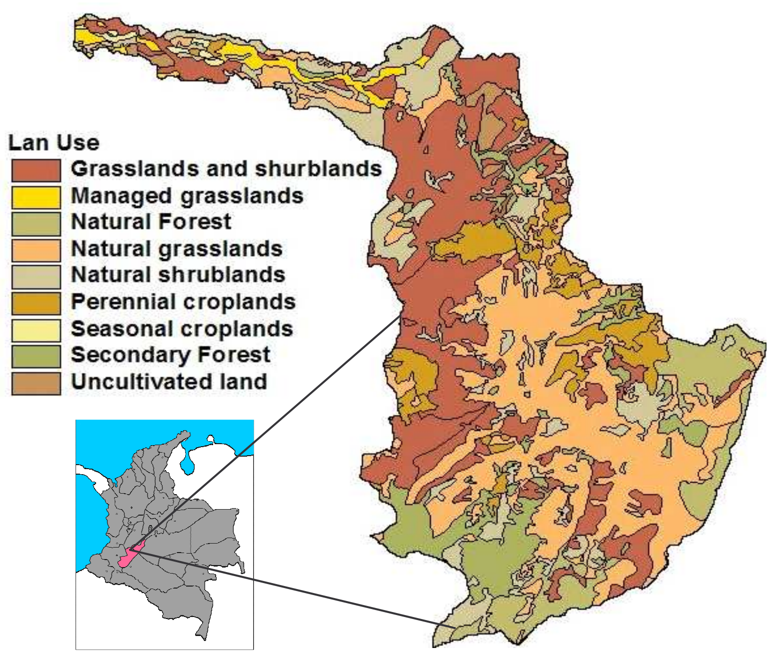

Figure 5.2 Land use map of Las Ceibas watershed.

As stated in Chapter 4, three scenarios were analyzed to improve the services, namely: rotational grazing (scenario 1), green manures and cover crops (scenario 2), and reforestation of grasslands (scenario 3). Figure 5.3 shows the change in the provisioning of the ecosystem services in fifty years. At the basin level the most positive change in the three scenarios was obtained with reforestation followed by rotational grazing and green manures. This result is directly related with the fact that grasslands cover a large area of the basin, and reforestation took place in 18% of the basin. At the HRUs level, it was possible to identify small areas with the same land use and same soil type that have a great impact on the final output. This was the case for perennial crops and some grassland. The discharge regulation improved more with the cover crops and green manures than with the rotational grazing due to the improvement of water infiltration. Although the changes obtained in the scenario of green manures and cover crops were the lowest for carbon sequestration and erosion control, the impact of the change was high taking into account that the area modified was just 8% of the total area.

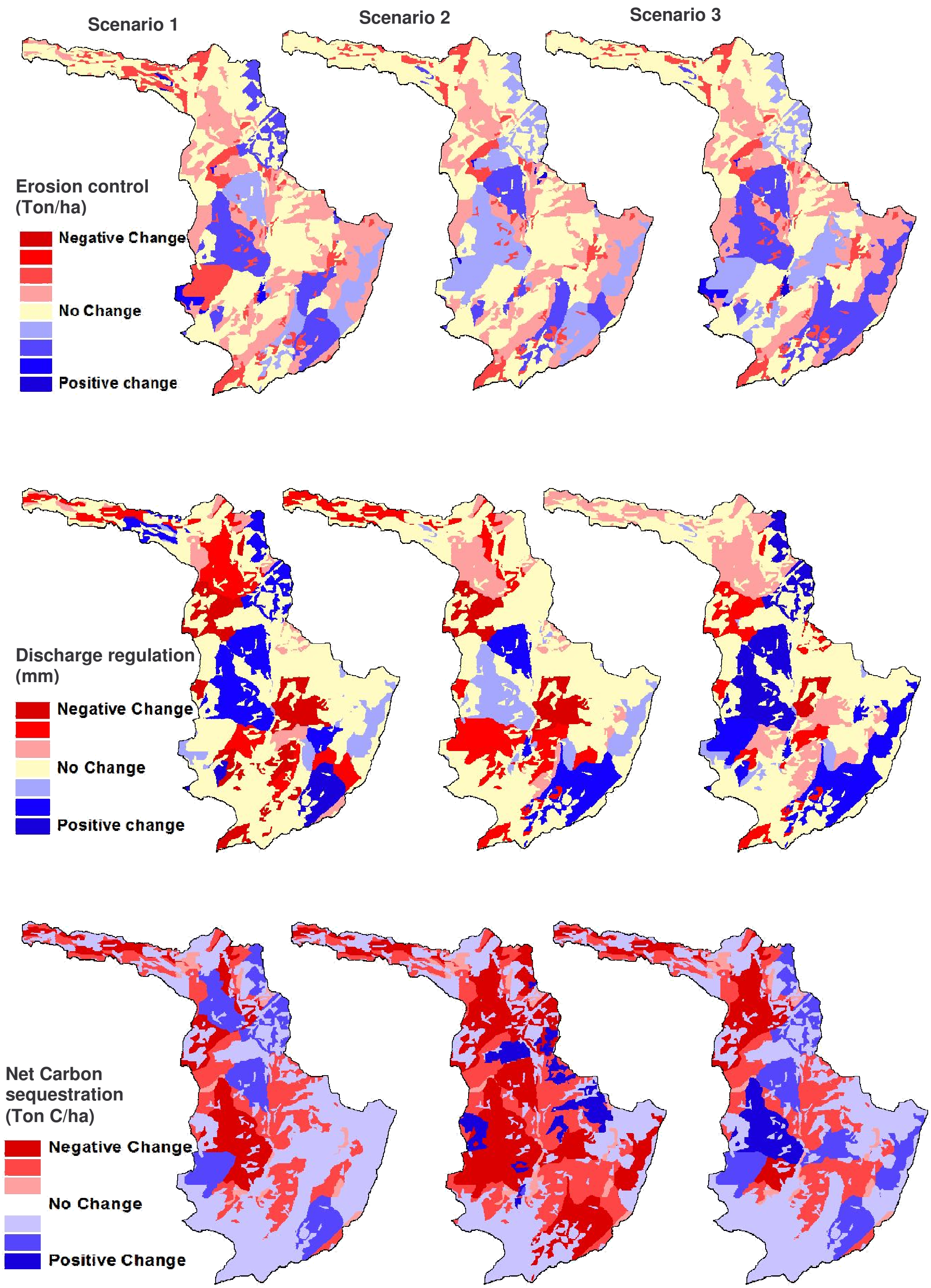

Figure 5.3. Maps of change of erosion control, discharge regulation and carbon sequestration for the three scenarios.

