Valuing water and sediment tradeoffs between forest and pasture in montane tropical environments in Puerto Rico

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Abstract

Effective land use policy must weigh both the private and public costs and benefits of converting forests to alternate land uses. This project assesses the private and public impacts of forest to pasture conversion in the montane regions of Puerto Rico. Due to the island's water supply problems, hydrologic ecosystem services were found to be the most significant resource impacted. The value of carbon sequestration lost through conversion was found to range from 9-36 $/ha/yr. The value of other ecosystem services, notably recreation and biodiversity, were found to be highly significant in certain localities but small on an average island-wide basis. The model created in this study found that the public costs of reservoir sedimentation resulting from increased erosion and the higher incidence of landslides on pastures outweigh the public benefits of increased runoff in areas where with slopes of approximately 21^{o} and a Revised Universal Soil Loss Equation topographic factor greater than 6.5. Results were highly dependent on the amount of sediment that is transported from the pasture to the reservoir (e.g. the sediment delivery ratio) and the marginal value of water. The private returns to pasture (400 $/ha/yr) were generally found to be greater than the sum of the public costs. The results suggest that policy-makers should take local environmental variation into account when designing forest conservation strategies. Policies should target areas with high slopes and high sediment delivery ratios.

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1. Introduction

The design of a socially optimal land use policy must consider both the private and public costs and benefits of land use changes. In many tropical montane areas, forest conversion to pasture is a financially profitable land use option for landowners. At the same time, this change in land use has consequences for the public provision of ecosystem services, including the quantity and quality of water available to downstream users. These public consequences are often assumed to be negative, and there are many programs in the tropics designed to promote forest conservation. However, the scientific and economic literatures suggest that forest conversion to pasture may have both positive and negative impacts on public benefits. This project provides insight into conditions under which conversion from forest to pasture may be socially optimal by evaluating the hydrologic externalities associated with forest-to-pasture conversions in the humid subtropical lifezone of Puerto Rico. Data on the economic benefits and costs provided by Puerto Rico's forests and pastures is collected and presented. Carbon sequestration, biodiversity, air quality improvements, recreation, existence values, and hydrologic ecosystem services are considered. Implications for tropical forest conservation policies are discussed.

1.1 Forest Conversions and Ecosystem Services

Ecosystem services are "ecological functions that sustain human life and create value for human users" (Daily 1997). These services can be classified into four general categories: provisioning services, regulating services, supporting services, and cultural services (Millennium Ecosystem Assessment 2003). Provisioning services include the direct provision of goods such as timber, food, and fuel. Regulating services include climate regulation and flood control. Supporting services include pollination and soil formation and other functions necessary to maintain other services such as biodiversity. Cultural services include the benefits that people receive from recreational use as well as aesthetic values.

Increased recognition of the value of ecosystem services provided by tropical forests has led to international concern over deforestation in the tropics. Forest preservation has become a major goal of international policy (Kyoto Protocol, 1998) and interest in developing incentives to encourage forest conservation and reforestation has risen. Unfortunately, the social benefits provided by the conservation of tropical forests are external to the decisions of local landowners, and therefore by economic theory will be under-provided by the market (Daily, 1997). Policy mechanisms such as land conservation easements and transfer payments have been designed to correct this market failure. However, in order to achieve an efficient allocation of land use, such mechanisms must accurately value and account for all the services provided.

In response to the needs of policy-makers, there has been increased interest in evaluating ecosystem services. Moreover, methods of ecosystem service valuation have been refined and improved since the most highly publicized early attempt to quantify the benefits of the global biosphere (Costanza et al., 1997). Common methods used to evaluate ecosystem services include Avoided Cost (AC), Replacement Cost (RC), Contingent Valuation Method (CVM) and Hedonic Pricing (HP) (Farber et al., 2002; Zhang and Li, 2005). Each method has strengths and weaknesses and efforts have been taken to develop an appropriate and common framework for evaluating ecosystem services on a global scale (Millennium Ecosystem Assessment, 2003) Government agencies such as the United States Environmental Protection Agency have also prepared guidelines for evaluating ecosystem services in order to inform policy choices (EPA, 2000).

Ecosystem services provided by tropical forests have been evaluated using a variety of methods (see EEP, 2003). Unfortunately, inconsistencies in their methods make direct comparisons difficult. Furthermore, most studies only analyze one function, while multiple-function studies and those that assess before-and-after states are the most relevant (Turner et al., 2003). Although economic valuation techniques have become more sophisticated and are widely accepted as valid, studies suffer from inadequate site-specific data and improper assumptions about ecological processes. Furthermore, policy decisions based on economic valuations may be complicated by differences of scale. In some cases the benefits of forest conservation may outweigh alternate land uses on global scale, but not on a local or national scale (Kremen et al., 2000). Finally, full economic valuation is hampered by the difficulty in accurately quantifying ecological processes that influence economic costs and benefits. Failure to consider all costs and benefits at the proper scale can lead to mistakes in determining whether the net effect on public benefits is positive or negative (Aylward, 2002).

1.2 Hydrologic Ecosystem Services and Tropical Forest Conversion

Hydrologic ecosystem services are ecosystem services related to the provision of water for human use. The provision of an adequate supply of clean water is a major concern to local and regional policy-makers and the majority of payments for ecosystem services programs in the tropics are related to water (Pagiola et al., 2004). In Puerto Rico and elsewhere, forest conservation and watershed protection have been suggested as ways of protecting the water supply (DRNA, 2005). Landscape and stream channel response to forest to pasture conversions can be considerable (Vanacker et al., 2005). However the quality and quantity of water available for human use depends on a complex variety of factors including climate, topography, vegetation, soil conditions, and anthropogenic factors. Land use change intimately affects hydrologic regimes, as different types of vegetation use different quantities of water, and impact soil properties in different ways. These relationships are complex and often site specific (Bonell and Bruijnzeel, 2004).

To evaluate the economic implications of land use change on hydrologic ecosystem services, the resulting change in hydrologic functions must be quantified. Conventional wisdom holds that forests increase rainfall, prevent floods, increase streamflow totals, increase flows during dry periods, prevent landslides, and reduce erosion. This suggests that all the hydrologic externalities associated with forest conversion are negative and that forest is always best land cover for hydrologic ecosystem service provision. However, the scientific literature suggests that this view is overly simplistic and that changes in hydrologic function due to conversion of forests are site-specific and highly dependent upon the subsequent land use and management practices (Bruijnzeel, 2004).

Recent literature reviews of literature on changes of hydrological function due to tropical forest conversion to pasture suggest that when forests are converted to pasture (Aylward, 2002; Bruijnzeel, 2004):

- Onsite erosion rates increase.
- Offsite sedimentation rates increase but the amount depends on the processes that remove sediments and the subsequent land use.
- Nutrient and chemical exports in stream water increase.
- Annual water yield increases
- Peak flow may increase but effects are often diminished downstream
- Dry season flows usually decrease but may increase depending on site-specific conditions.
- Effect on groundwater recharge is usually similar to effect on seasonal flows.
- Local precipitation is not significantly affected by marginal changes in forest cover, but effects of deforestation may be significant on regional scales.

The importance of considering study area-specific information is made particularly clear by the fact that there are exceptions to these general rules. For example, in upper montane cloud forests, where the interception of horizontal precipitation and cloud water makes up for evapotranspiration losses, a reduction in forest cover is likely to lead to a decrease in water yield rather than an increase (Bruijnzeel, 2001). Likewise, in some areas, depending on the type of vegetation and soil conditions, converting forest to pasture may lead to an increase in evapotranspiration and a decrease in annual water yield. During parts of the year when grasses grow vigorously, grasses may decrease streamflow in comparison to forest (Hibbert 1969). Annual runoff decreased when a lowland forested wetland in Puerto Rico was converted to tall, dense, deep-rooted grasses (van der Molen, 2002).

Aylwald (2002) reviewed the existing literature on valuations of hydrologic externalities. The majority of the literature on water quality focuses on costs of erosion. Economic costs incurred due to increased erosion and sedimentation include the loss of hydroelectric power generation and irrigation production due to reservoir storage loss, increases in operation and maintenance costs of water supply, dredging costs, losses to commercial fishers, and loss of tourism or recreational benefits. Although generally the economic consequences of changes in water quality resulting from deforestation are negative, results of studies on the magnitude of economic impact are mixed. Aylwald cites a study from the Philippines that estimates the cost of sedimentation of a downstream reservoir to be under one U.S. cent per hectare per year (Cruz et al., 1988).. However, others suggest that the benefits from erosion control outweigh the opportunity cost of forest conservation (Pimentel, 1995; Clark 1985). Several studies conclude that erosion costs are significant in the production of hydroelectric power in Latin America and the Caribbean (Veloz, 1985; Southgate and Macke, 1989). In contrast another study found that sedimentation in Costa Rican dam provided a benefit during the dry season because it occupied the dead storage of hydroelectric reservoirs and actually increased power production (Aylward and Echeverría, 2001).

Other aspects of water quality impact due to land use change have not been extensively valuated in the literature, but may be important. No studies were found that specifically valued changes in nutrient or chemical outflows due to land use change. One aspect of water quality cost that was not included in this review is the economic effects on housing or land value prices due to perceived differences in water quality. Nevertheless, Steinnes (1991) demonstrated that land values are negatively impacted by increased cloudiness of water, whether or not this is correlated with other scientific measures of water quality. Although the focus of this study is on externalities, it should be noted that soil conservation may have on-site as well as offsite benefits and costs (Thao, 2001; Pimentel, 1995).

The majority of studies reviewed indicated that tropical forest conversion to pasture increases annual water yield to some extent. These increases are considered an positive economic benefits. However, some studies describe water quantity benefits that are not supported by scientific evidence or contain other methodological problems (Aylward, 2002). In addition, the magnitude of benefits from changes in water yield and quality depends on the use of the water and the economic valuation technique used.

In summary, both the scientific economic literature suggest that the economic implications of forest conversion will be highly dependent on site-specific conditions, the magnitude of the different changes in hydrologic functions (both positive and negative), and the economic context relevant to valuation. Therefore, ideally, when designing incentive policies site-specific or region-specific data should be used. Unfortunately, gathering appropriate site-specific ecological and economic data is often time consuming and expensive. The problem for policy makers may be simplified if relevant pre-existing data is analyzed and region or ecosystem specific decision rules are developed. This project analyzes the cost and benefits of forest to pasture conversions in the subtropical wet forests in Puerto Rico with a focus on hydrologic ecosystem services.

2. Description of Study Area and Problem

2.1 Study Area

This case study assesses the ecosystem service values of forests and pastures in the lower montane regions of central Puerto Rico. Puerto Rico is the smallest of the islands in the Greater Antilles. Located at approximately 18^{o}N, 66^{o}W, Puerto Rico is located in the trade-winds. The climate is tropical, with much of the annual precipitation occurring between May and October. Due to the island's location, hurricanes are common. Much of the rainfall is orographic and the island's topography is dominated by an east-west trending central mountain range.

This study focuses on the subtropical wet forest zone, which supplies much of the island's water. Subtropical wet forest covers approximately 24% of the island (Ewel and Whitmore, 1973). Approximately 71% of the area of this lifezone is protected (Helmer, 2004). In Puerto Rico, the dominant native forest type in this lifezone is the Tabonuco-type forest, which is named for the dominant tree species (*Dacryodes* excelsa). This forest-type covers much of Puerto Rico's mountains, including the Luquillo mountains in the northeast and the Cordillera Central range, at elevations ranging from about 250 to 600. Mean monthly temperatures range between 21^{o}C and 25^{o}C. This zone receives abundant rainfall, and has a mean precipitation ranging from 2000-4000 mm per year. A significant amount of runoff is generated year round, more than 1600 mm per year at some recording stations (García-Martinó et al., 1996).

Since the 1940's, a shift from agriculture to urbanization has led to migration from the central mountains to urban areas along the northeast coast (Grau et al., 2003). This abandonment of marginal cropland has led to reforestation, particularly on the steep mountain regions. Since 1936, the dominant trend in the northeast mountains of Luquillo was conversion from intensive agriculture to secondary forest, although at the same time urban pressure has increased (Thomlinson et al., 1996). Since the 1970's, the total area of land in pasture increased as agricultural lands were converted to pasture, but at the same time both marginal abandoned agricultural and pasture lands have reverted to forest (Helmer, 2004).

Much of the land in the subtropical wet forest zone is too steep for mechanized agriculture (Caro-Costas and Vincente-Chandler, 1974). The average slope in this area was found to be 11 degrees (Author's calculations based on Digital Elevation Model from NSF Biocomplexity project). Shade-coffee has historically grown successfully in this region and a large portion of the life zone is covered by active and abandoned coffee plantations. A common alternate land use is pasture for cattle grazing. However, historically pasture management has been a problem because of weed growth problems and soil compaction (Ewel and Whitmore, 1973). Although the island does not have a significant timber industry, teak (*Tectona grandis*) and mahogany (*Swietenia macrophylla*) have grown well in plantations in this area.

2.2 Puerto Rico's Water Supply Problems and Forest Conservation Incentive Programs

The provision of an adequate supply of clean water for competing uses is becoming increasingly difficult in Puerto Rico. The growing population, urbanization and industrialization has led to increased water consumption and increased competition for the island's limited water supply. At the same time, sedimentation of the island's reservoirs has led to decreased storage capacity, and contamination of groundwater has led to decreased aquifer withdrawals (Hunter and Arbona, 1995). The significance of the water supply problem in Puerto Rico was highlighted in the 1994-1995 drought. During this drought, strict water rationing affected more than 1 million people in the San Juan metropolitan area. The resulting agricultural losses were valued at $165 million (Larsen, 2000). The fact that a comparable drought in 1966-1968 did not result in the need for water rationing suggests that demand, storage capacity, water production and losses, and per capita consumption have become increasingly important (Larsen, 2000).

