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Economic Incentives for Watershed Protection: A Report on an Ongoing Study of Arenal, Costa Rica

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Abstract

Tropical moist forests provide a range of goods and services to society. Traditionally, decisions regarding tropical forest land use have been made on the basis of major direct uses of forest land that generate local and national benefits. Typically, this has meant timber extraction and the conversion of forest to agricultural or livestock uses. In recent years increasing attention has been given to the important economic role non-market benefits may play in providing incentives for the conservation of tropical forests. A number of studies have explored the local, national and global benefits generated by non-timber forest products, ecotourism, pharmaceutical prospecting and carbon storage.

Another important ecological service that is often cited as an economic justification of conservation activities is the watershed protection function provided by tropical forests. Soil and water conservation may yield benefits to land-owners and alleviate damage to downstream economic activities. Nevertheless efforts to conserve watersheds are plagued by the difficult nature of the externalities involved. The off-site nature of many of the benefits of conservation activities makes both valuation and internalization of these externalities difficult, thereby preventing the development of 'sustainable' watershed protection programs. This is even the case in areas where pristine, mountainous forests provide downstream national benefits to hydroelectricity and irrigation schemes. The establishment of incentive systems that solve market, policy and institutional failures impeding watershed protection in such areas remains a vexing problem for policy-makers, scientists and communities in developing countries. Drawing on the literature and on-going research in Costa Rica, the paper outlines a collaborative research project investigating the potential for economic incentives for watershed protection in the Arenal region of Costa Rica.

Resumen

Los bosques tropicales húmedos ofrecen a la sociedad una gran variedad de bienes y servicios. Tradicionalmente, las decisiones tomadas con respecto al uso de la tierra de los bosques tropicales tienen como base el uso directo de las tierras forestales que generan beneficios locales y nacionales. Típicamente, esto es la extracción de madera y la conversión del bosque hacia usos agrícolas o ganaderos. En años recientes, se ha prestado una mayor atención al importante papel que pueden desempeñar los beneficios fuera del mercado para dar incentivos para la conservación del bosque tropical. Un número de estudios ha explorado los beneficios locales, nacionales y globales que generan los productos no-madereros de la selva, el ecóturismo, las prospecciones farmacéuticas y el almacenamiento de carbón.

Otro importante servicio ecológico citado a menudo como justificación económica de las actividades de conservación, es la función protectora de las cuencas que presta el bosque tropical. La conservación de la tierra y el agua puede ofrecer beneficios para los propietarios de la tierra y aliviar el daño a actividades económicas río abajo. Sin embargo, los esfuerzos por conservar las cuencas están acosados por el difícil carácter de las externalidades envueltas. El carácter no-local de muchos de los beneficios de las actividades de conservación hace que la valuación e internalización de estas externalidades sea difícil, de ese modo impidiendo el desarrollo de programas "sostenibles" para protejer las cuencas. Este también es el caso en áreas donde bosques prístinos y montañosos ofrecen beneficios nacionales río abajo para planes de hidroelectricidad e irrigación. Un punto de discordia para los tomadores de decisiones, científicos y comunidades de países en vía de desarrollo, es el establecimiento de sistemas de incentivos que resuelvan las fallas políticas, institucionales y del mercado que impiden la protección de las cuencas en estas áreas. Con base a información obtenida en la literatura y a la investigación que continua en Costa Rica, este ensayo diseña un proyecto de investigación colaborativa sobre el potencial de los incentivos económicos para la protección de las cuencas en la región de Arenal en Costa Rica.

Abrégé

Les forêts tropicales humides apportent aux sociétés humaines toute une gamme de produits et de services. Traditionnellement, les décisions relatives à l'utilisation des terres forestières tropicales ont été prises sur la base de grandes catégories d'utilisation directe des forêts aboutissant à des résultats bénéfiques au plan local et national. Cela s'est traduit, typiquement, par l'exploitation du bois et la conversion de ces terres forestières en terres arables ou pastorales. Ces dernières années, on a accordé une attention croissante à l'important rôle économique que des produits forestiers non directement commercialisés sont susceptibles de jouer en tenant lieu d'incitation à la préservation des forêts tropicales. Un certain nombre d'études ont traité des bénéfices locaux, nationaux et mondiaux engendrés par les produits forestiers non ligneux, par l'écotourisme, la prospection pharmacologique et le stockage du carbone.