5.5.1 Cost-benefit analysis

The Net Present Value (NPV) of current and possible future land uses was determined using cost-benefit analyses for a time period of 50 years. NPV is a wide used tool in discounted cash flow (DCF) analysis, and is a standard method for using the time value of money to evaluate long-term projects. Used for capital budgeting, and widely throughout economics, finance, and accounting, it measures the excess or shortfall of cash flows, in present value terms, once financing charges are met. The NPV of a sequence of cash flows takes as input the cash flows and a discount rate or discount curve and outputting a price; the converse process in DCF analysis, taking as input a sequence of cash flows and a price and inferring as output a discount rate (the discount rate which would yield the given price as NPV) is called the yield, and is more widely used in bond trading.

In order to calculate to NPV, each cash inflow/outflow is discounted back to its present value (PV) and then they are summed. Therefore NPV is the sum of all terms as,

$$\frac{R_t}{(1+i)^t}$$ Equation 5.2 where,

t: the time of the cash flow

i: the discount rate (the rate of return that could be earned on an investment in the financial markets with similar risk).

Rt: the net cash flow (the amount of cash, inflow minus outflow) at time *t*.

A key variable while calculating the NPV is the discount rate. For the Las Ceibas watershed, the discount rate was taken as the average inflation rate in Colombia over the last five year which is equal to 18%. The main economic costs were the ones associated with labor, supplies and materials. The benefits obtained from the croplands were determined by multiplying the annual yield by the sales price and then by diving this result by the number of hectares. The benefits obtained from the livestock systems were obtained by dividing the total sales by the number of hectares. Regarding the forest, the costs were associated with administration fees paid by the municipality and the benefits were equal to the total tax paid by the water users located downstream. A discount rate of 8% per annum to compute the net present values of commodity production across time was used in the calculations.

The annualized profits were grouped in upper, middle and lower catchments and then averaged given the high similarity among each bracket (Table 5.2). The highest annual profits were in the coffee production in the middle and upper catchments, with more than twenty six thousand and twelve thousand US dollars per hectare. The same activity in the lower catchments had losses at around two thousand US dollars per hectare. The livestock production is also most profitable in the middle catchments followed by the low catchments. The forests presented losses given the management costs associated with its conservation and the almost negligible operating profits gained from this type of activity. There are no data regarding profits and costs in the shrublands.

Table 5.2. Summary of the annualized profit of the main land uses in Las Ceibas watershed.

Watershed	Annualized Profit/ha/USD			
	Livestock	Coffee	Forests	Shrublands
Upper	$336	$12,046	-$853	$0
Middle	$2,689	$26,442	-$124	$0
Low	$1,143	-$2,042	-	$0

As shown in Figure 5.4, the highest total net present value was obtained in the first scenario. The benefits from the livestock production activity with the introduction of rotational grazing increased in fifty years. The lowest net present value corresponds to the reforestation scenario, in which some economic activities such as livestock production were replaced by forests.

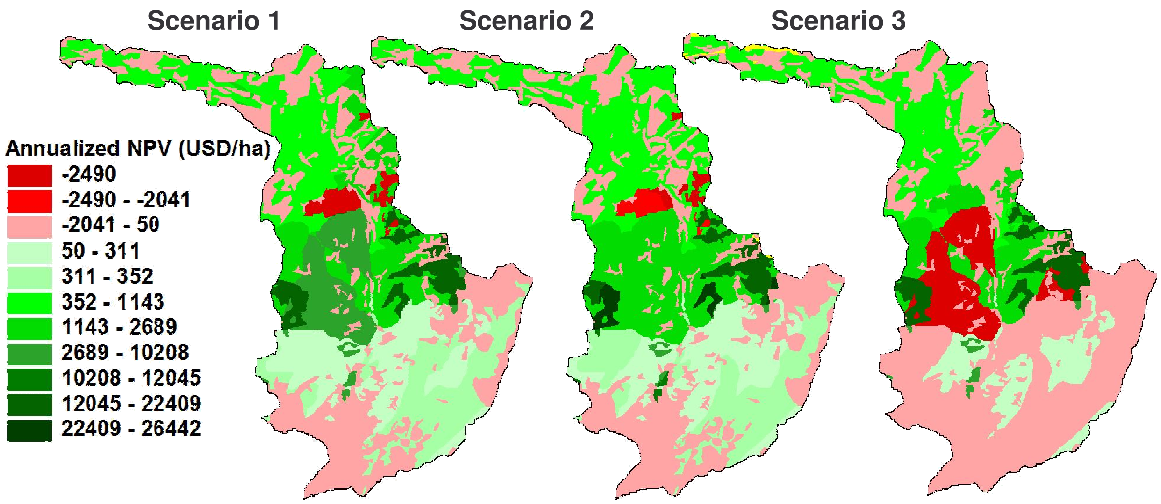

Figure 5.4. Annualized net present value for three land use scenarios.