The domestic sector is the dominant consumer of water in Puerto Rico. In 2004, domestic use accounted for 89% of the total 673 MGD consumed, or 598 MGD (DRNA, 2006). Seventy percent of the island's potable water is supplied by 29 reservoirs (Zayas et al., 2004). These reservoirs provide about 390 million gallons per day to the filtration plants of the Autoridad de Acueductos y Alcantarillados (AAA) (also known as the Puerto Rico Aqueduct and Sewer Authority (PRASA)). The total secure yield of the island's reservoirs is estimated at 502 MGD (DRNA, 2006).The Autoridad de Energía Eléctrica (AEE) operates hydroelectric plants in 12 of the dams, generating about 119,501 MwH per year, about 1.9% of total energy produced by AEE. Due to the fact that the proportion of average annual inflow that reservoirs can store is low (in the range of 4-40% for selected reservoirs), the reservoirs cannot always meet demand and are rapidly depleted in periods of below-average rainfall (Soler-López, 2001). The lack of adequate water supply is exacerbated by the fact that approximately 43% of water produced by AAA is lost in transmission (DRNA, 2005).

Sedimentation causes many problems for the water supply system in Puerto Rico. On average, every day approximately 1,000 people lose water service because of shutdowns of filtration plants due to high turbidity in the water (A. Garcia of PRASA, electronic communication, February 23, 2006). Individuals frequently incur costs to compensate for the unreliability of the water supply system. For example, some people install water-catching mechanisms at their residences. Others collect water from spouts in the mountains.

Reservoir capacity loss due to sedimentation is also a major problem (Table 2.2.1). Most of the reservoirs were constructed in the early and middle 20th century. All have experienced capacity losses due to sedimentation. Of the 14 major Puerto Rican reservoirs surveyed by the United States Geological Survey (USGS), capacity losses ranged from 12-81%, with an average loss of 35% (Soler-López, 2001). Annual rates of sedimentation have ranged from 4.6 to 277 acre-ft (5.6 – 341 thousand m^3) per year, and are highest in water basins in the north and east where there is the most rainfall and the most development (Zayas et al., 2004). In part due to changing land uses, most of the depositional rates of sediments into the reservoirs exceed design rates (Soler-López, 2001). Many of the reservoirs have zero dead storage and operate at full capacity (DRNA, 2006) Major floods and hurricanes also significantly increase deposition rates, and sedimentation surveys have shown that capacity losses may be two to five times higher in periods with hurricanes (Soler-López, 2001).

Table 2.2.1. Characteristics of seven major reservoirs in Puerto Rico draining subtropical wet forests. Soler-López (2001) estimated capacity loss of 14 major reservoirs in Puerto Rico through sedimentation surveys. These values are reported below for the seven surveyed reservoirs that drain basins in the study area. Primary uses of these reservoirs were obtained from National Atlas of the United States (2006) and are designated as follows: H: Hydroelectric, I: Irrigation, R: Recreation, S: water supply. The sediment delivery ratio was estimated based on basin size using values interpolated from values found in Boyce (1975)

Reservoir	Caonillas	Dos Bocas	Garzas	Guayo	La Plata	Loíza	Yahuecas	Average	Median
County	Utuado	Arecibo	Adjuntas	Adjuntas	Tao Alta	San Juan	Adjuntas	-	-
Drainage area, in km^2	126.65	310	15.6	24.86	469	538	45.17	218.47	126.65
Primary Uses	HR	HS	HS	HI	S	HS	HIS	-	-
Original capacity, in Mm^3	55.66	37.5	5.8	19.2	40.21	26.81	1.76	26.71	26.81
Const.year	1948	1942	1943	1956	1974	1953	1956	-	-
Age	52	57	53	41	24	41	41	44	41
Storage capacity, in Mm^3	42.27	18.04	5.11	16.57	35.46	14.2	0.33	18.85	16.57
Totalvol. loss,Mm^3	13.39	19.46	0.69	2.63	4.75	12.61	1.43	7.85	4.75
Loss in pecent	24	52	12	14	12	47	81	34	24
Long-term storage loss per year, in percent	0.5	0.9	0.2	0.3	0.5	1.1	2	0.79	0.5
Sediment yield, in $m^3/km^2/yr$	2,186	1,299	878	2,660	483	750	1,430	1,383	1,299
Storage loss, in $m^3/km^2/yr$	2,033	1,103	834	2,580	422	572	772	1188	834
Trapping Efficiency	0.93	0.85	0.95	0.97	0.87	0.76	0.54	0.84	0.87
Estimated sediment delivery ratio	0.12	0.10	0.21	0.18	0.08	0.08	0.17	0.13	0.12

Several policy options are available for dealing with water supply issues. One option is to increase water storage in the reservoirs lost to sedimentation by dredging. An analysis of the conditions of the principal reservoirs in Puerto Rico recommends immediate dredging of several major reservoirs, as well as initiation of permanent dredging programs (Zayas et al., 2004). Unfortunately, dredging is a necessary but short-term solution to the capacity loss problem. The Carraízo reservoir was dredged in 1997-1998 at a cost of $60 million. However, in part due to subsequent hurricanes, it has already lost a substantial part of the recovered capacity (Soler-López and Gómez-Gómez, 2005). While not all of the reservoirs require immediate dredging, a significant number of critical reservoirs are rapidly losing capacity. This is of particular concern for the future because it has been noted that the sites most appropriate for reservoir utilization are already in use (DRNA, 2005). A second recommended policy is to create watershed protection programs in reservoir basins (Zayas et al., 2004). This is a long-term policy option that has implications for both the long term quality and quantity of water supplied.

The recognition of the impact of land use on water quality has led to an interest in preserving forests. Evidence of increasing concern in Puerto Rico over protecting the water supply benefits provided by forested land can be found in a bill introduced to the U.S. House of Representative in April 2005 which calls for the acquisition of land in the Karst region in order to protect the water supply (H.R. 1644). While this program is specifically targeted to improve the water supply, other programs are in place that attempt to provide incentives to preserve forests in general, for functional as well as cultural and aesthetic reasons.

In Puerto Rico there are a variety of programs that provide incentives for farmers to preserve forested areas. The Ley de Bosques de Puerto Rico provides forested land in excess of 5 cuerdas (~2 ha) adjacent to agricultural land are exempt from property taxes and that forest products that come from lands classified as forestry lands are also exempt from taxation (Ley de Bosques de Puerto Rico, 2000). In the areas surrounding the Luquillo Experiemental forest, the value of this tax break is $80/ha/yr based on reported land values (Odum et al., 2000). The Ley para Unificación de los Bosques Estatales de Maricao, Susúa, Guánica, Toro Negro, Guilarte y Pueblo de Adjuntos calls for the design of incentives for landowners whose land may be included the relevant biological corridors (Ley para Unificación de los Bosques Estatales de Maricao, Susúa, Guánica, Toro Negro, Guilarte y Pueblo de Adjuntos, 1999). In addition, the Forest Enhancement Program (FLEP) authorized in the Farm Security and Rural Investment Act of 2002 provides technical and educational assistance to farmers. This program is intended "to promote sustainable forest management on non-industrial private forestland and to complement other sustainable forestry programs in the states" (DRNA, 2003). The amount provided to farmers under such programs varies. In 2005, individual farms in Puerto Rico received between $550 and $6800 per hectare under various programs (Congress, 2005; Manejo Comunitario Simposio, 2006).

3. Methods

This study consisted of several parts. First, data and information were collected from published sources and interviews to determine the private and public economic benefits of alternate land uses in Puerto Rico. Second, results of the extensive background and literature review were used to determine the most important trade-offs for consideration of policy-makers. This was found to be the trade-off between forest and pasture in terms of hydrologic ecosystem services. Third, a model was created to determine at what topographic slopes the hydrologic benefits of forests provided by increased runoff exceed the economic costs of increased erosion and sedimentation. The model was then modified to include the benefits of other ecosystem services, primarily carbon sequestration. Finally, private economic returns were compared to public costs and regional guidelines were developed for assessing land-use policies in the region.

3.1 Data Collection

In order to determine the economic trade-offs inherent in converting Puerto Rican forest to pasture, data and information were collected from economic and scientific literature, government documents, personal observations and interviews. Data collection focused on information specific to the subtropical wet forest lifezone found in the central mountain region of Puerto Rico. Information from other similar areas is included to provide a basis for comparison when study-area specific data was available and was used for value estimation when it was not. As applicable, values from past studies were converted to 2005 U.S. dollars using CPI conversion factors from the Bureau of Labor Statistics.

First, private returns to alternate land uses in the study area were estimated. Land uses considered were forest, agriculture, and pasture. Since conversion of forest to urban use in Puerto Rico generally occurs in flat areas and those near urban areas (Helmer, 2004), urban use is not considered as an alternate land use in this study. The private returns from forests that were considered were non-timber forest products (NTFPs) and timber production. NTFP's include products such as medicinal plants, dye stuff, resins, edible fruits. Possible returns from timber harvesting were based on published estimates of timber production in Puerto Rico (Odum et al., 2000) and published stumpage prices (Sedjo, 1999).

Estimate of the economic benefits of alternate land uses were obtained for the most common types of production in the area, including shade coffee and non-dairy cattle ranching. Revenues were estimated using data on market values and farm size from the USDA 2002 Census of Agriculture. When only revenue data was available a profit-to-revenue ratio of 0.3 was assumed. This ratio was based on a detailed study of dairy farming in Puerto Rico (Caro-Costas and Vicente-Chandler, 1974). To determine a range of potential values, estimates were also obtained from the literature (Perfecto et al., 1996) and personal communication with farmers. Estimated tax exemptions provided under the current tax policy were based on current tax laws and recent land purchases. As discussed below, this analysis indicated that the land use with the highest private return was pasture. Therefore the remainder of the study focused quantifying the public trade-offs between forest and pasture, including carbon storage, air quality, biodiversity, recreation, existence value, and water quality and quantity.

Values of carbon sequestered in forests were determined using published values of carbon sequestration in Tabonuco forests in Puerto Rico (Silver et al., 2004). As in other areas of the tropics, the difference between soil carbon sequestration in forests and pastures in Puerto Rico has been found to be negligible (Post and Kwon 2000; Murty et al., 2002; Lugo et al., 1986). Therefore, the difference in carbon sequestration between pastures and forests was evaluated as the amount of carbon sequestered aboveground in forest. Monetary values of the sequestration were estimated using market rates for carbon credits.

The benefits of forests to air quality were based on the American Forests CITYGREEN model study of the San Juan Metropolitan area (American Forests 2002). The values of biodiversity were estimated using published values in the literature on tropical forests (EEP, 2003; Pearce, 2001; Barbier and Aylward, 1996) and reports of government spending on conservation in Puerto Rico (Cruz, 2006; USFWS, 2004).

Values of recreation were obtained from literature on recreation benefits in tropical forests in general, data from spending on ecotourism in the Caribbean National Forest of Puerto Rico, and data collected in surveys by Hernández and Sánches (1986) and the NSF Biocomplexity Project on recreation at lakes and rivers in northeastern Puerto Rico. Recreation value of the land in the Caribbean National Forest to local people was estimated using data from the Biocomplexity Project. The average number of visitors per hour was estimated based on visitors observed on an hourly basis, and this was multiplied by 8 hours to determine the average number of visitors per day. The number per year was estimated by multiplying the average number of visitors per day by 90, based on the fact that most local people visit the park during the summer (USDA Forest Service, 2007). The total value for each site was calculated using the calculated average travel cost of $8/person and the average additional willingness to pay per person (NSF Biocomplexity Project; Scatena, 1994). Average additional willingness to pay per person of $30 was estimated based on survey responses to the question of whether the individual would be willing to pay an additional stated amount for the trip. Per hectare values were calculated by dividing the total value of sites by the number of hectares. Published existence values from other areas of the tropics were reported (Pearce, 2001).

The difference in the water quality benefits of water from forests and pasture was based on the off-site cost of sediment derived from each landuse. Estimates of the mass of sediment that reach a particular reservoir can be obtained using the Revised Universal Soil Loss Equation (RUSLE) and the sediment delivery ratio of the basin (SDR_b). Values for use in RULSE and the SDR_b were obtained from a study of a watershed in central Puerto Rico that has a climate and land use that is characteristic of the study area (Lopez et al., 1998). Since mass wasting is not accounted for by the RUSLE, data on the frequency and mass of landslides on forests and pastures in Puerto Rico was also collected to determine the expected mass of additional sediment from landslides on a yearly basis (Larsen and Torres-Sánchez, 1998; Larsen and Parks, 1997). The expected value of the additional sediment mass from landslides that enters the reservoir was calculated using a sediment delivery ratio (SDR_l) of 0.53. This ratio was determined based on the frequency of landslides by topographic location (Scatena and Lugo, 1995).

Costs of sedimentation included dredging, capacity loss, and water treatment. Additional costs of damage to ecological integrity or recreational benefits may be important but are not assessed in this study. In order for dredging costs to be an appropriate valuation tool, there must be a very strong reason to believe that dredging programs will be consistently carried out. This is the case in Puerto Rico, as dredging programs have been recommended as the least-cost solution to the problem of capacity loss in Puerto Rico (Vega and Terrasa-Soler, 1998) and the Commonwealth has recommended dredging programs as a primary means of dealing with this problem

(Zayas et al., 2004). Dredging costs were obtained from PRASA and compared to published values, while other costs were estimated using values in the literature. Water treatment cost was estimated using the cost/NTU data found in Dearmont et al. (1999) and a regression using USGS water quality data from Puerto Rico relating NTU to suspended sediment (USGS, 2007).

To evaluate the difference in water quantity benefits, data on runoff values for pasture and forest specific to the study area were obtained from the literature. The annual per hectare water use of cattle was calculated based on the daily intake of water by cattle Lardy and Stoltenow (1999), and the average number of cattle per hectare (Personal communication: Interviews with farmers, Adjuntas 2006) The value of water was estimated using prices from purchases of water for municipal and industrial purposes in the United States and consumer water price data from PRASA.

3.2 Model

A mathematical model was created to determine the minimum topographic slope factor at which the conversion of one hectare of forest to pasture has a negative effect on the provision of public hydrologic ecosystem services. This occurs when the increased cost due to sedimentation becomes greater than the increased benefit from increased water supply. For a list of variables used in the model see Tables 3.2.2 and 3.2.3.

The output of the model is the RUSLE topographic factor LS at which the costs of forest and pasture are equal. According to the RUSLE, the amount of erosion depends both on the slope length and the slope steepness. In the RUSLE, LS is the slope length and steepness factor, or topographic factor. Table 3.2.1 presents values for LS for difference slope and length combinations for rangeland and other consolidated soil conditions (Renard et al., 1997). From the output of the model, Table 3.2.1 can be used to determine at what combinations of slope lengths and slope steepness the solution of the model is exceeded. The output is also expressed as a slope in degrees by assuming a slope length of 72.6 ft, the RULSE unit plot length (Renard et al, 1997).