Il est un autre service écologique majeur souvent cité comme justification économique des activités de conservation, à savoir la fonction de protection des bassins hydrographiques qu'assument les forêts tropicales. La conservation des sols et de l'eau est susceptible de retombées avantageuses pour les propriétaires fonciers et peut remédier aux dégâts subis par les activités économiques d'aval. Néanmoins, les efforts de conservation des bassins hydrographiques sont handicapés par la nature délicate des facteurs externes qui les touchent. Le caractère éloigné de nombreux avantages tirés des activités de conservation rend difficile tant l'évaluation que l'internalisation de ces facteurs externes, bloquant ainsi l'élaboration de programmes «durables» de protection des bassins hydrographiques. On rencontre ce cas de figure même dans les régions où des forêts vierges montagneuses ont en aval des effets positifs, d'ampleur nationale, pour les réseaux hydroélectriques et d'irrigation. Dans les pays en développement, les politiques, les scientifiques et les communautés locales restent confrontés aux difficultés subies pour établir des systèmes d'incitations permettant de contrer les impasses (celles du marché, des politiques suivies et des institutions) faisant obstacle à la protection des bassins hydrographiques de ces régions. S'appuyant sur la littérature publiée et sur des recherches en cours au Costa Rica, ce texte esquisse les grandes lignes d'un projet de recherche en collaboration, dans le cadre duquel on étudie le potentiel d'incitations économiques favorables à la protection du bassin hydrographique de la région de l'Arenal, au Costa Rica.

Contents

Introduction

This paper presents an overview of an ongoing collaborative research project entitled *Economic Incentives for Watershed Protection: A Case Study of Arenal, Costa Rica.* The project is conducting an examination of soil and water conservation/degradation issues in the upper watershed area of Lake Arenal in Costa Rica. From a biophysical standpoint, actual land use allocation in the study area suggests that land resources are 'over-used'. The study team will employ economic and biophysical analyses to assess whether land use allocation in the area is economically efficient from private and social perspectives. In particular, the study seeks to include explicitly both the on- and off-site soil and water impacts of land use activities into the calculation of costs and benefits. Where inefficiencies are found, the role of market, policy and institutional factors in determining individual land use decisions will be highlighted. Critical failures in this regard will be investigated in further detail and potential changes in existing incentives systems and/or the design of new incentives schemes proposed and evaluated.

The study evolves out of two related research themes. The first area concerns the applicability of economic analysis to the issue of *land use decision-making*. The second theme (an extension of the first) is the integration of *environmental functions* (in this case the watershed protection function of forested land) into the economic analysis of land use decision-making. Through the Arenal case study the project aims to demonstrate the utility of including economic factors into the land use planning process and the practical significance of including erosion and hydrological impacts into an economic analysis of incentives for watershed protection. The effort to link biophysical aspects of the problem with economic valuation of environmental impacts and, then, proceed to the identification and evaluation of practical solutions should provide an important example in this context. It is hoped that the study will add value to current efforts to coordinate land use activities in the Arenal area. It is also hoped that the study will provide methodological guidance to other agencies and organisations interested in evaluating similar cases in the region.

Following a brief summary of the project, two background sections describe the origins of the project regarding the attempt to apply the theories and tools of environmental economics to the analysis of land use decision-making and the economic evaluation of environmental functions, in particular, that of watershed protection. An outline of the problem of land use and watershed protection in the case study site in Arenal, Costa Rica is followed by an exposition of the conceptual framework underpinning the study.