5.5.2 Environmental index

The environmental index in the whole watershed for each scenario indicates that the highest index was obtained for reforestation (2.55), followed by rotational grazing (1.73), and green manures/cover crops (0.59). The watershed index was obtained from:

$$EI_{watershed} = \frac{\sum (EI_{ij} \times Area_i)}{Area_{watershed}}$$

were j is each of the scenarios and i is the hydrological response unit.

The environmental index is a quantitative approximation of the combined contribution of environmental services and the discounted operational profits for each HRU. Figure 5.5 shows the distribution of the environmental index per hectare among the hydrological response units in the watershed for the three scenarios. In the first scenario, rotational grazing, the highest indexes were obtained in those areas where the land use change took place. In the cover crops and green manures there were no very high indexes as the ones obtained in the other two scenarios. The reforestation scenarios depicted the highest index values among the three scenarios. Regarding the spatial distribution, the highest indexes were obtained in the upper and middle catchments.

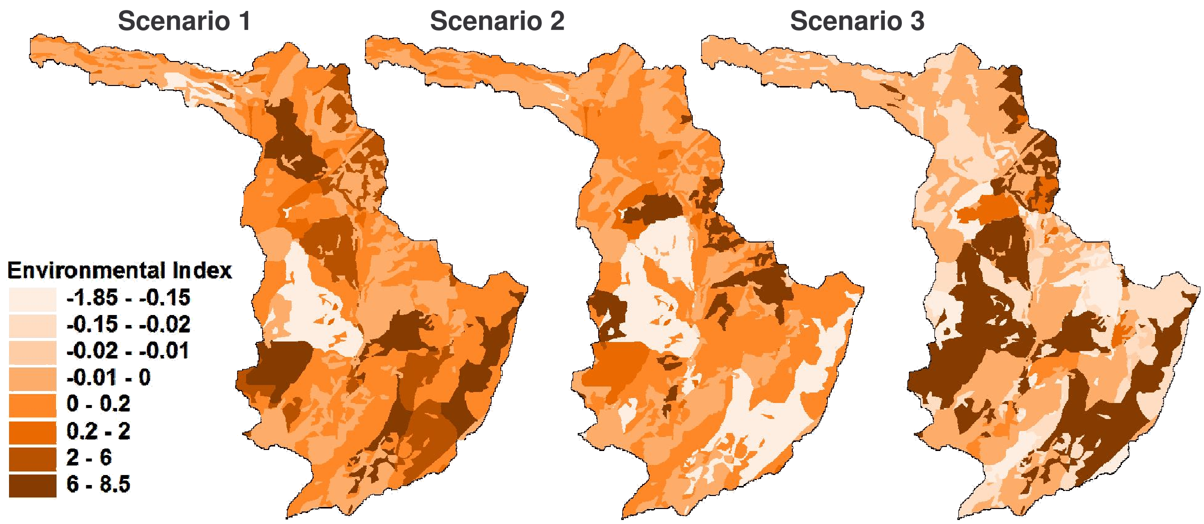

Figure 5.5. Environmental Index for three land use scenarios.

Figure 5.6 shows the contribution of each environmental service and the operational profits NPV for each scenario. The highest contribution to the index in the three scenarios is from carbon sequestration. The lowest contribution was from the sediments except in the reforestation scenario were the contribution from the NPV was lowest among the four elements. In the reforestation scenario the contribution from the NPV of profits was negative given the change of livestock production to forests. If the market value of sequestered carbon was taken into account the results would show a positive percentage in the reforestation scenario.

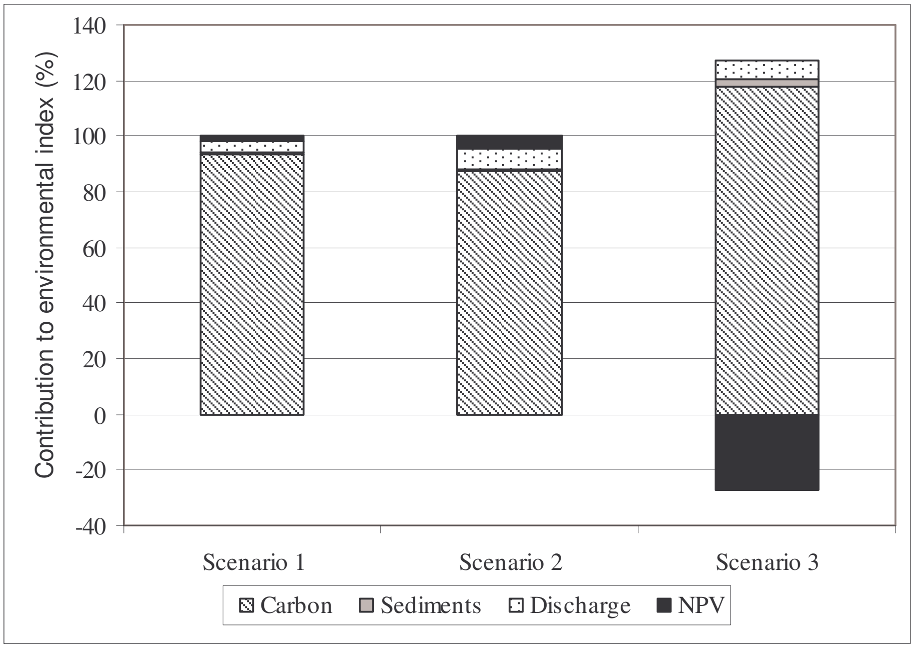

Figure 5.6. Contribution of each environmental service and NPV to the environmental index.

Little evidence of tradeoffs between environmental services was found in the Las Ceibas watershed due to the land use change scenarios. The three changes selected: rotational grazing, green manures and reforestation, enhanced sediment control at the same time they improve discharge regulation and carbon sequestration. There is a clear tradeoff between annualized profits and two environmental services, sediment control and carbon sequestration (Figure 5.7). For example, the reforestation scenario (3) enhanced environmental services but decreased the profits obtained from agricultural production.