The model describes a hypothetical watershed in which all surface runoff travels to a reservoir, is treated in a water treatment plant, and is then consumed by households (Figure 3.2.1). Annual runoff volumes per hectare are constant and depend only on land use. The additional water available for human consumption from pasture is the difference in runoff between forests and pastures minus the amount of water consumed by grazing cattle. The value of the runoff is determined using an estimate of the average value of water. The amount of sediment produced by a hectare of land depends both on land use and the slope. The total amount of sediment produced at the site of the hectare is determined using RUSLE and landslide incidence and mass data (Larsen and Torres-Sánchez, 1998; Larsen and Parks, 1997). The amount of sediment that reaches the reservoir is reduced by the sediment delivery ratio determined in an independent study (Lopez, 1998). Of the sediment that reaches the reservoir, some is trapped in the reservoir and the rest reaches the water treatment plant. The amount that remains in the reservoir is determined by the trapping efficiency of the reservoir. This sediment must be dredged on an annual basis at the cost reported by PRASA. The sediment that reaches the water treatment plant adds to the cost of water treatment.

Table 3.2.1 Values for topographic factor in Revised Universal Soil Loss Equation
LS topographic factor corresponding to various combinations of slope (%) and horizontal slope length for rangeland and other consolidated soil conditions with cover, reported in Renard et al., 1997.

	Horizontal slope length (ft)																
Slope (%)	<3	6	9	12	15	25	50	75	100	150	200	250	300	400	600	800	1000
0.2	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05	0.05
0.5	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.08	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.09	0.09
1.0	0.12	0.12	0.12	0.12	0.12	0.13	0.13	0.14	0.14	0.15	0.15	0.15	0.15	0.16	0.16	0.17	0.17
2.0	0.20	0.20	0.20	0.20	0.20	0.21	0.23	0.25	0.26	0.27	0.28	0.29	0.30	0.31	0.33	0.34	0.35
3.0	0.26	0.26	0.26	0.26	0.26	0.29	0.33	0.36	0.38	0.40	0.43	0.44	0.46	0.48	0.52	0.55	0.57
4.0	0.33	0.33	0.33	0.33	0.33	0.36	0.43	0.46	0.50	0.54	0.58	0.61	0.63	0.67	0.74	0.78	0.82
5.0	0.38	0.38	0.38	0.38	0.38	0.44	0.52	0.57	0.62	0.68	0.73	0.78	0.81	0.87	0.97	1.04	1.10
6.0	0.44	0.44	0.44	0.44	0.44	0.50	0.61	0.68	0.74	0.83	0.90	0.95	1.00	1.08	1.21	1.31	1.40
8.0	0.54	0.54	0.54	0.54	0.54	0.64	0.79	0.90	0.99	1.12	1.23	1.32	1.40	1.53	1.74	1.91	2.05
10.0	0.60	0.63	0.65	0.66	0.68	0.81	1.03	1.19	1.31	1.51	1.67	1.80	1.92	2.13	2.45	2.71	2.93
12.0	0.61	0.70	0.75	0.80	0.83	1.01	1.31	1.52	1.69	1.97	2.20	2.39	2.56	2.85	3.32	3.70	4.02
14.0	0.63	0.76	0.85	0.92	0.98	1.20	1.58	1.85	2.08	2.44	2.73	2.99	3.21	3.60	4.23	4.74	5.18
16.0	0.65	0.82	0.94	1.04	1.12	1.38	1.85	2.18	2.46	2.91	3.28	3.60	3.88	4.37	5.17	5.82	6.39
20.0	0.68	0.93	1.11	1.26	1.39	1.74	2.37	2.84	3.22	3.85	4.38	4.83	5.24	5.95	7.13	8.10	8.94
25.0	0.73	1.05	1.30	1.51	1.70	2.17	3.00	3.63	4.16	5.03	5.76	6.39	6.96	7.97	9.65	11.04	12.26
30.0	0.77	1.16	1.48	1.75	2.00	2.57	3.60	4.40	5.06	6.18	7.11	7.94	8.68	9.99	12.19	14.04	15.66
40.0	0.85	1.36	1.79	2.17	2.53	3.30	4.73	5.84	6.78	8.37	9.71	10.91	11.99	13.92	17.19	19.96	22.41
50.0	0.91	1.52	2.06	2.54	3.00	3.95	5.74	7.14	8.33	10.37	12.11	13.65	15.06	17.59	21.88	25.55	28.82
60.0	0.97	1.67	2.29	2.86	3.41	4.52	6.63	8.29	9.72	12.16	14.26	16.13	17.84	20.92	26.17	30.68	34.71

The total public cost of one hectare of pasture was initially modeled as the total cost of sediment as a function of slope minus the benefit of the increased runoff. The total public cost of one hectare of forest is the total cost of sediments as a function of slope. The model can be extended to include other ecosystem services as well as private benefits. In this case, the total cost of one hectare of forest is the total cost of sediments as a function of slope minus the difference in benefits from other ecosystem services. In the following section, the subscript x is used in general equations to represent land cover type, either p (pasture) or f (forest). Descriptions of all cost variables and benefits variables used in the model are found in Table 3.2.2 and Table 3.2.3, respectively.

In general, the model is described as follows:

For each land use, Total Cost = Total Cost – Total Benefits, or:

$$TC_x = C_x - B_x \tag{1}$$

When only hydrologic services are considered, (1) is equivalent to:

Total Hydrologic Public Cost = Cost of Sediment – Benefit of Additional Runoff

The cost of sediment is a function of the total amount of sediment that reaches the reservoir, which is a function of slope. The total amount of sediment coming off one hectare of land includes the mass of sediment moved by erosion processes, as described by the RUSLE equation, and sediment removed by landslides. The amount of this sediment that actually reaches the reservoir is calculated using a basin-wide sediment delivery ratio and a sediment delivery ratio for landslides (2). The cost is calculated using the trapping efficiency and the dredging and water treatment costs (3).

$$S_x = (SDR_b*R*K*C_x*P*c*)LS + SDR_l*M_l*\rho_x(LS) \quad (2)$$

$$C_x = S_x*TE*\$D + S_x(1\text{-}TE)*\$WT \quad (3)$$

In the hydrologic model, benefits include only the value of the additional runoff from pasture (4). In the full model, costs are determined as above and benefits is extended to include other ecosystem services and private benefits (5). For simplicity, a utilitarian social welfare function is assumed, meaning that public and private benefits are given the same weight.

$$B_x(\Delta RO_x) = \$RO*\Delta RO_x \quad (4)$$

$$B_x(\Delta RO_x, O_x, P_x) = B_x(\Delta RO_x) + B_x(O_x) + B_x(P_x) \quad (5)$$

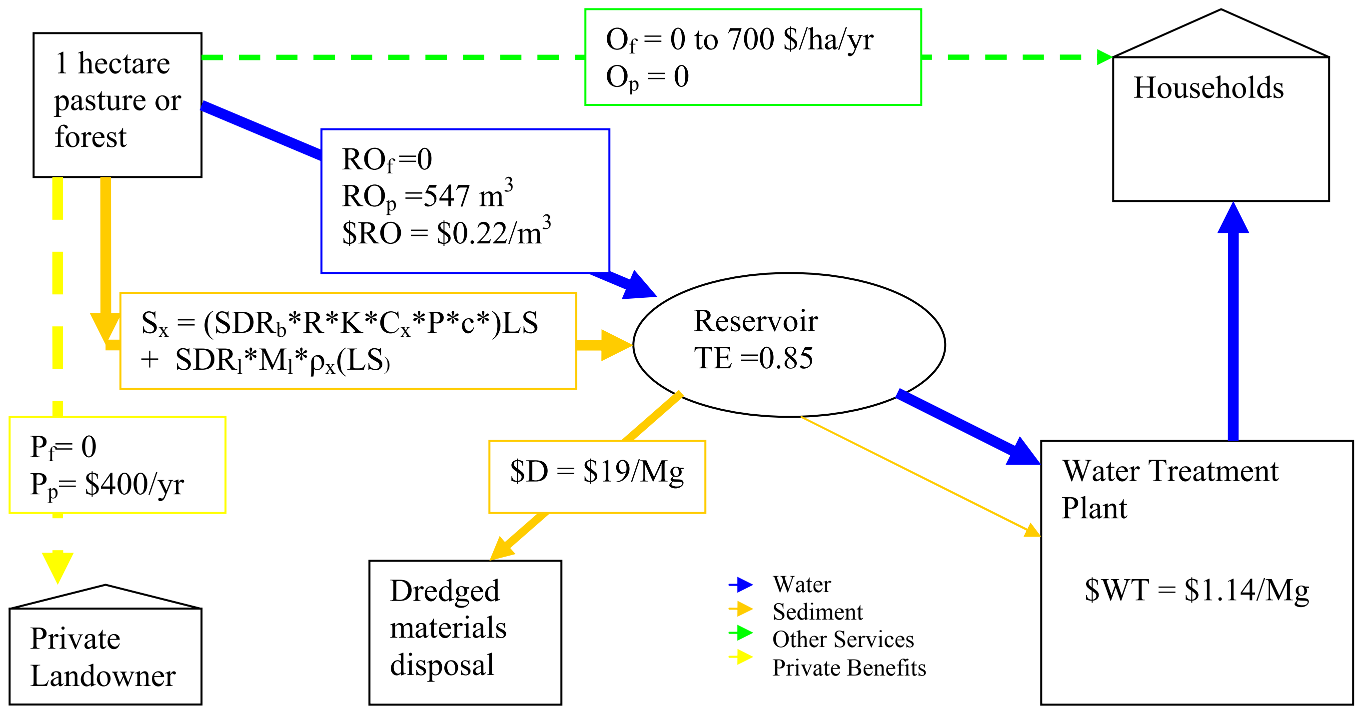
Table 3.2.2. Description of cost variables

Variable	Description	Units	Value	Source
x	Subscript denotes land cover (f for forest, p for pasture)	-	F or P	-
S_x	Mass of sediment that reaches reservoir	Mg/ha/yr	$(SDR_b*R*K*C_x*P*c)*LS + SDR_l*M_l*\rho_x(LS)$	-
$C_x(S_x)$	Cost of sediment	$/ha/yr	$C_x = S_x*TE*\$D + S_x(1-TE)*\$WT$	-
LS	Slope length factor/slope steepness factor	dimensionless	Independent Variable	-
SDR_b	Sediment Delivery Ratio, basin	ratio	0.17	SDR for Guadiana Watershed, Puerto Rico (López et al., 1998)
SDR_l	Sediment Delivery Ratio, landslides	ratio	0.53	Calculated based on the percentage of slides near stream channels (Scatena and Lugo, 1995)
R	Rainfall factor	tons/acre/yr	415	Average for Puerto Rico (López et al., 1998)
K	Soil erodibility factor	tons/acre/yr	0.17	K factor for Pellejas type Soil Series (López et al., 1998)
C_F	Cover and management factor forest	dimensionless	0.014	C factor for closed canopy forest (López et al., 1998)
C_P	Cover and management factor pasture	dimensionless	0.023	C factor for pasture (López et al., 1998)
P	Erosion control factor	dimensionless	1	Assumed (López et al., 1998)
c	Conversion factor to convert from English tons/acre/yr to Mg/ha/yr	Mg*ha/ton*acre	2.5105	-
M_l	Average mass of a landslide	Mg/landslide	1620	Calculated from Larsen and Parks (1997)
$\rho_x(LS)$	Probability of a landslide in any given year	# landslides/ha/yr	0.00080 x = f, LS>=4 0.00090, x = f, LS<4 0.00403, x = p, LS>=4 0.00183, x = p, LS < 4	Calculated from Larsen and Torres-Sànchez, 1998
TE	Trapping Efficiency	ratio	0.84	Average Trapping Efficiency, Calculated from Soler-López (2001)
$D	Unit Cost of Dredging	$/Mg	19.14	$13.4/m^3 dredging cost (PRASA) multiplied by bulk density 0.7 g/cm^3 (Richard Webb, electronic communication, May 3, 2006)
$WT	Unit Cost of Water Treatment	$/Mg	1.14	Extrapolated from Dearmont (1998)

Table 3.2.3. Description of benefit variables

Variable	Description	Units	Value	Source
B_x	Benefits from 1 ha land use x	$/ha/yr	$B_x(RO_x) + B_x(O_x) + B_x(P_x)$	-
$RO	Value of water	$/m^3	0.22	Average value of water for domestic use in U.S. (Frederick,1996)
RO_f	Runoff forest	m^3/ha/yr	28,132	Annual runoff tabonuco forest (van der Molen, 2002)
RO_p	Runoff pasture	m^3/ha/yr	28,679	Annual runoff Fajardo grassland (van der Molen, 2002)
W_c	Water used by cattle	m^3/ha/yr	80	Annual water intake of 5 average cattle (Lardy and Stoltenow, 1999)
ΔRO_f	Additional runoff forest	m^3/ha/yr	RO_f - RO_f	-
ΔRO_p	Additional runoff pasture	m^3/ha/yr	$RO_p - W_c - RO_f$	-
O_f	Other Ecosystem Services forest	$/ha/yr	36	Maximum benefits from carbon. See Table 17 for other possible values
O_p	Other Ecosystem Services pasture	$/ha/yr	0	-
P_f	Private benefits forest	$/ha/yr	0	See Table 6.
P_p	Private benefits pasture	$/ha/yr	400	Minimum benefits from cattle ranching. See Table 6 for other possible values.

Figure 3.2.1. Model schematic



4. Results

4.1 Private and Public Benefits of Forest and Pasture in the Subtropical Wet Forest Lifezone In Puerto Rico

<u>Private Economic Benefits</u>

Forest: Timber and Non-Timber Forest Products (NTFPs)

In some areas in the tropics, timber production can provide significant economic benefits. In Puerto Rico plantations of commercially valuable trees has been considered as a means of reforestation (Odum et al., 2000). While some forms of timber production such as high rotation clear-cutting are unsustainable, other ways of producing timber can preserve forests and maintain ecosystem service functioning. Two commercially valuable trees that have been grown successfully in Puerto Rico are mahogany (Odum et al., 2000) and teak (*Tectona grandis*) (Devall and Parresol, 2003). However, the estimated returns to a 60-year mahogany plantation are negative (Table 4.4.1). This is partly due to the high costs. The costs reported here reflect the cost of a reforestation project. Since the goal of this project was to achieve a closed canopy, not to maximize profits from timber production, these costs may be higher than those for a commercial timber project. In general, costs of sustainable timber operations are higher than those of traditional methods, and generate lower financial returns (EEP, 2003; Pearce, 2001). This suggests that if timber plantations are to be used as a means of achieving forest conservation in Puerto Rico, they are unlikely to provide private benefits to landowners in the absence of financial incentives from the government.