Economics and Land Use Decision-Making

Actual land use patterns often do not correspond to assessments of potential land use capacity based on biophysical analysis; this leads to the designation of such lands as either 'under-used' or 'over-used.' In the case of 'over-use' the inference normally drawn is that environmental degradation is occurring. In other words unsustainable production patterns will erode the future productivity of the land in question (and, possibly that of adjacent areas) in the absence of a correction to the prevailing land use. Accordingly policy recommendations stemming from a biophysical determination of 'over-use' is simply that land use activities be 'reduced' in order to reflect more adequately potential land use capacity. This is viewed as necessary in order to ensure sustainability (of a biophysical nature) in the long-run.

However, viewed from an economic perspective a number of questions and criticisms can be levelled at this approach. First, assuming that society is trying to maximise economic welfare rather than optimise the match between actual land use and land use capacity, it is possible that certain activities that degrade the environment may prove to be rational (and even contribute to sustainable development) from an economic standpoint.[1] This is essentially the same as indicating that economic optima in terms of land use allocation are unlikely to be equivalent to such optima determined solely on biophysical grounds. Put simply, the inclusion of economic objectives, constraints and agents into the system under evaluation is likely to lead to the determination of a different optimum for the system.

For example determination of a biophysical optimum for land use is often arrived at through the use of land use classification or capability systems that combine a variety of biophysical data (such as soil type, elevation, slope, climate) to predict the most intensive type of land use that is 'sustainable.' Such systems often ignore important socio-economic variables (eg, proximity to market, local tastes) as well as failing to account for the economic trade-offs between different production and land use management systems.

Another oft-cited biophysical optimum for land use is the soil loss tolerance factor. In theory this factor sets the maximum permissible amount of soil erosion equal to the amount of soil formation. The slow rate of soil formation and practical difficulties with estimating these rates have led to the use of soil loss tolerances that are dubious at best (Morgan 1986). In any case, such tolerance levels completely ignore the dynamic nature of investment decisions regarding soil conservation. For example, if both farm output prices and the costs of soil conservation technology are expected to decline over time, it may be economically optimal for a land user to allow a greater than 'tolerance' level of erosion in the short- to medium-term and, at the same time, maintain the option to invest in soil conservation in the future. In other words, simple biophysical assessment of tolerances does not account for the fact that soil erosion is not an irreversible process, but is rather a matter of investment timing (ie, at what point is the cost of adding or conserving an additional marginal unit of soil or nutrients

[1]In other words the view taken here is that there is a degree of substitutability between different types of capital stock (e.g. natural, physical, human and financial) implying that sustainability does not necessarily mean the maintenance of a constant physical stock of natural capital (*see* Pearce, Barbier and Markandya (1990) for more on an economic definition of sustainability).

back to the soil outweighed by the benefit of doing so).

Thus, from an economic standpoint there are reasons to question the use of such biophysical optima as the last word on land use planning. This problem is exacerbated in most developing countries where the information and/or the methodology required to determine precise, site-specific optimum land uses may be lacking. For example, it is not unusual to find tropical countries using a land use classification system originally developed for temperate zones. This is not to say that it is easier, or less expensive, to derive land use optima based on economic criteria. Rather it is to make the point that land use prescriptions emerging from land use classification systems in developing countries are incomplete at best. This may in part explain the frequent finding that existing land uses represent an 'over-use' according to land use capacity maps.

A second limitation to biophysical assessments of land use capacity is that they may not adequately account for actual land use and inter-temporal degradation. In the first case, the failure to consider the difference between the actual use of the land and its capacity means that policy recommendations based on use capacity do not account for the costs of altering land use or the opportunity costs of doing so. For example, given an existing 'over-use' of the land the direct, transaction and opportunity costs of initiating and carrying out a change in land use may outweigh the net benefits generated by the more 'optimal' land use.