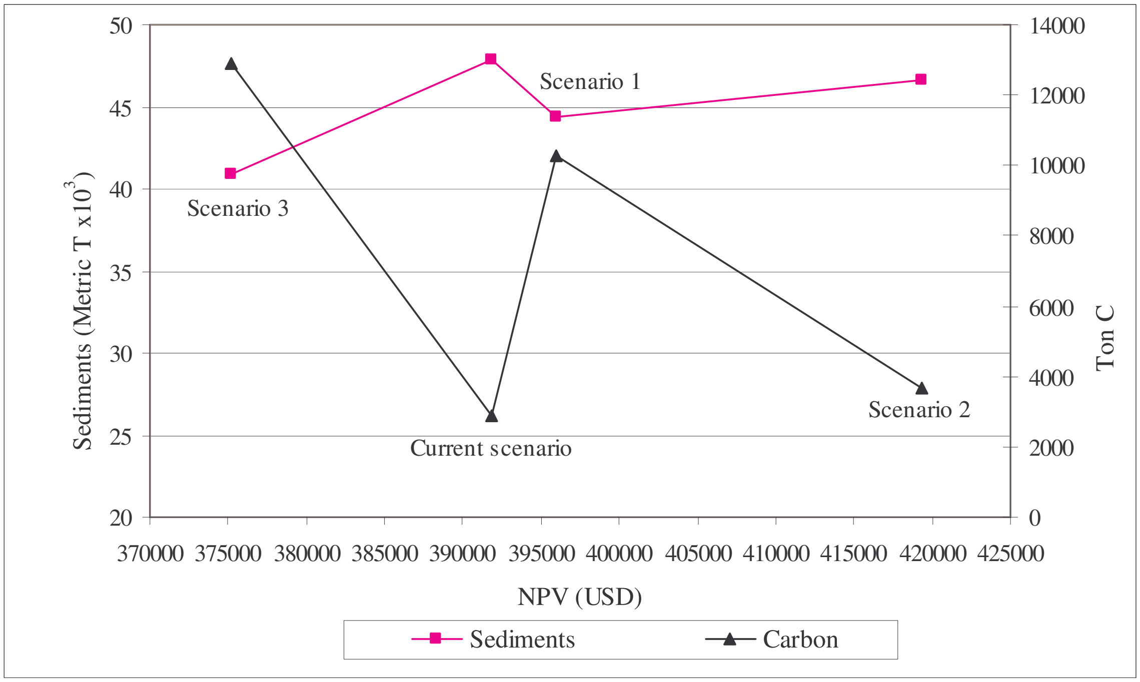

Figure 5.7. Tradeoffs between net present value of annualized profits and sediments and carbon sequestration.

5.6 Conclusions

The environmental index is a quantitative approach that results in the relationship between the efficiency levels of four elements: carbon sequestration, erosion control, discharge regulation and the operational net present value of profits. By applying the index it was possible to calculate the contribution as percentage for each element. In the Las Ceibas watershed the main environmental service to be enhanced was sediments control, which was generating a negative externality to a water supply system downstream. With the index it was possible to identify that even though the policies were mainly targeting sediment control, carbon sequestration was the element contributing the most to positive environmental change given the significant improvements made in this regard. Meanwhile the service that was least impacted in the three scenarios was the sediment control.

Synergies were found between environmental services in the three scenarios. The highest difference compared with current trend was obtained in the reforestation scenario, where the environmental services were positively impacted. Tradeoffs were found between annualized profits from agricultural activities with sediment control, and annualized profits from agricultural activities with carbon sequestration. In this study discharge regulation was seen as a positive impact as the water retention was greater upstream and therefore decreased the flood risk at the catchment outlet. Given that discharge is also needed at the outlet to maintain a natural balance and for human consumption, a minimum level should be established for these purposes. Further research is needed regarding this elemental environmental service and the associated tradeoffs with reforestation.

Based on the environmental index, the recommended scenario would be green manure and cover crops, followed by the rotational grazing from both environmental and economic perspectives. Even though the reforestation scenario gave the most positive environmental impact there is a high tradeoff with profits. Future implementations would need to look at sustainable compensations from selling carbon credits.

It is a challenge for policy makers to generate complementary alternatives that integrate multiple objectives such as job generation, profitability, environmental conservation and social equity. Current ecosystem valuation methodologies are limited by their welfare economic approach and the lack of adequate ecological assessments. Such assessments help to improve the understanding of ecosystem processes and functions in a consistent manner and aid stakeholders to explore management alternatives with a participatory approach. Economic valuation and ecological assessments should be seen as complementary tasks and not as substitutes for policy making.

6 Critique, future work and final reflections

6.1 Summary of research achievements

This research set out to integrate environmental modeling for bundling ecosystem services available from a watershed. This final chapter considers the success in achieving this aim by summarizing the findings of the major chapters and relating them to the three objectives specified in Section 1.3, namely:

1. To assess quantitatively the current situation of hydrological services,
2. To estimate the potential to enhance hydrological services under different land use change scenarios,
3. To assess the carbon sequestration in current land uses, and those land uses selected to improve hydrological services,
4. To estimate the costs and benefits of potential land use change scenarios,
5. To evaluate land use change scenarios based on environmental and economic considerations.

Chapter 3 covered the first objective, presenting the results of a hydrological modeling to quantify hydrological services and looked at scale as a critical issue in spatial science and modeling. Three DEMs with resolutions of 90m, 50m and 30m were used to determine watershed delineation, flow and sediment yields. Although the delineations obtained with the 90m and 50m DEM were less accurate than the results obtained with the 30m DEM, these resolutions still reflected the real topographic features in the landscape. This chapter emphasized the importance of scale and the difficulties faced by geographic research while working with scarce data especially in the developing world.