Table 4.1.1 Estimated net present value of a mahogany plantation

Parameter	Value	Source
Cost to achieve closed canopy ($/ha)	$9,712	(Odum et al., 2000)
NPV of cost divided over 60 years	$3,662	Calculated
Estimated total yield for 60 year cycle (m3/ha)	152.4	(Odum et al., 2000)
First Harvest at 30 years (m3/ha)	76.2	Assumed
2nd Harvest at 60 years (m3/ha)	76.2	Assumed
Stumpage price ($/m3)	$15	(Sedjo, 1999)
Discount Rate (%)	4%	Assumed
PV First Harvest ($/ha)	$352.40	Calculated
PV Second Harvest ($/ha)	$108.70	Calculated
PV Total Revenues ($/ha)	$461.10	Calculated
NPV ($/ha)	($3,200.90)	Calculated
Average per year ($/ha/yr)	($53.35)	Calculated

Currently only one or two small mills are located on the island and no active timber industry in Puerto Rico (Fred Scatena, personal communication, March 2007). This is another indication that returns to timber are small compared to other land uses. In addition, investment in timber plantations in Puerto Rico is particularly risky due to the frequency of hurricanes.

In other areas of the tropics, published returns to timber range from $20 to $5,000 per hectare per year. However, some methods of timber harvesting are unsustainable and impair ecosystem service functioning. These methods generally have higher financial returns in the short run. Returns to sustainable logging practices in the tropics have been found to range from approximately $30-290/ha/yr. For a summary of values of forests used for timber production in various tropical countries, see Table 4.1.2.

While mahogany plantations have been successful in the subtropical wet forests of the lower mountain region, few farmers in Puerto Rico willingly choose to convert their land to timber plantations. However, in the case of reforestation by plantations, commercially viable timber is a side benefit of reforestation schemes and may contribute to the economic feasibility of such projects. While some ecosystem services such as biodiversity preservation may be affected by plantations, it has been suggested that commercial forestry has the potential to sequester substantial volumes of carbon (Sedjo and Sohngen, 2000) and many of the hydrologic functions are likely to remain intact.

Non-timber forest products include edible plants and animals and other useful non-timber derived from forest lands. Estimates of the value of NTFPs from tropical forests in developing countries range from \$7 to \$357 per hectare per year (Chopra, 1993; Godoy et al. 2002). However, non-timber benefits are rarely considered in valuation studies in developed countries because very few people in developed countries use NTFP's directly (EEP, 2003). Because of the urban and industrialized nature of Puerto Rico, most NTFP's have little or no value in Puerto Rico. Nevertheless, informal observations and conversations suggest that it is common for residents in central Puerto Rico to collect water from streams and springs in the mountains to use for drinking water. Some individuals collect enough to provide approximately 1 gallon per day of drinking water. However, since these benefits do not accrue to the landowner, they are not considered a private benefit.

Table 4.1.2 Net present value of forests used for timber production in 2005 U.S. dollars

Source	Description	Location	Discount factor	Annualized NPV 2005$/ha/yr
(Almeida and Uhl, 1995)	Extensive timber over 90 years Intensive timber over 30 years	Brazil	6%	45 -46
(Browder, 1988)	Foregone timber values	Brazil	n/a	843
(Veríssimo, Barreto et al., 1992)	Average price of logging concession Extraction done by rancher	Brazil	n/a	94 287
(Peters, Gentry et al., 1989)	Clear-cut timber harvesting Plantation harvesting timber and pulpwood	Peru	5%	1,576 5,014
(Sedjo, 1988)	Saw timber plantations Fast-growing pulpwood plantations	Indonesia	6%	3,573 4,828
(Paris and Ruzicka, 1991)	Financial profit to old growth logging Including environmental damage	Philippines	n/a	171 -1,323 to -100
(Sedjo and Bowes, 1991)	Immediate exploitation of 60-year mahogany stand Immediate exploitation of 110-year mahogany stand	Washington, USA	n/a	1,651 3,637
(Pearce, 2001)	Annualized returns to conventional logging Annualized returns to sustainable logging	Tropical countries	10%	22 to 485 33 to 293
Author's Calculations	Estimated annualized returns to mahogany plantation	Puerto Rico	4%	-53

Agriculture: Coffee

Returns to agriculture depend highly on the type of agriculture practiced, site-specific conditions, and market conditions. The mountainous regions of Puerto Rico have been described as inappropriate for most types of agriculture (Ewel and Whitmore, 1973). However, coffee is successfully produced in the area. Coffee can be grown either in the sun or in the shade. Sun coffee generally produces a higher yield, but shade coffee can produce higher quality and can maintain many of the ecosystem services of forests (Perfecto et al., 1996). Traditional shade coffee has lower production costs, and Perfecto et al. (1996) report that in 1994 shade coffee had a net revenue of approximately $350/ha/yr, higher than the net revenue of non-shade coffee. With a profit-to-revenue ratio of 0.3, this translates to a return of approximately $140/ha/yr. Using the USDA data, returns to shade coffee in Puerto Rico were estimated at $210/ha/yr. While estimated private returns per hectare to coffee are lower than returns to ranching, shade coffee has the benefit of preserving valuable ecosystem services, most notably biodiversity (Wunderle and Latta, 1996) and erosion control.

Pasture: Cattle Ranching

Like returns to agriculture, returns to ranching depend on the type of ranching practices, site-specific conditions, and market conditions, and estimates of the NPV of ranching vary between studies and between countries. In other areas, estimated private returns to ranching have ranged from negative $258/ha/yr for extensive ranching in Brazil, to $1053/ha/yr for large scale ranching in Costa Rica (Almeida and Uhl 1995;, Aylward and Echeverría, 2001). In Puerto Rico, the estimated returns to cattle ranching range from

$400/ha/yr (Personal communication: Interviews with farmers in Adjuntas, 2006) to $500/ha/yar (Calculations based on data from USDA Census of Agriculture, 2002).

Additional Private Benefits – Land Prices and Taxes

Under the current Puerto Rico Forestry Law, landholders receive tax benefits for holding forested land. This may be seen as an additional private benefit to forests. However, it is also a public cost. The value of the land determines the value of the tax incentive provided by Puerto Rico's Forestry Law. If a farmer already owns at least 5 cuerdas of forested land, the Commonwealth Forestry Law provides for the exemption of taxes on forested land. Property taxes amount to $28/$1000 assessed land value (Odum et al., 2000). Thus the value of such a tax exemption would ranges from $30-105 per hectare per year.

Table 4.1.3. Tax benefits from Puerto Rico Forestry Law based on the reported value of land in various areas of Puerto Rico.

Land Location	Source	Reported Value	Tax Benefit
Karst belt	L. Jorge, electronic communication, 2006	$1,040	$29
Central Mountains	R. Salguero, personal communication, 2006	$1,555	$43
Karst belt	L. Jorge, electronic communication, 2006	$2,829	$79
Luquillo, adjacent to the Caribbean National Forest	Odum et al., 2000	$3,753	$105

Summary

The land use with the current highest private value is pasture (Table 4.1.4). This is supported by observed land use patterns in the past several decades (Helmer, 2002). Estimated returns to pasture exceed private returns to forest, even assuming the highest reasonable value of returns from timber and the highest land value for taxes. These estimates also indicate that on some land the private returns to forest from timber production would be negative even including tax benefits.

Table 4.1.4. Estimated private benefits to alternate land uses in subtropical wet forest lifezone in Puerto Rico

Land Use	Annual returns 2005$/ha/yr	Source
Forest – Timber	-50 to 290	Author's calculations for Mahagony plantation, Pearce, 2001, NPV of sustainable logging of tropical forests, annuitized at 10%
Forest - Tax Benefit	30 to 105	Likely value of Forestry Law tax exemption. Land value reported by DNER (Luis Jorge, electronic communication, March 11, 2006) and tax assessment reported in (Odum et al., 2000)
Forest - Timber + Current Tax Benefit	-20 to 395	Sum of returns to timber and tax benefits
Agriculture – Coffee	140 to 210	Estimated from revenue reported in Perfecto et al. (1996) Estimated from market values and production reported by USDA
Agriculture – Pastures: Cattle Ranching	400 to 500	Interviews, estimated from market values and production reported by USDA

Non-Hydrologic Public Benefits of Forests and Pasture in Puerto Rico

Carbon

Carbon sequestration is a global service that often dominates the contribution of other services to the non-market values estimated in forest valuation studies (EEP, 2003). Most valuation studies focus on carbon storage rather than carbon sequestration. Carbon storage is a stock value, while sequestration is a flow. Generally carbon storage is estimated as a percentage of biomass. Estimates of the value of carbon vary greatly in the literature, depending on which monetary values are used. Previous studies have reported values ranging from 30 to 160 tonnes of carbon stored per hectare for different types of forest. NPVs of the storage are estimated to range from $650 to $3400 per hectare (Adger et al., 1995). Another study of forests in Costa Rica estimates 74 to 238 tons stored per hectare resulting in estimated NPVs of only $197 to $300 per hectare (Aylward and Echeverría, 2001). The usefulness of these numbers in determining optimal land use is questionable because they do not indicate the amount of carbon that would be stored under alternate land use scenarios. Ideally, what is being valued is the amount of carbon that would be released into the atmosphere if the forest were converted to an alternate land use. In addition, valuing carbon storage may not be appropriate when payments for ecosystem services are made on a yearly basis. In this study, valuing carbon sequestration is more appropriate than valuing carbon storage because carbon sequestration continues over time.

In Puerto Rico, the average 80-year aboveground accumulation of moist forest following reforestation of a pasture has been estimated at 1.4 Mg/ha/yr, and the belowground accumulation at 0.5 Mg/ha/yr (Silver et al., 2004). This rate is lower than rates reported for young secondary forests in Puerto Rico but higher than those reported for old-growth forests. Over time the benefits of forest in terms of sequestration tend to decrease because the highest rates of sequestration tend to occur in the first 20 years of forest re-growth. The literature also suggests that the difference in soil carbon sequestered by the pasture and forest in Puerto Rico are negligible (Lugo et al., 1986). Therefore, using a time horizon of 80 years, the net benefit of keeping land in forest rather than pasture is assumed to be the value of the aboveground carbon sequestered by the forest, or 1.4 Mg/ha/yr. Using the commonly-cited value of stored carbon of $20/tonne (Fankhauser, 1995; Adger et al., 1995; Pearce, 1996), this amounts to

$28/ha/yr. Using the 2005 market values of CO_2 equivalent (Lecocq and Capoor, 2005), the value of carbon sequestration could range from $8.72 to $36.40/ha/yr.

Air Quality

According to calculations of the American Forests CITYGREEN model, the 26,229 hectares of urban tree cover in the San Juan Metropolitan area remove 9.5 million pounds of contaminants each year from the air. This model estimates the value of this removal at $22.6 million/yr, or $88/ha/yr. Fortunately, because the tradewinds regularly refresh the island with clean Atlantic air from the east, air quality in Puerto Rico has never been considered a significant problem. Therefore the value of forests in reducing air pollution may be smaller on the island that is considered by the CITYGREEN model and potentially as low as zero.

Biodiversity

Puerto Rico is part of "one of the world's centers of biodiversity and endemism" (Helmer et al., 2002). The effects of past land use on forest biodiversity have been extensively studied in Puerto Rico (Marcano-Vega et al., 2002; Grau et al., 2003; Lugo and Helmer, 2004). Although biodiversity is recognized as important in the scientific literature, no attempts have been made to quantify the economic significance of the differences in biodiversity between different land types in Puerto Rico. However, a general estimate is obtained from values from the literature.

Most studies that attempt to value biodiversity have estimated the value of potential pharmaceutical options. These values are listed in Table 4.1.6. While biodiversity is valuable for reasons other than pharmaceutical prospecting, very few studies have investigated diversity-ecosystem function values in a complete way, and none have considered multiple services or a systems approach (Kremen and Ostfeld, 2005). Valuable services that have been shown to decline with decreased diversity include crop pollination by wild bees and dilution of Lyme disease risk by vertebrates (Kremen and Ostfeld, 2005).

One indication that biodiversity conservation is of concern to the international community is the willingness of governments and other organizations to pay for conservation programs. A survey of funding for conservation investments found that between 1990 and 1997, 3,489 conservation projects worth a total of $3.26 billion were funded in the Latin American and Caribbean region (Castro et al., 2000). The United States Fish and Wildlife Service (USFWS) spent over $1.1 million on 7 endangered species in Puerto Rico in 2004 (USFWS, 2004). The largest expenditure on a single species, $1,573,500, was spent on the Puerto Rican parrot. This parrot has a range of only 1600 hectares, all of which is within the Caribbean National Forest (IUCN, 2006). The average expenditure of the USFWS on protecting the Puerto Rican parrot alone was therefore $983/ha in 2004 (Table 4.1.5).

Expenditures on endangered species indicate that society places a high value on select areas of forest that provide habitat for certain endemic species in Puerto Rico, much of which is contained in the Caribbean National Forest and other protected areas. However, they provide little insight into the economic value of preserving biodiversity in general and in other parts of Puerto Rico. Unfortunately, the willingness of the U.S. government to pay for biodiversity conservation does not seem to correspond with level of threat or taxonomic distinctions (Restani and Marzluff, 2001). Nor is it based on economic considerations. The willingness of governments to pay for biodiversity preservation in other parts of the world as indicated by current expenditures on conservation may be seen as an indication that biodiversity preservation has value, but is fairly unhelpful in determining the true economic value of biodiversity preservation.

Table 4.1.5. U.S. government spending on endangered species in Puerto Rico, 2004. Values reported by the United States Fish and Wildlife Service (2004).