In the second case, if use capacity assessments fail to incorporate the effects of degradation over time, they may fail to account for the increasing time or money costs of moving back to the use dictated by biophysical assessment. This may occur if land use capacity is not regularly updated or if degradation is not adequately accounted for in the analysis of land use capacity. These costs may be substantial when the case refers to an indicated use of protection forestry as versus an actual use in pasture or agriculture. In such cases, attempts to push land use all the way towards the perceived biophysical optimum may fail to account for the trade-offs implicit in land use change and the potential for decreasing economic returns to investment in such activities as environmental degradation progresses. In other words, the more degradation occurs the more it will cost - in terms of up-front investment in ecosystem restoration or in reductions to future benefit streams resulting from delays in the restoration of ecosystem services - to convert to the more 'sustainable' land use.

For the above reasons, economists cannot take a biophysical indication of land 'over-use' as necessarily indicative of an *inefficient* use of land. It is likely that an economic optimum in such case will fall short (in terms of environmental conservation) of the biophysical optimum.[2]

Determination of an economically inefficient allocation of land resources would lead to policy recommendations having the same direction of change in resource use as that resulting from a biophysical determination of inappropriate land use. In the case of 'over-used' land, change is clearly needed to move the ecosystem back towards a more optimal use of resources. The difference between economic and biophysical approaches to land use decision-making is not

[2] See Bishop (1992) for a comparison of economic and biophysical positions on 'optimal' rates of soil erosion.

only that the 'optima' may be different, but that the economic approach is likely to provide better clues on how changes in management and land use can be initiated.

Clearly, a purely biophysical approach to land use decision-making is likely to be limited in its ability to prescribe solutions. Such an approach often leads to recommendations regarding desired land uses, management strategies and technologies. In order to arrive at these uses quantitative land use regulations (eg, zoning) or technology transfer mechanisms (through extension programmes) are often employed in an attempt to correct over-use. These programmes may suffer from their dependence on fairly rigid, technocratic, and top-down planning efforts, and the need for appropriation and expenditure of public funds. In addition, if such programmes do not take into account the farmer's self-interest, they may fail to deliver on their objectives. Financial incentives, eg, easy credit, subsidies, etc, may also be employed to reach a more desirable pattern of land use allocation. However, if such incentives are based merely on biophysical data and objectives - ie, they are unsupported by any underlying economic analysis or rationale indicating the preferred type or magnitude of such incentives - they may end up creating more distortions than they solve in the economy-environment relationship.

Aside from its potential role in assisting with the formulation of objectives for land use allocation, another advantage, then, to the integration of economic analysis into the process of land use planning is that additional information is provided to guide the selection of interventions that influence land use decision-making.[3] By analysing the economic incentives driving current land it is possible to widen, in an informed fashion, the number of available mechanisms for enabling individual land use decision-makers to make improved allocation decisions.[4] This can be done through the use of measures of private and social profitability (Gittinger 1982). For example, government policies such as input subsidies, cheap credit or export promotion programmes may exaggerate the profitability of farming activities relative to more 'conservation' oriented land uses. In turn these 'conservation' land uses may lower downstream sedimentation levels, thereby providing important regional or national positive externalities such as reducing damage to hydropower and irrigation schemes. In many cases, then, an understanding of why private actions seem to be in conflict with the public (or social) good is the first step towards the design of more effective policy.

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[3]Indeed, there are some 'environmental-economists' who feel that economics sole participation in policy and planning should be this role: the determination of 'efficient' means of reaching allocation and distributional goals previously specified on the basis of biophysical and social efforts respectively (Daly 1992).

[4]The popular term 'disincentives' is implicit in the use of the word 'incentives'.

Economic Evaluation of Environmental Functions

The second issue that the study will tackle is the incorporation of environmental functions into the economic evaluation of land use decision-making. Environmental functions, or 'ecosystem services' are not directly used in productive or consumptive activities, but rather support and protect these economic activities. Extension of neoclassical economic theory to include these environmental functions is a fairly recent phenomena (Barbier 1989). The database of applied research into the 'indirect' use value of the services provided by environmental functions in developing countries is limited (Aylward and Barbier 1992). Nor is it often connected to an effort to resolve the inefficiencies identified. As such, this project provides an opportunity to conduct not only applied economic research into the watershed protection function of tropical forests and its economic importance, but to link such studies with the identification of policies and instruments that may assist in moving land use in a more economically sustainable direction.