Chapter 4 focused on objectives two and three, where the contribution of the main land uses to hydrological services and carbon sequestration was investigated through environmental modeling. Three scenarios were analyzed to improve the services, namely: rotational grazing (scenario 1), green manures and cover crops (scenario 2), and reforestation of grasslands (scenario 3). It was possible to find synergies between services at the basin level and the hydrological response unit level. Reforestation, which mainly targeted the increase of carbon sequestration, also improved the provisioning of hydrological services. At the basin level the most positive change in the three environmental services was obtained with reforestation followed by rotational grazing and green manures. The discharge regulation improved more with the cover crops and green manures than with the rotational grazing due to the improvement of water infiltration. Environmental modeling proved to be a powerful tool to quantitatively understand the provisioning of services from different perspectives: basin and sub-basin levels, hydrological response units and land use categories.

Chapter 5 addressed the two last objectives by developing an environmental index to integrate the results from the biophysical analysis and the economic valuation. With the index it was possible to identify synergies and tradeoffs between services. Synergies were found in the three environmental services evaluated, and, tradeoffs were found between the annualized profits from agricultural activities and sediment control together with carbon sequestration.

6.2 Critique of methods

Like most research projects, this one has been a continuing learning process where it is possible to highlight limitations or alternative approaches that could have produced similar or maybe better results. Below are some general issues relating to the methods and specific questions that arose from the case study.

6.2.1 Critique of environmental modeling

The SWAT hydrology model was selected among other distributed models like MIKE-SHE and AGNPS (Agricultural NonPoint Source pollution model), which have the three major components of hydrology, sediments and chemistry, which were needed for this research. MIKE-SHE was not used because of its costs and as it is a more computationally intensive model compared to SWAT. It also requires more data, which was a problem taking into account the restricted availability of data in this tropical region of Colombia where the watershed is located. AGNPS is a lumped-parameter model that uses one time step (storm duration) events and generates a single value for the output variables. Therefore it cannot predict time-varying water, sediment, and chemical discharges which are required for analyses regarding flood warning, floodwater management, watershed assessment, and best management practices evaluations. Additionally, AGNPS does not have a subsurface flow component.

The main advantages of SWAT over other models are: (1) SWAT is a continuous simulation model and useful for analyzing long-term effects of hydrological changes and watershed management practices, especially agricultural practices; (2) The SWAT-GIS linkage incorporates advanced visualization tools capable of statistical analysis of output data; (3) SWAT subdivides large river basins into homogenous parts or what is called Hydrological Response Units (HRUs), which is a useful division when analyzing impacts of land use change practices. The results are also available at subbasin level, which are useful when the watershed is managed based on river coverage.

The main limitations of SWAT are: (1) The SWAT hydrology model is based on the water balance equation. A distributed (Soil Conservation Service) SCS curve number is generated for the computation of overland flow runoff volume, given by the standard SCS runoff equation. Since it is an empirical model and adapted only for US watersheds, the major limitation of using SWAT in this research is that some parameters are well not defined for the tropics; (2) SWAT requires a considerable amount of work for parameterizing the watershed, which in this case resulted in 369 runs to identify the most sensible parameters.

The CO2Fix model was used to calculate the carbon sequestration in the Las Ceibas watershed. It adequately simulated carbon sequestration in tropical regions and it permits the simulation of a wide variety of forest types including agro-forestry systems, selective logging systems, and post harvesting mortality. One of the main limitations of using the CO2Fix model was that there is no basic wood density data specifically for Las Ceibas. It was instead obtained from the literature (Brown 1997). Environmental data like the one required for modeling carbon sequestration is very scarce given the political conflicts of the region during the last decades.

6.2.2 Critique of Environmental index

The index combines the final provisioning of environmental services and the net present value of the profits. The valuation was conducted by applying the cost-benefit analysis, which is a method that does not take into account the social perceptions of the generated values. It also fails to take into account the direct economic effect on the value of other services. Alternate methodologies such as contingent valuation (Pearce 1998), multi-criteria analysis (Proctor & Drechsler 2003), scenario planning (Peterson et al. 2003) or economic games (Cardenas & Carpenter 2008) are future possibilities in which participatory methods integrate the stakeholders in the valuation process and are useful where uncertainty is high.

As it is proposed the environmental index is relative to the current provisioning of services in a watershed. In other words, the index focuses on proportional changes in indicators rather than absolute levels. This fact would constrain this index while comparing two basins with very different initial state. The index will indicate how effective are measurements, meaning that a movement from a negative situation to a bit less negative is rated higher than staying in a positive situation.

6.3 Further research opportunities

6.3.1 Research opportunities for environmental modeling

In a joint modeling output from the field of hydrology and agricultural economics, trade-off relations were established for the Las Ceibas watershed. Even though important local impacts of land use change are lost due to the aggregation process. Therefore it is crucial to look at the effect of land use change in a spatially distributed way to assess the range of the local impact and to develop the required protection measures associated with land use changes. Complementary field work to collect data at local level regarding suspended sediments, carbon content and curve number could improve the current data and therefore the parameterization of the model. These data can be used for a better calibration of the model. In this research SWAT was calibrated with a single point located at the outlet, which measured discharge and suspended sediments (Guayabo station). A major improvement of the model for the Las Ceibas watershed, would be the calibration of discharge and sediments in at least other point upstream of the basin.

Further research regarding the impact of reforestation on water yield in the tropics will clarify the interaction between services produced in upper catchments and downstream users. Reforestation is sometimes seen as the best option for carbon sequestration but depending on the tree species it could have a large impact on downstream stakeholders and the surrounding environment.

Another erosion control mechanism is a vegetated buffer as corridors along the stream. A better protection can be provided by increasing the streamside protection buffer, which can also have positive impact on the local habitat. This scenario could be modeled as a fourth alternative in the Las Ceibas watershed, where sites with focalized erosion need this type of measurements.