Species	U.S. Government Spending, $
Puerto Rican broad winged hawk	20,000
Puerto Rican sharp-shinned hawk	4,000
Puerto Rican nightjar	35,000
Puerto Rican parrot	1,573,500
Puerto Rican pigeon	45,500
Puerto Rican boa	153,073
Puerto Rican crested toad	40,000
Total all species	1,871,073
Average spending per hectare, Puerto Rican Parrot	983

Some government programs in the tropics specifically target biodiversity by encouraging farmers to adopt methods of agriculture that preserve biodiversity. Payments made under these programs are one indication of the willingness to pay of governments for biodiversity preservation. Costa Rica has developed an elaborate Payment for Ecosystem Services program under its 1997 Forestry Law. Under this program, payments for biodiversity due to conversion from degraded pasture to secondary forest are worth $67/ha/yr (Pagiola et al., 2004). This is the value considered most relevant to this study, and is still considered an upper bound.

In Puerto Rico, there has recently been interest in encouraging shade coffee plantations in part due to the fact that biodiversity has been found to be greater in shade coffee plantations than in sun coffee plantations (Perfecto et al., 2002; Borkhataria, 1993).

In 2006, the secretary of Agriculture of Puerto Rico announced the allocation of funds to a program to develop 1,000 new cuerdas of shade coffee at a one-time cost of $1422/ha (Cruz, 2006). This figure is considered an upper bound for the willingness of the Puerto Rican government to pay for biodiversity. While it provides an idea of the magnitude of funding that is being made available for projects that involve biodiversity preservation, it cannot be used as an indicator of the economic value of services provided by biodiversity There are several reasons why this number is unlikely to be an accurate indication even of the government's willingness to pay for biodiversity preservation. First, there are many possible reasons to support shade coffee production, only one of which is biodiversity preservation. Another caveat is that this is only one value from one program and the reality is that overall sun and shaded coffee plantations are equally likely to get government assistance in the form of subsidies (Borhkataria, 1993).

Table 4.1.6. Estimates of biodiversity value in humid tropical forests

Source	Location	Type of Valuation	$/ha/yr
EEP (2003)	Various	pharmaceutical	$0.20 to $695
Pearce (2001)	Various	pharmaceutical	up to $3000
Barbier and Aylward (1996)	Costa Rica	pharmaceutical, in biodiversity hotspots	up to $20
Pagiola et al. (2004)	Costa Rica	payment for ecosystem services, conversion from degraded pasture to forest under Regional Integrated Silvopastoral Ecosystem Management Prograom (RISEMP)	$67
Author's calculation	Puerto Rico, Carribean National Forest	U.S. Government spending on Puerto Rican parrot	$983
Cruz (2006)	Puerto Rico	payment by Puerto Rico government to develop 1,000 cuerdas of shade coffee	$1,422 (one time payment)

In addition to pharmaceutical and non-use values, the preservation of biodiversity contributes indirectly to the recreation value of forests. This is particularly true for biologically unique sites, which are highly valued by wildlife watchers. These values are further discussed in the following section.

Recreation

Benefits from recreation have been estimated using survey-based contingent valuation studies and by the travel cost method which is based on the idea that the amount that people pay to travel to a particular site is a minimum estimate of their willingness to pay for the recreation at that site. Wildlife watching is a significant economic activity. According to the 2001 National Survey of Fishing, Hunting and Wildlife-Associated Recreation, over 66.1 million people in the United States participated in wildlife watching activities and spent a total of $108 billion in 2001. The net economic value per year for a wildlife watcher in their resident state is $257 per year in 2001 US dollars. Wildlife watchers who travel outside their state have a different demand curve and have a net economic value of $488 per year (La Rouche, 2001). A review of the literature on tropical forest valuation estimates the value of recreation in tropical forests to range from 2-470 $/ha/yr in general, to be about $750 $/ha/yr for forests near towns, and about 1000 $/ha/yr for unique forests (Pearce, 2001).

In tropical countries, sites that are particularly interesting for eco-tourism may be able to take advantage of the high willingness to pay of ecotourists. An on-site survey at the Monterverde Cloud Forest Reserve in Costa Rica to value tropical rainforests by US ecotourists in Costa Rica estimated that U.S. tourists value ecotourism in Costa Rica at US$1150 per visit (Menkaus and Lober, 1996). Puerto Rico has several sites that are highly popular among ecotourists, most notably El Yunque, the Caribbean National Forest. El Yunque is home to 240 species of trees and at least four endangered species. Approximately 750,000 view the cloud forests in El Yunque each year (Scatena, 1994). Approximately half of the visitors to the park are from outside of Puerto Rico and typically visit during the winter and early spring (USDA Forest Service, 2007).

Rivers and lakes in forested areas also provide recreational opportunities for to local people in Puerto Rico. The most popular river recreation sites are located in the Caribbean National Forest. The half of the visitors of the Caribbean National Forest who are from Puerto Rico typically visit in July and August (USDA Forest Service, 2007). These are the hottest times of the year, when shade is a particularly desirable characteristic of a recreation area. At these sites, the most common recreational activities are picnicking, enjoying nature, visiting with friends, relaxing, and swimming (Table 4.1.7).

Average travel time is between 40 and 60 minutes and average travel cost was found to be $8/person based on gasoline expenditures. However, most survey respondents indicated additional willingness to pay for their recreation at the rivers. Respondents were presented with a price between 0 and $200 and asked if they would be willing to pay that additional price. When the price was under $100, eighty percent of respondents indicated they would pay the additional price. Thirty percent of respondents presented with numbers between $100 and $200 indicated a willingness to pay the extra price. A lower bound for the average additional willingness to pay is $30. Therefore, the willingness to pay of local residents per trip is estimated at $38. Estimates of the total value of selected river sites range from insignificant to $2.4 million per year (Table 4.1.8).

Per hectare recreation values for local visitors for the portions of the watersheds within the Caribbean National Forest range from $281 to $2053 (Table 4.1.9). Over the whole park the average value is $620/ha/yr. These values are likely underestimates because they only include travel costs and they only include value to local visitors during the summer months. While these calculations demonstrate that in some forested areas, the recreational value alone of forest exceed private returns to cattle ranching, this is not likely to be the case for all forests across the island. The sites located in the National Forest are among the highest-value recreational forest sites on the island.

Table 4.1.7. River visitor characteristics and activities, Caribbean National Park. Data collected through surveys conducted in 2005 by the NSF Biocomplexity Project.

Gender	
Female	47.7%
Male	52.3%
Age Group	
Adults	58.9%
Teenagers	11.8%
Children	24.2%
Reported Recreational Activities	
Picnicking/eating/drinking	39.9%
Sun bathing	18.8%
Enjoying nature	63.2%
fishing/shrimping	1.8%
Visiting with family & friends	78.8%
Relaxing	39.0%
Spiritual renewal/Therapy	11.3%
Swimming/Wading in River/cooling off in River	59.4%
Trip Characteristics	
Average length of time at river (min)	183.0
Average Enjoyableness of Visit (1-10)	8.7
Average Travel time (min)	60.8
Average Gasoline Cost ($)	$8.3
Average number of annual trips to river	2.6
Minimum number of annual trips to river	0
Maximum number of annual trips to river	329

Table 4.1.8. Estimated value of most popular river recreation locations to local visitors. Based on number of visitors, travel cost and average additional willingness to pay of local residents per trip calculated from surveys conducted by the NSF Biocomplexity Project.

River Location	Watershed	Average # Visitors/day	Average # Visitors/yr	Estimated Value $/yr
Charco Frio 1	Espirito Santo	720	64,800	2,462,400
La Mina Falls	Mameyes	570	51,300	1,949,400
Puente Roto	Mameyes	423	38,070	1,446,660
Charco Frio 2	Fajardo	366	32,940	1,251,720
La Vega	Mameyes	218	19,620	745,560
La Coca	Mameyes	201	18,090	687,420
Es Waterfall	Espirito Santo	181	16,290	619,020
El Verde	Espirito Santo	85	7,650	290,700
Juan Diego	Mameyes	78	7,020	266,760
Angelito Trail	Mameyes	65	5,850	222,300
Charco Frio Vereda	Fajardo	59	5,310	201,780
Sonadora	Espirito Santo	44	3,960	150,480
Jimenez Waterfall	Espirito Santo	40	3,600	136,800
Total		3,050	274,500	10,431,000

Table 4.1.9. Per hectare recreation value of forested watersheds within the Carribean National Forest to local visitors. Area obtained from Read and Lauituri (2005) and total value calculated using values from Table 4.1.8.

Watershed	Area (ha)	Total Value ($/yr)	Value $/ha/yr
Espirito Santo	9,065	1,653,840	404
Mameyes	2,590	267,120	2053
Fajardo	5,179	275,040	281
Total	16,834	2,196,000	620

The inland lakes in the study area are also used for recreation, particularly by local families. Unfortunately, little data is available on the recreational activities in these areas. Available (albeit slightly outdated) data suggests that although lakes have “high potential” for recreational use, they remain largely undeveloped (Hernández and Sánches, 1986). Many of the lakes lack public facilities including parking areas, restrooms, and trash cans as well as marked access roads. Surveys conducted at Guajataca, La Plata, and Dos Bocas indicate that the most common uses of the lakes were fishing and picnicking. The average visitor engaging in recreation at the lakes was male and had a yearly income of less than $10,000. Most lived less than an hour from the lake and came by car. Survey results indicate that the lakes are visited frequently by relatively small numbers of people and that demand for recreation increased with income and education. Although it is not feasible to determine a per hectare value, it is clear that the forests immediately surrounding some lakes have recreational value, particularly since "naturalness" was cited as the most important recreational characteristic of the area. The study concludes that development near the lakes should be curtailed to preserve their recreational value. In addition, the recreational value of the lakes could be increased by improvements in access.

While some areas in Puerto Rico are extremely valuable for recreational purposes, the majority of the forests in the study area are not adjacent to water sources and do not have high potential for ecotourism. For example, although birding is an economic activity in Puerto Rico, most birders focus on the "hot" birding sites which allow them to check of birds on their birding lists. Most birders can spot most Puerto Rican endemics by taking a trip to El Yunque and the Guanica dry forest, and perhaps a trip to the Maricao Commonwealth (J. Wunderle, electronic communication, January 30, 2006). Therefore the direct, onsite recreational value of forests that maybe converted to pastures to birders or other recreational uses are generally minimal. However, there may be indirect benefits if the continued existence of the birds in the hot spots depends on the existence of contiguous areas of habitat that includes some of these not-so-often visited forests.

Nevertheless upland forests and pastures can indirectly impact downstream recreation by affecting water quality. Excessive erosion and sedimentation may affect the number of fish available for fishing, or make swimming and picnicking appear less attractive. However despite decreases in the quality of the islands freshwater resources

(Hunter and Arbona, 1995) the use of reservoirs and streams for recreation has increased in recent years (Fred Scatena, personal communication, January 2007). Therefore, the current level of sedimentation in these reservoirs and waterways does not seem to be sufficient to deter recreational activities and this potential cost is not considered significant.

Existence Value

Existence value is the benefit people derive from knowing that a particular entity exists, even if it is never used. Existence values are often derived from cultural values. Attempts have been made to quantify existence values for tropical forests using survey-based methods. For example, Kramer and Mercer (1997) estimated U.S. residents' willingness to pay (WTP) a one-time donation to a hypothetical fund to protect an additional 5% of tropical rain forest. In their study, mean household WTP ranged from $21 to $31 and by applying those values to the U.S. population, total U.S. household WTP ranged from $1.91 billion to $2.82 billion. While it is evident that tropical forests in Puerto Rico probably have an existence value to some people, there is likely to be extreme variation between individuals and there are many problems with survey methods. In addition, generalizations from studies about tropical forests in general are not necessarily applicable to particular forests in Puerto Rico and quantification on a per hectare basis would be extremely difficult. A review of the literature find existence values for non-unique tropical forests range from 2-12 $/ha and about $4400/ha for unique areas (Pearce, 2001).

Hydrologic Public Benefits

Hydrologic ecosystem services were found to be the most economically significant of the services provided by forest and pasture in the study area. The conversion of forest to pasture provides both public costs and benefits. Since forests lose more water to evapotranspiration than pastures, conversion provides a benefit by increasing the quantity of water available for human consumption. On the other hand, conversion results in higher erosion which has negative impact on water quality. Therefore, the net effect depends on the magnitude of the benefits of increased water quantity versus the cost of decreased water quality.

Water Quantity

In the study area, forests conversion to pasture results in increased runoff. Larsen and Concepción (1998) found that that forested rural watersheds lost more water to evapotranspiration than did more urban watersheds. In northeast Puerto Rico it has been found that runoff from pastures is greater than runoff from forests, and that the difference is seasonal, with a larger difference in the wet season (Lopez-Rodriguez, 2006). Interestingly, one study in the coastal plains of Puerto Rico found that under certain conditions runoff was higher from forests than from tall grasses, indicating that the assumption that forests reduce the quantity of water available is not always valid (van der Molen, 2002). However, in the montane subtropical wet forests in question, available data suggests that conversion of forests to pasture would result in higher volumes of water available for human consumption (Table 4.1.10).

Table 4.1.10. Runoff in subtropical wet forests in lower montane regions of Puerto Rico.

Measurement Description (Source)	Pasture	Forest	Pasture-Forest difference
Daily runoff in wet season m3/ha/day (Lopez-Rodriguez, 2006)	36.89	21.03	16
Daily runoff in dry season m3/ha/day (Lopez-Rodriguez, 2006)	11.1	8.1	3
Annual runoff m3/ha/yr, northeast Puerto Rico (Lopez-Rodriguez, 2006)	10,132	6,588	3,723
Annual runoff m3/ha/yr, Fajardo grassland v. tabonuco forest (van der Molen, 2002)	28,679	28,132	547

Since the pasture is used for cattle grazing, the net additional amount of water available for human consumption is the additional runoff minus the amount of water used by the grazing cattle. In the central mountains of Puerto Rico, one hectare of pasture supports five to seven cattle per hectare (Personal communication: Interviews with farmers in Adjuntas, 2006). Based on daily water intake of cattle reported by Lardy and Stoltenow (1999) the annual intake of water by lactating cows, dry cows and heifers, and bulls are 20.2 m^3, 12.1 m^3, and 15.8 m^3, respectively. Therefore the annual intake of water per hectare ranges from 60.5 to 141.4 m^3, depending on the type of cow and the number per hectare. Using the average intake of the three types of cattle and assuming five cows per hectare, water use of cattle is estimated at 80 m^3 per hectare in this study.