Valuation and Tropical Forests

For the purposes of this study the focus is on the services provided by tropical moist forests, in particular the watershed protection function. Tropical forests provide a range of goods and services to society (as shown in Table 1). These goods and services may be distinguished by the type of value they provide - direct or indirect use values or non-use values. They may also be distinguished in terms of the distribution of the benefits they produce. Economic benefits derived from tropical moist forests are captured at the local, national and global levels. Traditionally decisions regarding tropical forest land use have been made on the basis of major direct uses of forest land that generate tangible local and national benefits that pass through markets. Typically, this has meant timber extraction and the conversion of forest to agricultural or livestock uses.

In recent years increasing attention has been given to the important economic role non-market benefits - including those generated by environmental functions - may play in providing incentives for the conservation of tropical forests. Peters, Gentry and Mendelsohn (1989) used an analysis of the returns to non-timber forest products in Mishan, Peru to demonstrate that traditional extractive use of the Amazon was a superior economic use of the forest to plantation forestry and cattle ranching. The list of non-timber forest products is practically endless (Duke 1991) and a range of subsequent studies have attempted to show the economic importance of traditional uses of forest products in local livelihoods (Scoones, Melnyk and Pretty 1992). Frequently, however, public policies are set at the national level that ignore local use values and lead to perverse incentives to degrade and convert tropical forest lands to inferior uses (Browder 1985).

At the global level, research efforts have focused on the importance of tropical forests in providing habitat valuable for its ecotourism potential, ameliorating the effects of global climate change and providing a rich source of species for industrial research and development. While 'rainforest ecotourism' has yet to play as pervasive a role in the economics of forest ecosystems as it does in coastal ecosystems, a small number of studies have emphasized tourists' willingness to pay for visits to rainforest parks in developing countries. Studies of

Costa Rica's Monteverde Cloud Forest Reserve have demonstrated that use of the Reserve for ecotourism generates social benefits that exceed local alternative uses of the forest (Echeverría, Hanrahan and Solórzano (forthcoming); Tobias and Mendelsohn (1991)). The difficulty with rainforest ecotourism - and ecotourism in general - is that the benefits accrue to global, national and local actors, often failing to provide financial incentives to private or public park operators to maintain or upgrade their services.[5]

The threat of damage to the international economy from the accumulation of greenhouse gases in the atmosphere and subsequent changes in climatic conditions has led economists to devote considerable energy to estimating the magnitude of these effects. Van Kooten *et al.* (1992) report on a number of studies suggesting carbon sequestration values of from $2 to $275 per ton of carbon. Fankhauser (1993) has suggested a central value of $20 per ton of carbon. Following on Pearce (1990) this would imply that the damage done by deforesting a hectare of tropical forests would be worth $2,000. At this level the value of tropical forests for carbon storage is likely to outweigh the benefits of most alternative land uses.[6] The agreement on climate change signed at the Rio Conference in 1992 involved signatories committing their countries to a reduction of emissions to 1990 levels by 2000. Currently - through the mechanism of joint implementation agreements - we are seeing the first opportunity to capture the value of carbon offsets through this artificial market. Such progress notwithstanding, scientific uncertainty over the direction and magnitude of the long-run effects of climate change continues to bedevil efforts to achieve widespread political agreement on a comprehensive action plan in this field.

The economic value of biodiversity - of which a good share resides in tropical forest ecosystems - for use in the derivation of new industrial products is often cited as an important global rationale for conserving tropical forests. Aylward (1993) reviews studies of the value of biodiversity in a particular industrial area - as an input in the pharmaceutical R&D process - and finds a range of values between $15.00 per species to $24 million per species (on an annual basis). Recent calculations from Aylward (1993) indicate that not only is the biodiversity value likely to be at the lower end of this range, but that the capturable value of biodiversity from industrial sources is small relative to the costs of conservation. Practical initiatives, such as revenue-sharing, that provide incentives for conservation activities are more advanced in the case of biodiversity prospecting than in the case of the other global values of tropical forests, yet their impact is likely to be marginal on the overall level of forest conservation.