6.3.2 Research opportunities for bundling of environmental services

This research focused mainly in the biophysical understanding of the provisioning of environmental services. Further research regarding the social context of the watershed would be required as a parallel step to clearly identify the trade-offs when establishing a PES scheme, especially at the local level. Identification and characterization of the beneficiaries, modifiers and intermediaries of the services will provide the tools to understand the cause-effect relationships between the provisioning of multiple services. For example, questions to these stakeholders regarding their level of their discretion over the way the ecosystem is used and managed, their rights to modify the structure of the ecosystem or their access to alternative supplies of the ecosystem services or good substitutes for those services can help to identify the relation between actors and between actors and services.

Another important step forward is to look at other environmental services in the Las Ceibas watershed, especially biodiversity which is important in that region of the tropical forests of Colombia. Other environmental services as landscape beauty, provision of opportunities for tourism and water quality regarding nitrates and phosphates are also relevant for Las Ceibas. Bundling this kind of services with those already studied could improve the opportunities to setup PES.

6.4 Final reflections

This thesis provides a way forward for modeling the bundling of environmental services, and forms part of an ongoing worldwide research on PES schemes. It has shown that synergies between services can be achieved when targeting a particular service or land use change. It demonstrates the importance of environmental modeling for building a more comprehensive picture of the patterns and relationships in spatial phenomena in a watershed.

Since the start of this thesis, many papers, books and theses have been published and have made significant contributions to the literature on environmental services and modeling. These investigations are continually improving our ability to increase our understanding of synergies and trade-offs between environmental services and their relation with the actors involved. Sometimes the results from multiple models and multiple scales are more complex or more difficult to interpret than previous analysis has shown. Sometimes the results are counter-intuitive or suggest that past assumptions may not be valid across the board. Understanding and representing this complexity of environmental modeling is a key goal of enhancing environmental services across scales.

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Appendix A: SWAT input Data

Crop Input Data

ICNUM	CPNM	IDC	CROPNAME	BIO_E	HVSTI	BLAI	FRGRW1
1	CAFT	6	Cafe tecnificado	39.00	0.55	4.00	0.15
2	WATR	6	Rios	25.00	0.50	6.00	0.15
3	BOSQ	7	Bosque	15.00	0.76	5.00	0.05
4	VEPA	7	Paramo	47.00	0.90	6.00	0.10
5	CEPO	4	Centros poblados	25.00	0.50	2.50	0.15
6	PACU	6	Pastos y cultivos	33.00	0.80	3.75	0.08
7	PANM	6	Pastos no manejados	35.00	0.90	4.00	0.05
8	CUME	4	Cultivos generales medio	27.50	0.50	3.20	0.15
9	CUCA	4	Cultivos generales calido	30.00	0.70	4.00	0.15
10	PARA	6	Pastos y rastrojos	35.00	0.90	3.50	0.05
11	RAST	6	Rastrojo	34.00	0.90	2.00	0.05
12	CANA	6	Cana de azucar	25.00	0.50	6.00	0.15
13	RABO	6	Rastrojos y bosques	28.00	0.90	3.00	0.05
14	ARRO	4	Arroz	22.00	0.50	5.00	0.30
15	BOSE	7	Bosque secundario	15.00	0.76	5.00	0.05
16	PLFO	7	Plantaciones forestales	15.00	0.76	5.00	0.15
17	SUPP	6	Superparamo	18.00	0.90	1.00	0.05
18	AERO	6	A. Erosionadas	18.00	0.90	1.00	0.05
19	SUBP	6	Subparamo	40.00	0.90	5.00	0.08
20	BOAA	7	Bosque alto-andino	15.00	0.76	5.00	0.05
21	CUFR	4	Cultivos generales frio	25.00	0.50	2.50	0.15
22	SINV	4	sin Vegetacion	25.00	0.50	2.50	0.15
23	CUPA	4	Cultivos generales paramo	25.00	0.40	2.50	0.12
24	XERO	6	Vegetacion xerofitica	18.00	0.90	1.00	0.05
25	CATR	6	Cafe tradicional	30.00	0.60	4.50	0.10

Crop Input Data

CPNM	FRGRW2	LAIMX2	DLAI	CHTMX	RDMX	T_OPT	T_BASE	CNYLD
CAFT	0.50	0.95	0.70	2.00	1.50	24.00	10.00	0.0140
WATR	0.50	0.95	0.75	3.00	2.00	25.00	11.00	0.0015
BOSQ	0.40	0.95	0.99	12.00	3.00	22.00	8.00	0.0015
VEPA	0.20	0.95	0.70	2.50	2.20	11.00	3.00	0.0160
CEPO	0.60	0.95	0.90	1.25	1.00	20.00	8.00	0.0300
PACU	0.50	0.95	0.96	0.90	0.90	20.00	8.00	0.0250
PANM	0.49	0.95	0.99	0.50	0.60	20.00	8.00	0.0234
CUME	0.60	0.95	0.90	2.00	1.50	21.00	7.00	0.0275
CUCA	0.50	0.95	0.65	1.50	1.25	22.00	7.00	0.0150
PARA	0.42	0.90	0.80	0.70	0.60	22.00	10.00	0.0200
RAST	0.25	0.70	0.35	1.00	1.20	22.00	10.00	0.0160
CANA	0.50	0.95	0.75	3.00	2.00	25.00	11.00	0.0015
RABO	0.30	0.80	0.55	4.00	1.70	22.00	9.00	0.0117
ARRO	0.70	0.95	0.80	0.50	0.90	25.00	10.00	0.0136
BOSE	0.40	0.95	0.99	3.00	2.00	20.00	10.00	0.0015
PLFO	0.25	0.99	0.99	10.00	3.00	25.00	3.00	0.0015
SUPP	0.30	0.70	0.35	0.20	0.10	11.00	1.00	0.0160
AERO	0.30	0.70	0.35	0.20	0.10	11.00	1.00	0.0160
SUBP	0.20	0.80	0.50	2.50	2.20	12.00	7.00	0.0160
BOAA	0.40	0.95	0.99	3.00	2.50	18.00	8.00	0.0015
CUFR	0.60	0.95	0.90	1.25	1.00	20.00	8.00	0.0300
SINV	0.60	0.95	0.90	1.25	1.00	20.00	8.00	0.0300
CUPA	0.70	0.95	0.80	0.80	1.00	20.00	9.00	0.0280
XERO	0.30	0.70	0.35	0.20	0.10	11.00	1.00	0.0160
CATR	0.50	0.95	0.80	4.50	2.50	24.00	10.00	0.0090