An additional concern in terms of water quantity is that deforestation may lead to a reduction in precipitation. While precipitation levels are dominated more by large scale climate effects than by local land use changes, there is some evidence that significant deforestation may lead to a long term reduction on the magnitude of 1-4 mm/yr on islands (Bruijnzeel, 2006). Microclimate simulation suggests that complete deforestation of coastal plains and conversion to pasture would lead to a decrease in precipitation in

Puerto Rico (van der Molen, 2002). These effects are not considered in this study because the scientific literature is sparse and focuses on upwind coastal areas rather than downwind montane regions. Current evidence suggests that such effects are small, and particularly since effects are likely to only be significant when there are large scale changes in land use the marginal effect of converting one hectare of forest is considered extremely small and are only likely to reach significance if there are large-scale changes in land use.

The cost of maintaining forest rather than pasture is the value of the water that would be available for human consumption if the forest were converted to pasture. The market value of raw water is often difficult to determine. Prices are often set by the government, and tend to reflect the cost of service rather than the economic value of water. In Puerto Rico, prices are set by the Autoridad de Acueductos y Alcantarillados (AAA). Rates for water are found in Table 4.1.11.

Table 4.1.11. Current rates charged by the Autoridad de Acueductos y Alcantarillados for residential water provision in Puerto Rico. Residential rates depend on the amount of water consumed per month, and are increasing in consumption levels (AAA, 2006).

Bloques	Consumption (m3)	$/m3
Bloque 1	11 to 15	1.1
Bloque 2	16-35	1.6
Bloque 3	>35	2.16

The value of raw water is significantly less than the value of drinking water from a tap. Raw water can be seen as one input into the production of tap water. The value of water from a tap includes the value of the transport and treatment. As a result, when the distribution system is poor, the value of raw water is decreased because more raw water is required to produce tap water. In Puerto Rico, 43% of the water is lost in transmission, which significantly decreases the value of runoff (DRNA, 2005).

A survey of the literature from 1996 reported the maximum and average values of domestic water consumption in the U.S. as $0.46 and $0.15 respectively (Frederick, 1996). In U.S.2005$ this is a maximum of $0.66 and average of $0.22. Adams et al. (2004) compiled purchases of water reported in two trade publications between 1990 and 2001. As these are market transactions, they are a good indication of the market value of water. The majority of these purchases were for municipal and industrial use (M&I), which makes them relevant to this study. There is significant variation between states and over time. Average water purchase prices ranged from $0.17 to $4.53 per m^3. The average for all states considered was $1.57 and the median was $1.03. The states involved in these transactions are some of the most water-restricted states in the country, including Colorado and Nevada. However, since Puerto Rico has a much higher availability of water than these states, a reasonable assumption is that the value of water in Puerto Rico falls in the low range of these numbers. Hubbert et al (2004) report a range of $0 to 0.33 based on M&I purchases in Washington. The average value of $0.22 is considered reasonable for Puerto Rico and is used in this study.

Water Quality

Forest conversion to pasture generally has a negative effect on water quality. The most significant and most easily quantified water quality problem resulting from forest conversion is increased erosion. Erosion rates are dependent on many factors, including topography, soil conditions, and land use. In Puerto Rico in recent years, increasing urbanization and expansion of low-density housing has reversed the trend of increasing forest area in the late 1900s (Grau et al., 2003). The rate of sedimentation of the reservoirs has been linked to land use patterns, with construction activities and agricultural uses resulting in significantly higher rates of erosion than pasture or undisturbed forest (Gellis, 2006). The effects of land use on mean annual erosion and sediment discharge have been quantified in the Guadiana watershed in Puerto Rico, using GIS and the Revised Universal Soil Loss Equation (RULSE) (López et al., 1998) and the Lago Loiza basin using bathymetric surveys of the reservoir and land use history (Gellis, 2006). Both studies found that pastures resulted in significantly less erosion than cropland, but more than forests (See Table 4.1.12). Since the erosion rates are highly dependent on slope and soil type, the differences between forest and pasture will vary depending on the location.

Table 4.1.12. Comparison of erosion resulting from different land uses in Puerto Rico.

Measurement Description (Source)	Bare soil	Cropland	Pasture	Forest	Pasture-Forest difference
Sheetwash erosion in ppm, Lago Loiza Basin (Gellis, 2006)	61,400	47,400	3,510	2,050	-1,460
Median soil erosion rates in Mg/ha/yr, Gudiana watershed (López et al., 1998)	534	22	17	7	-10

The amount of sediment eroded from a pasture or forest that reaches the reservoir depends on the sediment deposition of the particular basin, which varies according to the size of the basin as well as other characteristics. Sediment yield ratios, or the ratio of sediment eroded within the basin and sediment delivered to the basin outlet, range from 0.08 for large watersheds (about 80 ha) to 0.33 for small watersheds (about 0.25 ha)

(Boyce, 1975). The sediment delivery ratio of the basin that contains the 2,380 ha La Plata reservoir has been estimated at 0.17 (López et al, 1998). The amount of sediment that settles in the reservoir leading to a loss of capacity is known as the trapping efficiency of the reservoir, which is the proportion of the sediment yield entering the reservoir to the capacity loss. The trapping efficiency has been determined for 14 major reservoirs in Puerto Rico, with an average of 0.85 (Soler-López, 2001).

Landslides also contribute to sedimentation problems, particularly in areas with steep slopes. Landslides result in large releases of sediments in short periods of time. The incidence of landslides has been found to increase with forest conversion. The difference in incidence is greater at higher elevations and higher slopes. For elevations relevant to the study area (i.e. greater than 400 m), at slopes less than 12 degrees, the average incidence of landslides in forests is 0.8/km^2/decade compared to 1.83/km^2/decade for pastures (Larsen and Torres-Sànchez, 1998). At slopes greater than 12 degrees, the average incidence for forests did not increase significantly, while the average incidence for pastures was greater, at 4.03/km^2/decade. The average landslide away from roads results in the movement of 1620 Mg of sediment (calculated from Larsen and Parks, 1997). The expected value of the mass of sediment displaced by landslides per hectare per year was calculated based on the frequency of landslides and the average mass from landslides (Table 4.1.13), Although these events are relatively infrequent, since each landslide is massive, the expected value of sediment movement due to landslides in any given year is significant compared to the amount of sediment moved by surface erosion.

Table 4.1.13. Expected value of mass of sediment displaced each year by landslides on one hectare of forest or pasture. Probability of landslide occurrence for each land use calculated from Larsen and Parks (1998) and average mass of landslide obtained from Larsen and Torres-Sànchez (1998).

Elevation & Slope	Forest	Pasture	Difference
>400m & <=12^{o}	1.4580	2.9700	1.5120
>400m & >12^{o}	1.2960	6.5340	5.2380

In addition to increasing sediment concentrations, urban development has also been shown to impact other water quality criteria such as metal concentrations and dissolved oxygen levels in Puerto Rico (Herler, 2002). There are significant concerns about water quality in the reservoirs, as the majority of reservoirs contain high concentrations of nutrients (Zayas et al., 2004). Unfortunately these aspects of the effect of land use change on water quality have not been sufficiently documented for use in this study.

On-site costs of erosion are also considered to be minimal in these areas of Puerto Rico. Studies of forest recovery on recent landslides indicate that because atmospheric inputs of nutrients are high and the soils are deep and relatively nutrient rich, productivity quickly recovers, even in severely eroded areas (Zarin and Johnson, 1995). Moreover, onsite erosion does not seriously deplete soil resources and result in prolonged degradation of pasture or the need for fertilization. Therefore, only off-site costs of erosion are considered here.

Off-site costs associated with sedimentation of the reservoirs include dredging costs, reservoir capacity loss and water treatment costs. Dredging costs are considered the most significant. The majority of Puerto Rico's reservoirs are operating with zero dead storage (DRNA, 2006). As a result, any additional sediment reduces the capacity available for water storage and will need to be dredged in order to maintain the maximum operating capacity.

The cost of dredging sediment used in this study is \$13.4 per m^3, the cost of dredging a reservoir in Puerto Rico reported by PRASA (PRASA, personal communication, May 2006). Costs of dredging from other studies indicate that this value is comparable to dredging costs in other areas. Published costs of dredging range from \$1.95 to \$17 per m^3 for navigational and reservoir purposes, and from \$34 to \$1409 per m^3 for projects that require environmental remediation (Table 4.1.14). Using a bulk density of 0.7 g/cm^3 (Richard Webb, electronic communication, May 3, 2006), this amounts to \$19 per tonne of sediment removed.

Table 4.1.14. Dredging costs. \$/Mg is calculated from \$/m^3 using the bulk density reported by PRASA (0.7 g/cm^3).

Source	Description	\$/m^3	\$/Mg
(Hansen et al., 2002)	Navigational dredging	-	<5
(Henshaw et al., 1999)	Great Lakes navigational Dutch Transport navigational Modeled hydraulic dredging costs	1.95- 2.87 12 – 14 7.61 - 15.25	- - -
(Crowder 1987)	Reservoir dredging	2.03	-
(Blazquez et al., 2001)	Mechanical costs of dredging Environmental dredging projects	<17 34 – 1409	- -
PRASA	Reservoir dredging Puerto Rico	13.4	19.14

Between dredging projects there is a loss of capacity which induces additional costs on society when water that would have been available in the absence of the increased sedimentation is not available for consumption. The cost of lost capacity depends on the use of the water and whether the capacity of the reservoir is limiting withdrawals. This loss may be valued using the value of the foregone water. A related cost estimate is the cost of building new capacity. A survey of reservoir building projects up to 1987 found that new reservoir capacity cost $300-700 per acre ft, or $0.28 to 0.41 per m^3 in 2005$ (Crowder, 1987). The problem with estimating cost in this manner is that it implicitly assumes that new reservoirs can be built. However, in Puerto Rico, most of the appropriate sites for reservoirs have already been utilized (Hunter and Arbona, 1995). Therefore this cost was not included in this analysis.

Although increased water treatment costs decreased water quality are often cited as a concern, the available literature suggests that such costs are fairly small. One study found that on average sediment discharges to surface waters induce treatment costs of only $17.11 per thousand tons of sediment (Holmes, 1988), or $0.028 per ton in 2005 U.S. dollars. Similarly, Pimentel (1995) suggests an average cost of $0.03 per ton. Dearmont et al. (1999) reported that a one percent increase in turbidity increases chemical costs of water treatment by one fourth of a percent. A cost of $1.14 per ton of sediment was estimated by extrapolation from data presented in Dearmont et al. (1999). This is very high compared to estimates in the literature. For demonstration, the dredging and water treatment cost values were used to calculate sediment costs for seven major reservoirs in Puerto Rico for one hectare of land at the average slope in the study area (11 degrees) (Table 4.1.15).

Table 4.1.15 Example sediment costs for seven major reservoirs in Puerto Rico assuming average slope. All sediment values are in Mg/ha/yr and cost values are in $/ha/yr. Sediment and cost values were calculated as described in Section 3.2. SDR_b was estimated based on basin size.

Reservoir:	Caonillas	Dos Bocas	Garzas	Guayo	La Plata	Loíza	Yahuecas	Average	Median
Forest *Total Sediment*	*1.89*	*1.64*	*2.79*	*2.50*	*1.52*	*1.47*	*2.36*	*2.02*	*1.89*
RUSLE	1.20	0.95	2.10	1.81	0.83	0.78	1.67	1.33	1.20
Landslides	0.69	0.69	0.69	0.69	0.69	0.69	0.69	0.69	0.69
Cost of sediment	*33.61*	*26.74*	*50.48*	*46.23*	*25.42*	*21.63*	*25.47*	*32.80*	*26.74*
Dredging	33.46	26.45	50.32	46.14	25.20	21.23	24.22	32.43	26.45
Treatment	0.15	0.28	0.16	0.09	0.22	0.40	1.25	0.36	0.22
Pasture *Total Sediment*	*6.22*	*5.64*	*8.26*	*7.61*	*5.36*	*5.24*	*7.29*	*6.52*	*6.22*
RUSLE	2.76	2.18	4.80	4.15	1.90	1.78	3.83	3.06	2.76
Landslides	3.46	3.46	3.46	3.46	3.46	3.46	3.46	3.46	3.46
Cost of sediment	*110.37*	*91.94*	*149.60*	*140.57*	*89.75*	*77.34*	*78.61*	*105.46*	*91.94*
Dredging	109.87	90.96	149.12	140.31	88.97	75.91	74.75	104.27	90.96
Treatment	0.50	0.98	0.48	0.26	0.78	1.43	3.86	1.18	0.78
Difference in cost ($/ha/yr)	***76.76***	***65.20***	***99.12***	***94.34***	***64.33***	***55.71***	***53.14***	***72.66***	***65.20***

Summary

The range of values for the private and public costs and benefits of maintaining forest cover rather than pasture are presented in Table 4.1.16. From a private perspective, forest conversion to pasture is the most profitable option. However, from a public perspective it is unclear whether forest or pasture is most beneficial. The range of values for the cost of water quantity loss and benefits of water quality improvements suggest that in some conditions, or rather at some slopes, the costs will outweigh the benefits and vice versa. In addition, including benefits other than water quality have the potential to change the direction of the public externalities. The following section describes the results of a simple model created using the data summarized in the table below. The purpose of the model is to determine at which slopes the benefits of forest conversion outweigh the costs.

Table 4.1.16. Private and public costs and benefits of subtropical wet forests in Puerto Rico. Unless otherwise stated, all units are in $/ha/yr. Range of values reported in literature for tropical forests (Almeida and Uhl, 1995; Aylward and Echeverría, 2001; Pearce, 2001; Aylward, 2002) compared with estimates compiled for subtropical wet forest zone in Puerto Rico.