An important environmental service that is also often cited as an economic justification of conservation activities is the watershed protection function provided by tropical forest. Watershed protection, it is commonly held, conserves soil and water, thereby yielding local and national benefits. For example, in Ruitenbeek's (1989) valuation of the Korup Project in

[5]Note that this is not always the rule. For example, the Monteverde Cloud Forest Preserve is largely self-sustaining based on tourism revenues. For more on profitability of private parks *see* Langholz (1994).

[6]A subtle distinction that may prove important in this regard is that sequestration - i.e. permanent 'putting away' of carbon - is different than the more temporary 'storage' of carbon in forests. This is particularly the case for plantations where a good portion of the carbon stored will be liberated at harvest. As a result carbon storage may not provide the same level of value in the near term as do true carbon 'sinks.'.

Cameroon, the benefits from watershed protection were estimated to be almost half of the direct conservation benefits. Nonetheless, watershed degradation continues to be widespread in developing countries. This is, in part, because establishing incentive systems that solve market, policy and institutional failures that hamper watershed protection efforts remains a vexing problem for policy-makers, scientists and communities in those countries.

Progress is being made on resolving the difficulties inherent in ensuring that these less recognised values of tropical forests are incorporated into the decision-making process. Whilst the 'global' values are the most in vogue at the moment their potential impact on the conservation of tropical forests may be limited. Only so many areas in a country are likely to receive high tourist visitation rates and the number of days the average tourist is likely to spend 'in the forest' is also circumscribed. Biodiversity prospecting has a long way to go to demonstrate it can make a significant contribution even in isolated cases and countries. Policy initiatives to 'capture' carbon storage benefits appear promising, but remain in their infancy and are likely carry a high degree of political risk.

Indeed, it may be that recent preoccupation with capturing these 'global values' have diverted attention (and funds) away from locally and nationally motivated actions and policies that would promote sustainable management of tropical forest lands. Such efforts, for example, might be undertaken with the express purpose of supporting local livelihoods and providing watershed protection to important productive activities. Finding and achieving local and national solutions to the problems at hand will not be painless - therefore the enthusiasm at the prospect of capturing outside funds to avoid difficult choices. However, it can be argued that it would be preferable for countries and communities to first eliminate the perverse incentives leading to degradation of soil and water resources and then consider which global values can be captured to add additional impetus to conservation activities. An internally motivated step towards reversal of resource degradation may ultimately prove more sustainable than one based, in the first instance, on expectations regarding inflows of foreign conservation funding.

The Economic Value of Watershed Protection

The biophysical basis for providing incentives for watershed protection are well known, although there remain areas in which the impacts of land use change are not yet fully understood. Probably the most visible impact of watershed degradation linked to poor land use allocation in developing countries has to do with water quality, specifically sedimentation of streams and rivers. Watershed protection stabilises soil, thus preventing surface soil erosion, gully erosion and mass soil movement (Gregerson *et al.* 1987). Soil degradation - due to changes in land use practices and changes in forest cover - rapidly, and often irreversibly ,degrades the productive value of forest land. In addition, productive activities downstream, such as hydropower facilities, irrigation projects and fisheries, are often damaged by sedimentation resulting from upstream changes in land uses.

A number of methodologies for evaluating the on- and off-site costs of soil degradation exist (Bishop 1992; Easter, Dixon and Hufschmidt, 1986; Gregerson *et al.*, 1987). Techniques based on market prices, land values (hedonic pricing), productivity effects, replacement costs, and relocation costs may all be employed in estimating the on- and off-site costs of soil degradation. Much of the empirical work on on-site costs concerns temperate sites, eg in the

USA, or African ecosystems such as Zimbabwe (Stocking 1986), Mali (Bishop and Allen 1989) and Malawi (Bishop 1990). Only a small number of researchers have studied the on-site costs of soil degradation in the context of deforestation of tropical moist forest ecosystems (Cruz, Francisco and Conway 1988; Magrath and Arens 1989; Veloz *et al.* 1985). A number of studies have estimated the off-site costs of soil degradation in tropical moist forest environments (Cruz, Francisco and Conway 1988; Hodgson and Dixon 1988; Magrath and Arens 1989; Ruitenbeek 1989).