Crop Input Data

CPNM	CPYLD	BN1	BN2	BN3	BP1	BP2	BP3	WSYF	USLE_C
CAFT	0.0016	0.0470	0.0177	0.0138	0.0048	0.0018	0.0014	0.550	0.020
WATR	0.0001	0.0100	0.0040	0.0025	0.0075	0.0030	0.0019	0.010	0.028
BOSQ	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003	0.010	0.022
VEPA	0.0150	0.0350	0.0150	0.0038	0.0014	0.0010	0.0007	0.900	0.012
CEPO	0.0040	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.450	0.083
PACU	0.0035	0.0600	0.0231	0.0134	0.0084	0.0032	0.0019	0.760	0.030
PANM	0.0033	0.0600	0.0231	0.0134	0.0084	0.0032	0.0019	0.900	0.020
CUME	0.0037	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.450	0.069
CUCA	0.0017	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.550	0.044
PARA	0.0030	0.0600	0.0231	0.0134	0.0084	0.0032	0.0019	0.900	0.028
RAST	0.0022	0.0200	0.0120	0.0050	0.0014	0.0010	0.0007	0.900	0.023
CANA	0.0001	0.0100	0.0040	0.0025	0.0075	0.0030	0.0019	0.010	0.028
RABO	0.0016	0.0200	0.0120	0.0050	0.0014	0.0010	0.0007	0.630	0.018
ARRO	0.0013	0.0500	0.0200	0.0100	0.0060	0.0030	0.0018	0.250	0.010
BOSE	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003	0.010	0.025
PLFO	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003	0.600	0.014
SUPP	0.0022	0.0350	0.0150	0.0038	0.0014	0.0010	0.0007	0.900	0.050
AERO	0.0022	0.0350	0.0150	0.0038	0.0014	0.0010	0.0007	0.900	0.069
SUBP	0.0150	0.0350	0.0150	0.0038	0.0014	0.0010	0.0007	0.900	0.002
BOAA	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003	0.010	0.020
CUFR	0.0040	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.450	0.083
SINV	0.0040	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.450	0.083
CUPA	0.0034	0.0550	0.0200	0.0120	0.0060	0.0025	0.0019	0.400	0.100
XERO	0.0022	0.0350	0.0150	0.0038	0.0014	0.0010	0.0007	0.900	0.069
CATR	0.0011	0.0470	0.0177	0.0138	0.0048	0.0018	0.0014	0.350	0.012

Crop Input Data

CPNM	GSI	VPDFR	FRGMAX	WAVP	CO2HI	BIOEHI	RSDCO_PL	OV_N
CAFT	0.005	4.000	0.750	8.000	660.000	45.000	0.050	0.14
WATR	0.005	4.000	0.750	10.000	660.000	33.000	0.050	0.01
BOSQ	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.10
VEPA	0.005	4.000	0.750	8.500	660.000	54.000	0.050	0.15
CEPO	0.007	4.000	0.750	7.500	660.000	33.000	0.050	0.10
PACU	0.006	4.000	0.750	9.000	660.000	35.000	0.050	0.12
PANM	0.005	4.000	0.750	10.000	660.000	36.000	0.050	0.15
CUME	0.007	4.000	0.750	6.000	660.000	34.000	0.050	0.09
CUCA	0.005	4.000	0.750	9.500	660.000	37.000	0.050	0.07
PARA	0.005	4.000	0.750	10.000	660.000	37.000	0.050	0.24
RAST	0.005	4.000	0.750	10.000	660.000	39.000	0.050	0.50
CANA	0.005	4.000	0.750	10.000	660.000	33.000	0.050	0.09
RABO	0.004	4.000	0.750	9.500	660.000	32.000	0.050	0.40
ARRO	0.008	4.000	0.750	5.000	660.000	31.000	0.050	0.04
BOSE	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.10
PLFO	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.10
SUPP	0.005	4.000	0.750	10.000	660.000	31.000	0.050	0.06
AERO	0.005	4.000	0.750	10.000	660.000	31.000	0.050	0.04
SUBP	0.005	4.000	0.750	9.000	660.000	50.000	0.050	0.20
BOAA	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.10
CUFR	0.007	4.000	0.750	7.500	660.000	33.000	0.050	0.12
SINV	0.007	4.000	0.750	7.500	660.000	33.000	0.050	0.04
CUPA	0.007	4.000	0.750	8.000	660.000	33.000	0.050	0.10
XERO	0.005	4.000	0.750	10.000	660.000	31.000	0.050	0.04
CATR	0.004	4.000	0.750	8.000	660.000	33.000	0.050	0.14