	Tropical forests literature review*	Estimate for Subtropical Wet Forest Zone, Puerto Rico
Private		
Conventional logging	20 to 440	n/a
Sustainable logging	30 to 266	-53 to 290
Fuelwood	40	n/a
NTFPs	0 to 100	Unquantified--value of direct use of water gathered from water spouts
Opportunity cost of cattle ranching	-1053 to -258	-500 to -400 (Personal Communication, USDA)
Public		
Watershed benefits – quantity	-1,100 to 15	-820 to 0 (Lower bound from high estimate of difference in runoff valued at $0.22.)
Watershed benefits – quality	0.25 to 850	9 to 160 (Values for the average reservoir, SDR = 0.17 with slopes ranging from LS =1 to LS = 17)
Recreation	2 to 470 (general) 750 (forests near towns) 1000 (unique forests)	2 to 470 (general) n/a 280-2050 (Caribbean National Forest)
Climate benefits (Carbon)	360 to 2200 gross present value, not annualized	8 to 36 (Sequestration value)
Air quality	n/a	0 to 88 (estimated by CITYGREEN model, American Forests, 2002)
Genetic information	0 to 3000	n/a
Biodiversity other than genetics	?	0 to 67 (RISEMP payments for biodiversity from conversion of pasture to forest)
Nonuse values	2 to 12 4400 (unique areas)	n/a

4.2 Model Results

The outputs of the model for several different scenarios are presented in Table 4.2.1. Figure 4.2.1(a) shows the total hydrologic costs of each land use as a function of topographic factor. Figure 4.2.1(b) shows the same total hydrologic costs as a function of slope, assuming a slope length of 72.6 ft. Figure 4.2.2 shows the total costs, public and private, including hydrologic ecosystem services, carbon sequestration, and the private benefits from pasture. When only hydrologic ecosystem services are taken into account the benefits of forest begin to outweigh the benefits of pasture at a topological factor of 6.49. When the benefit of carbon sequestration is included, this result is reduced to 4.00. For some slope lengths, this factor is similar to the LS factor that would occur on the average slope in the lifezone. However, when private benefits to pasture are included, the sum of the public and private benefits of pastures is greater than benefits of forests at all topographic factors.

The results are very sensitive to changes in certain variables. The sediment delivery ratio of the basin is very important in determining costs of sediment. For illustration, if a sediment delivery ratio of 0.8 is used, there are some slopes at which the public cost from sedimentation outweighs even the private benefits of pastures. In addition, small changes in the value of water can result in very different outcomes. For example, any scenario in which the value of water is greater than $\$0.42/m^3$ results in the outcome that the benefits of pasture outweigh the benefits of forest on all slopes.

Table 4.2.1. Model results: LS is minimum LS where cost of pasture begins to exceed cost of forest. Slope (degrees) is calculated assuming a slope length of 72.6.

Scenario	**LS**	**Slope**
Public Only – Hydrologic	6.49	20.89
Public Only - Hydrologic and Carbon	4.00	12.02
Public and Private	none (pastures always better)	none
Alternate Public and Private, SDR_b= 0.8	11.02	38.77
Any scenario where value of water > 0.42	none (pastures always better)	none

Figure 4.2.1(a). Total hydrologic cost as a function of the topographic factor

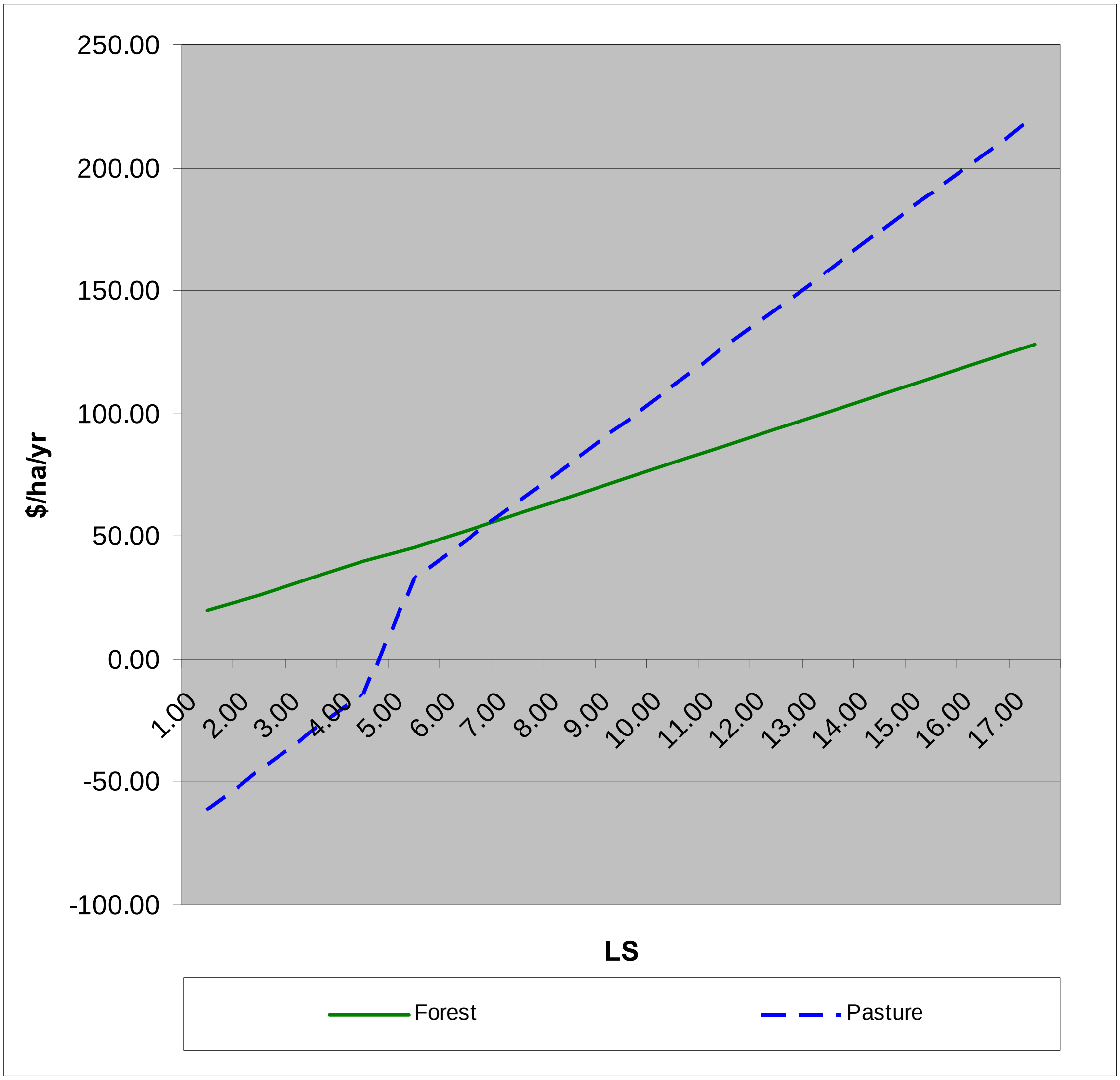


Figure 4.2.1(b). Total hydrologic cost as a function of slope

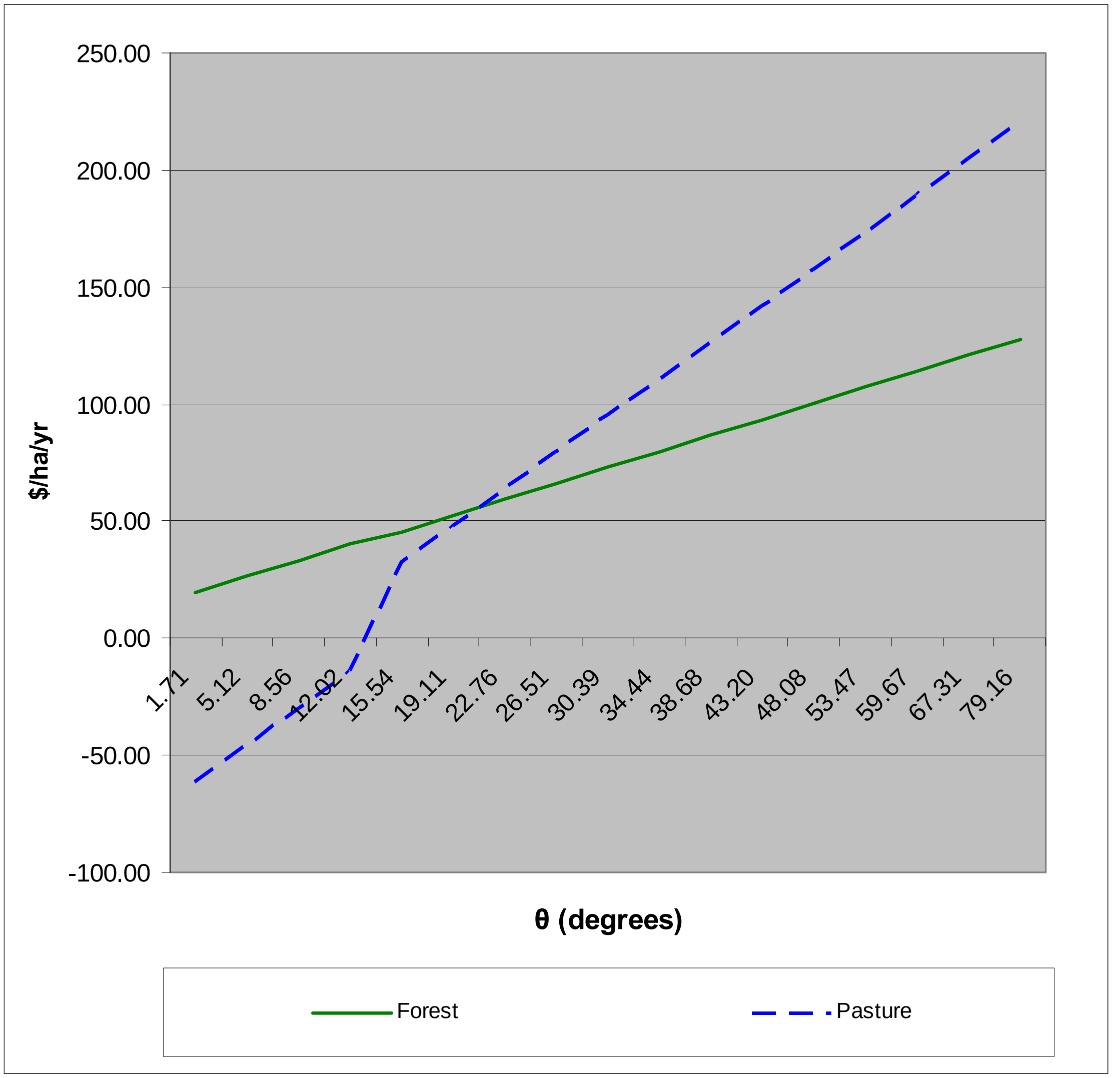
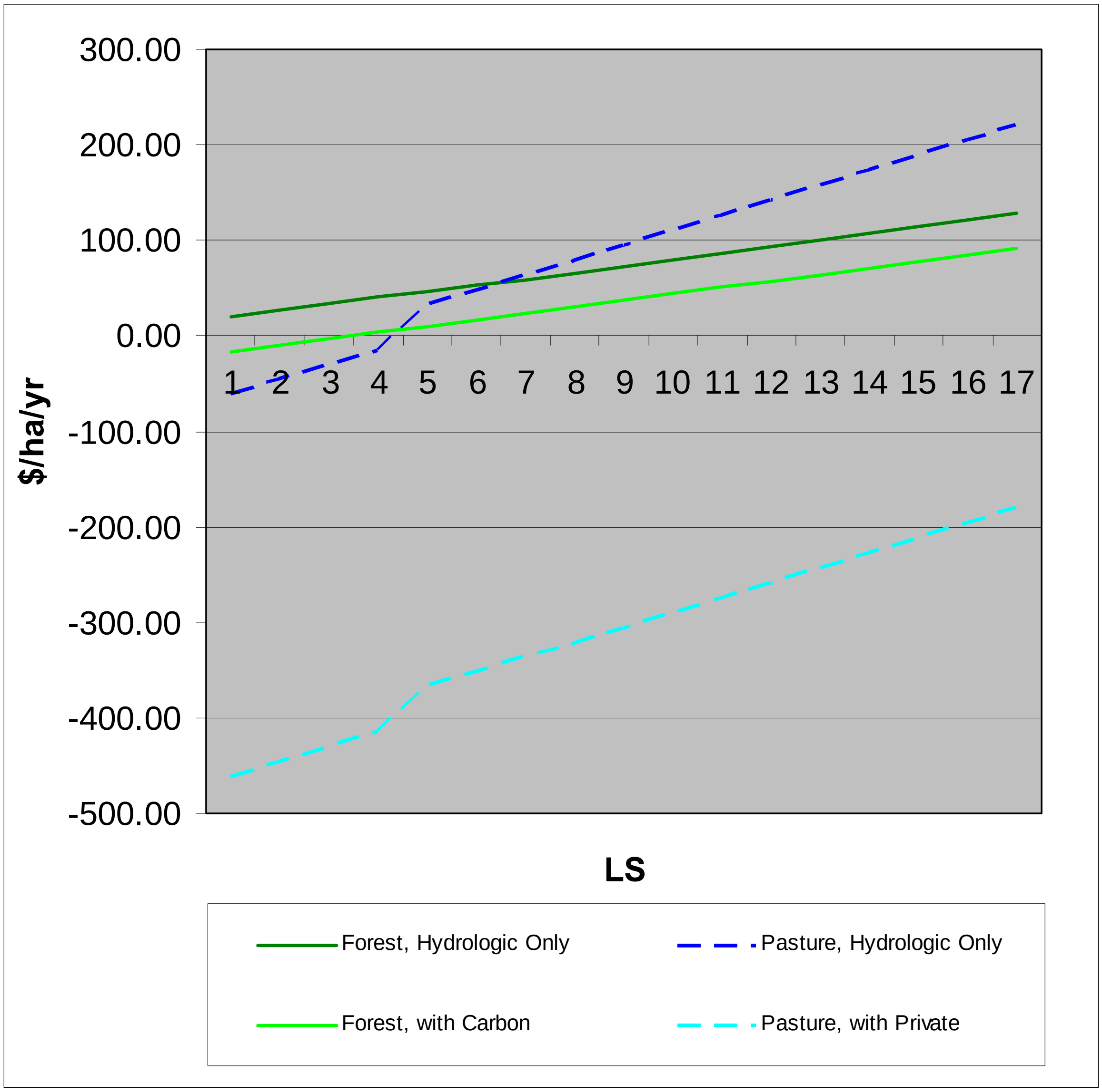


Figure 4.2.2. Public and private costs and benefits



5. Discussion

5.1 Outcomes and Limitations

According to this analysis, the direction of hydrologic externalities under forest conversion to pasture may be positive or negative, depending on certain local conditions. In particular, the local topography and the sediment delivery ratio are very important in determining whether the cost of sediment is greater than the benefit of increased water supply. For example, when only hydrologic services are considered, the direction of externalities due to forest conversion depends on the slope and becomes negative at slopes of approximately 21^{o}. When public externalities include hydrologic externalities and carbon, this slope is reduced to 12^{o}, about the average slope in the lifezone. However, when private returns to pasture are included in the analysis, they always outweigh the benefits of forest by several hundred dollars.