Crop Input Data

CPNM	CN2A	CN2B	CN2C	CN2D	FERTFIELD
CAFT	67.00	77.00	83.00	87.00	FALSE
WATR	92.00	92.00	92.00	92.00	FALSE
BOSQ	36.00	60.00	73.00	79.00	FALSE
VEPA	35.00	56.00	73.00	81.00	FALSE
CEPO	31.00	59.00	72.00	79.00	FALSE
PACU	54.00	71.00	80.00	85.00	FALSE
PANM	49.00	69.00	79.00	84.00	FALSE
CUME	67.00	77.00	83.00	87.00	TRUE
CUCA	58.00	73.00	80.00	85.00	TRUE
PARA	46.00	67.00	78.00	83.00	FALSE
RAST	36.00	57.00	71.00	78.00	FALSE
CANA	67.00	77.00	83.00	87.00	TRUE
RABO	36.00	58.00	71.00	78.00	FALSE
ARRO	90.00	90.00	90.00	90.00	TRUE
BOSE	45.00	66.00	77.00	83.00	FALSE
PLFO	40.00	62.00	74.00	80.00	FALSE
SUPP	72.00	84.00	90.00	93.00	FALSE
AERO	72.00	84.00	90.00	93.00	FALSE
SUBP	35.00	56.00	72.00	80.00	FALSE
BOAA	33.00	57.00	70.00	74.00	FALSE
CUFR	67.00	77.00	83.00	87.00	TRUE
SINV	77.00	86.00	91.00	94.00	FALSE
CUPA	67.00	77.00	83.00	87.00	TRUE
XERO	72.00	84.00	90.00	93.00	FALSE
CATR	58.00	72.00	80.00	85.00	FALSE

Soil Input data

SUBBASIN	HRU	LANDUSE	SNAM	NLAYERS	HYDGRP	SOL_ZMX
1	1	RAST	VXBa	4	B	1500.00
1	2	RAST	PXEe3	2	A	300.00
1	3	PANM	VXBa	4	B	1500.00
1	4	PANM	PXEe3	2	A	300.00
2	1	CATR	MRAf2	2	B	400.00
2	2	RAST	MRAf2	2	B	400.00
2	3	RAST	MXAf2	3	B	900.00
2	4	PANM	MRAf2	2	B	400.00
2	5	PANM	MXAf2	3	B	900.00
2	6	SINV	MRAf2	2	B	400.00
2	7	SINV	MXAe	3	B	900.00
2	8	BOSQ	MRAf2	2	B	400.00
2	9	BOSQ	MXAf2	3	B	900.00
3	1	RAST	MXAf2	3	B	900.00
3	2	RAST	PXEe3	2	A	300.00
3	3	PANM	MXAd	3	B	900.00
3	4	PANM	MXAf2	3	B	900.00
4	1	CATR	MQCf2	4	B	1100.00
4	2	CATR	MQAg2	3	B	1500.00
4	3	PANM	MQAg2	3	B	1500.00
5	1	RAST	MQAg2	3	B	1500.00
5	2	RAST	MRAf2	2	B	400.00
5	3	PANM	MQAg2	3	B	1500.00
5	4	PANM	MRAf2	2	B	400.00
6	1	CATR	MQEg2	3	B	1500.00
6	2	PANM	MRAf2	2	B	400.00
6	3	PANM	MQEg2	3	B	1500.00
7	1	PANM	MRAf2	2	B	400.00
7	2	PANM	MQEg2	3	B	1500.00
8	1	CATR	MQEg2	3	B	1500.00
8	2	PANM	MQEg2	3	B	1500.00
8	3	BOSQ	MQEg2	3	B	1500.00
9	1	CATR	MQEg2	3	B	1500.00
9	2	PANM	MQEg2	3	B	1500.00
10	1	CATR	MQEg2	3	B	1500.00
10	2	PANM	MQEg2	3	B	1500.00
10	3	BOSQ	MLBf2	4	B	870.00
10	4	BOSQ	MQEg2	3	B	1500.00

SUBBASIN	HRU	LANDUSE	SNAM	NLAYERS	HYDGRP	SOL_ZMX
11	2	RAST	MQEg2	3	B	1500.00
11	3	PANM	MQEg2	3	B	1500.00
12	1	PANM	MQAg2	3	B	1500.00
12	2	PANM	MLBf2	4	B	870.00
12	3	BOSQ	MLBf2	4	B	870.00
13	1	CATR	MQAg2	3	B	1500.00
13	2	CATR	MLBf2	4	B	870.00
13	3	PANM	MQAg2	3	B	1500.00
13	4	BOSQ	MQAg2	3	B	1500.00
13	5	BOSQ	MLBf2	4	B	870.00
14	1	PANM	MQEg2	3	B	1500.00
15	1	RAST	MQEg2	3	B	1500.00
15	2	PANM	MQEg2	3	B	1500.00
16	1	RAST	MLBf2	4	B	870.00
16	2	RAST	MQEg2	3	B	1500.00
16	3	PANM	MLBf2	4	B	870.00
16	4	PANM	MQEg2	3	B	1500.00
16	5	BOSQ	MLBf2	4	B	870.00
16	6	BOSQ	MQEg2	3	B	1500.00
17	1	RAST	MLBf2	4	B	870.00
17	2	PANM	MLBf2	4	B	870.00
17	3	PANM	MQEg2	3	B	1500.00
17	4	BOSQ	MLBf2	4	B	870.00
18	1	PANM	MLBf2	4	B	870.00
18	2	PANM	MQEg2	3	B	1500.00
18	3	BOSQ	MLBf2	4	B	870.00
19	1	PANM	MLBf2	4	B	870.00
19	2	BOSQ	MLBf2	4	B	870.00
20	1	PANM	MQAg2	3	B	1500.00
20	2	PANM	MLBf2	4	B	870.00
20	3	BOSQ	MLBf2	4	B	870.00
21	1	RAST	MLBf2	4	B	870.00
21	2	PANM	MQAg2	3	B	1500.00
21	3	PANM	MLBf2	4	B	870.00
21	4	BOSQ	MLBf2	4	B	870.00
22	1	CATR	MRAf2	2	B	400.00
22	2	RAST	MXAf2	3	B	900.00
22	3	PANM	MXAf2	3	B	900.00