Hydrologic ecosystem services were found to be the most economically significant ecosystem service in the study area. When calculated using the difference in water yield and the estimated value of water, the benefits from additional runoff from pasture were calculated at $102/ha/yr. This is of higher value than the erosion prevented from forests even at some slopes above the average in the study area. The most significant costs were found to be dredging costs. Water treatment costs were found to be fairly insignificant. In addition, the inclusion of landslides significantly changed the outcome. When only the erosion from RUSLE is evaluated, the benefits of extra runoff outweigh the benefits of erosion control at all but the very steepest slopes.

On an island-wide basis, the benefits of other ecosystem services provided by forests are smaller than expected, and not high enough to obviously outweigh private benefits to cattle ranching. The most significant and easily-evaluated of these other services is carbon sequestration. When this value was included, public costs of pastures began to exceed costs of forests at about the average slope for the lifezone. Recreation values are high enough in local areas, such as the Caribbean National Forest, to single-handedly outweigh the private benefits of alternate land-uses. However, this is unlikely to be the case in general. Recreation values may increase in the future if recreation sites along reservoirs become more developed. Biodiversity values as evidenced by government willingness to pay for preservation are likewise very high in select areas, but low in general. These values would have to be evaluated on a case-by-case basis.

This study does not consider issues of spatial or temporal scale. The goal of this study was to get a sense of the direction of hydrologic externalities resulting from forest conversion to pasture on a general island-wide basis for a particular lifezone. As a result, the model uses average constant values for many parameters. This fails to take into account issues of scale and the difference between marginal and average costs, as well as local variations. The outcomes may be different when looking at the possibility of conserving or converting large areas of forests, rather than one hectare at a time. For example, it may be that there is a critical level at which sediment becomes a problem for recreation, which would be a problem if many hectares of forest are converted, but not just one. The marginal benefit of forests is likely to increase as total area decreases.

In addition, the results indicate that even within the lifezone, local conditions are important. Therefore the use of average constant values for many parameters may not be appropriate in all areas. For example, private benefits to cattle ranching are assumed to be constant for all slopes, which may not be the case. Other parameters such as the sediment delivery ratio and value of additional runoff are also likely to vary on a case-by-case basis. These issues are discussed further below.

The method of evaluation used in this study is very specific to the water supply situation in Puerto Rico and makes several assumptions about the costs of sedimentation and benefits of water supply that are important in determining the model outcomes. Most importantly, the model considers only water supply costs, and the major cost is considered to be the loss of reservoir capacity which results in the need for dredging. The first important assumption is that dredging programs will be carried out on a regular basis and that all sediment that reaches the reservoirs will be dredged. Otherwise, using dredging costs is not an appropriate estimation of costs of erosion. While the government currently plans to carry out dredging programs, it is possible that the planned projects will not be carried out, or that other policy options will be chosen in the future. In either case, other methods of valuing costs of erosion will be necessary depending on the policy chosen. There are other ways of dealing with sediment problems, such as changing the trapping efficiency, which may have other costs. In some areas, costs have been estimated using costs of building new reservoir capacity. However, this method implicitly assumes that new reservoirs can be built, while in Puerto Rico, most of the appropriate sites for reservoirs have already been utilized (Hunter and Arbona 1995). Given the current policy recommendations of the government, valuing erosion prevention based on dredging costs appears to be appropriate but this may change in the future.

The second critical assumption is that only sediment that reaches the reservoir incurs costs. Only sediment that reaches the reservoir is included in cost calculations. As a result, the sediment delivery ratio is critical in determining the cost of erosion.

Sediment delivery ratios exhibit considerable local variation. An accurate evaluation of the sediment costs in a particular area requires an accurate estimation of the sediment delivery ratio, which can be done using GIS modeling techniques. In addition, this assumption ignores any in-stream damage to aquatic resources that occurs as a result of erosion. The significance of these effects is not assessed in this study.

A third critical assumption is that all of the additional runoff enters the water supply, and can be valued using an average per value per cubic meter. Determining the economic value of water was a challenge because of a lack of data from Puerto Rico. Since the value of extra runoff largely determines the direction of the hydrologic externalities, estimating the value of water is critical to providing accurate policy guidance. The value of additional water depends both on the demand and supply, which may vary locally and over time. Although there are some areas where demand is underserved, there is not a shortage of water in Puerto Rico. While a lack of reservoir storage capacity is a big problem in times of drought, many of the water supply problems are related to inefficiencies and problems in the delivery system rather than a physical lack of water. This suggests that policies that address water quantity issues should focus more on improving the infrastructure system than on increasing the amount of runoff available. For comparison, in 2002, AAA lost 326 million m^3 in transmission (Vargas, 2005), 215 million m^3 more than would be lost by an average, well-functioning system that loses only 15% in transmission. If all of the land in the lifezone were converted from forest to pasture (an overestimate because not all the lifezone is currently in forest), the additional runoff would be only 112 million m^3. This is only about half of the amount that could be saved by infrastructure improvement

There are several reasons to believe the marginal value of the extra water coming off Puerto Rican pasture may be close to zero. First is due to the difference between marginal and average value. One method of valuing water involves deriving a demand curve for water using price and quantity data and estimating the change in consumer surplus resulting from the change (Young, 2005). The average value of water is obtained by dividing the surplus by the change in volume. In this case the change in value from 1 hectare of water is extremely small compared to the amount of water already being consumed, and therefore the change in consumer surplus is essentially zero. The implication is that valuing water may be more appropriate when considering a larger land-use plan than when trying to determine the cost of a one-hectare change.

The second reason that the marginal value of the extra water may be close to zero has to do with seasonal variation and depends on the capacity of the reservoir to store the extra volume. Soler-López (2001) has suggested that because Puerto Rican reservoirs are small, they have a low capacity to store available runoff. If the extra runoff occurs during a time in which water is not scarce and is not stored in the reservoir, this extra water has no extra economic value in terms of water provision. Major water shortages occur in times of drought, and it is during the dry season when there is the smallest difference between pastures and forests in terms of volume of runoff.

Additional data is necessary to improve this model. Most importantly, there are conflicting estimates of the physical difference in runoff volume between pastures and forests. For an accurate estimate, the difference must be evaluated considering pastures and forests that experience the same local conditions and have the same slopes, preferably adjacent plots. Other pieces of data that need to be more accurate and evaluated on a case-by-base basis are the value of water and the sediment delivery ratio. Additionally, a more accurate analysis would include seasonal differences and reservoir capacity restrictions. Other issues that are not considered which may be important include the long-term effects of cattle grazing on soil quality, erosion, and nutrient run-off, as well as other in-stream costs.

5.2 Policy Implications and Recommendations

The Puerto Rican government currently provides a tax break for forest conservation. A subsidy has been defined as "an incentive provided by government to enable and persuade a certain class of producers or consumers to do something they would not otherwise do, by lowering their opportunity cost or otherwise increase the potential benefit of doing so" (Hoek-Smit, 2003). A tax break can be seen as a subsidy which reduces the opportunity cost of foregoing the land's alternate uses. The two questions that must be addressed when designing such a subsidy scheme are: (1) Does the tax break change behavior in the intended way? (2) Does the benefit to taxpayers from the change in behavior exceed the cost of the tax break?

A detailed behavioral study would be required to answer the first question. However, the data on private returns provides some insight into whether the tax break is likely to be effective. The value of the tax break was found to range from about $30 to $105/ha/yr. Private returns to cattle ranching were found to be significantly higher than this amount, in the range of $400 to 500/ha/yr. Meanwhile, private returns to forest were found to be insignificant, which suggests that the opportunity cost of foregoing forest conversion is at least $400. Given the large difference in the opportunity cost of foregoing forest conversion and the size of the tax break, it seems unlikely that the tax break in itself would provide a strong incentive for a farmer to forego forest conversion. If the farmers are choosing not to convert the land to pastures for reasons other than the tax incentive, then the money is not being put to good use but is merely serving as a transfer of wealth from taxpayers to individual farmers.

The results of this model suggest that the answer to the second question depends on local conditions. While some of these conditions are not addressed in the model and need to be further studied, the results of the model provide some insight into when the tax break may be appropriate. This determination is complicated by the fact that the tax break is determined by the market value of the land and varies in different areas. On the one hand, the higher the tax break,the more likely it is to change behavior. On the other hand, the higher it is the more likely it is to exceed its worth to society. However, if the tax break is assumed to change behavior and induce forest conservation, then it is worthwhile to taxpayers if the difference in public cost between pastures and forest is greater than the amount of the tax break. According to this model, when the tax break is $30, this occurs at an LS factor of about 8, while when the tax break is $105 this occurs only at the very steepest of slopes. An LS factor of 8 is about twice the average slope in the area. While the model could be improved in many ways, this result suggests that even if the tax break were successful in inducing behavior changes, the cost to society of the tax break is not necessarily always less than the benefits of forest conservation.

Both the results of this model and the literature review emphasize the importance of site-specific variables. The literature review supports the notion that local conditions are important, and that using average values for all areas may not be appropriate. For example, one study in the Andes found that the response of the river in question to land use change "mainly depends on the changes in spatial organization and connectivity of various land cover types within the catchment, and not so much on the overall amount of land-use/cover change" (Vanacker et al, 2005). Furthermore, in Puerto Rico specifically, simulations suggest that the reforestation of the 5% of the watershed with highest erosion rates would decrease basin-wide erosion rates by 20% (López et al., 1998).

Policy recommendations identified through this analysis are as follows:

- **Most importantly, an effective land use policy should be targeted to specific areas, rather than providing blanket provisions that apply to all forested land.** The areas most sensitive to erosion (e.g. with high slopes, meaning slopes above 12^{o}) and with the highest sediment delivery ratios (e.g. above reservoirs) should be identified and prioritized as targets for land use policies. Once identified, well-targeted regulations may be designed or land purchases may be undertaken to restrict development in these areas.

- **More effort should be put into understanding the incentives of farmers.** The results of this model suggest that the public benefits of forest only unambiguously outweigh the public benefits of pasture at high slopes (greater than 12^{o}). This suggests that the willingness of the public to pay to prevent conversion to pasture should be higher for areas with higher slopes. However, it is also possible that these areas with higher slopes are less profitable than other areas and would therefore require a smaller incentive to induce behavior changes. Although they may be politically popular, handouts at the taxpayer's expense should not be given to farmers who would have kept their land forested in the absence of the subsidy. The blanket provisions of the current system do not appear to take into account either the differences in public value of forest conservation in different areas, or the incentives of individual farmers.

- **Policy-makers must consider changes in incentives and other factors over time.** One problem with the current incentive scheme is that tax breaks incur public costs on a yearly basis. However, incentives for farmers as well as other factors change over time. A farmer may be paid for several years to preserve land in forest, and then decide to convert the forest to another land use. In addition, the direction of hydrologic externalities may also change over time. Although the current highest-private-value land use in the area is pasture, this may change in the future as the population increases and the interior of the island becomes more developed. If more of the land area is used for residential development, the situation will change significantly. While this study views the water lost to evapotranspiration as a cost of forests, other studies have cited the flood control benefits of decreased runoff as a benefit of maintaining forest cover (American Forests, 2002). Unfortunately, determining the exact site-specific public and private costs and benefits of preventing forest conversion on every piece of land on a yearly basis would require so much data as to be impossibly expensive and time consuming.

- **Consider land purchases.** Under some circumstances, land purchases are a more effective use of tax dollars than providing yearly incentives. For areas that are identified as particularly sensitive or particularly valuable for recreation or biodiversity preservation, the public value of preserving the land is high. Buying the land outright would save the stream of tax dollars and ensure the preservation of the forest in these critical areas. Land purchases also avoid the cost of creating and enforcing regulations that restrict development on lands with high topographic factors. A simple calculation can determine how many years of tax benefits would pay for the purchase of the land. If only the costs of taxes are considered and discount rates are significant, land purchases will take a very long time to pay off. For example, even with a very low discount rate of 2%, a purchase of land valued at $1000 would avoid a net present value of $1000 in tax payments only after 63 years. However, in cases where it is likely that incentives will change in favor of pasture or urban development and forest conversion has high public costs, land purchases may be worthwhile. A policy of land purchasing is being pursued in the Karst region to protect the quality of the groundwater. Similar policies should be considered to protect surface water quality in other parts of the island.

- **Encourage best management practices and agro-forestry.** An additional approach, which is also being currently pursued in Puerto Rico, is to encourage good management of pastures or agro-forestry rather than forest conservation. From an economic perspective, these programs have the advantage of preserving some of the functions of forested areas, such as erosion prevention, without sacrificing all private benefits from agricultural production. Particularly when private returns to pasture are high compared to forest, this approach may have the highest total society-wide economic benefits. However, prior to designing new schemes to encourage agro-forestry, existing schemes must be evaluated in order to avoid a situation with conflicting incentives.

6. Conclusion

In Puerto Rico, as in many other parts of the tropics, forest conservation provides public benefits. In particular, due to Puerto Rico's dependence on reservoirs for its water supply, erosion control is a socially valuable function of forests. The idea that forests provide public benefits has been used to justify public spending on forest conservation. However, the conversion of forests to pasture not only results in private benefits to the landowner, but in some cases may provide public benefits greater than the costs of increased erosion. Other ecosystem services provided by forests may be significant enough to outweigh the private benefits of pasture in some areas, but not in others. It is important for policy-makers to consider such trade-offs and recognize the importance of site-specific variables and appropriate targeting when designing land use policies.

Although perfectly site-specific evaluations would prove prohibitively costly, the framework of this model may be adjusted to account for more site-specific or basin-specific conditions. Further research should focus on designing a more accurate framework for determining the value of water, and on increasing the availability of data on water yield differences between forests and pastures, sediment delivery ratios and private returns to land with different characteristics.

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