The biophysical impacts of conversion of tropical forests on water yield are less well understood. It is generally agreed that total water yield in streams is inversely related to the level of vegetative cover. A review of 94 catchment studies by Bosch and Hewlett (1992) indicated a very consistent linkage between deforestation and increased run-off, and reforestation and decreased run-off. In the case of reduction of deciduous hardwood or scrub a decrease in watershed forest cover of 10 percent is correlated with an increase in annual flow of 10-25 mm. Most of these studies, however, dealt with the impacts of logging and not with conversion to alternative uses. Nonetheless, it is fairly well established that evaporation (interception) is higher in forests as opposed to grasslands, and that in dry conditions transpiration from forests exceed that of grasslands due to deep-rooting of trees. While increased run-off following deforestation does increase the risk of local flooding, major flood events are often more closely linked to major climatic and geomorphic events than to land use per se (Hamilton and Pearce 1986).

These factors also explain why it would normally be expected that average dry season flows will increase following removal of forest cover. Hamilton and Pearce (1986) suggest that the common conception of the forest as a 'sponge' - soaking up water in the rainy season only to release it during the dry season - has no scientific basis and is in fact contrary to reality. Nevertheless, two exceptions to the impact of deforestation on dry season flow exist. First, forest conversion and subsequent uses may lead to increased soil compaction and surface run-off (as opposed to infiltration) during rainfall events. This can cause a reduction in the recharge of the soil reservoir and consequently a reduction in base flows during the dry season. The second exception is in the case of fog or cloud forests where the forest actually 'captures' additional precipitation. In such cases, horizontal precipitation may make an important contribution to dry season flows in forested areas (Calder no date).

According to Gregerson *et al.* (1987), valuation of changes in water quality and quantity effects is possible using much the same types of valuation techniques as employed in valuing soil degradation. However, there exists little to no empirical work estimating the economic benefits of water conservation benefits - both in terms of total flow and flow variability - generated by tropical watershed protection. Ruitenbeek (1989) provides some rough estimates of flood control benefits to be generated by the Korup Project in Cameroon; however no empirical verification of the physical or economic relationships and data utilised in the study is provided.

In sum, maintenance of tropical forest cover conserves both water and soil within the forest ecosystem. Soil conservation means maintaining on-site productivity of forest lands and preventing damage to off-site economic activities. Water conservation implies that the forest appropriates a share of the water in the hydrological cycle for its own use in transpiration whilst returning an important portion back to the hydrological cycle through interception and evaporation. Thus, the soil and water conservation functions of tropical watersheds are intertwined, ie there may be trade-offs between different aspects of the two functions at varying levels of provision of forest cover.

Aylward and Barbier (1992) have suggested that consideration of trade-offs between environmental functions will be an important element in the economic analysis of these services. Consideration of the economic value of watershed protection should, therefore, examine the interrelationships between the soil and water conservation components of the watershed protection function. Such an analysis must incorporate the contribution of water and soil conservation to the production of other economic goods and services - the timber, non-timber, ecotourism, carbon sequestration and biodiversity benefits referred to earlier. Thus, the analysis of trade-offs between soil and water conservation activities must acknowledge their respective marginal roles in the production of these economic goods and services. Clearly, the nature, extent and complexity of these tradeoffs will vary from one forest ecosystem to the next, depending on biophysical, social and economic conditions.

Economic Incentives for Watershed Protection

Maintaining the watershed protection function may entail cessation of all but the least harmful extractive activities, as well as ensuring careful conservation practices and investments are in place. Theory suggests that, in the absence of market, policy or institutional failures, landowners would be inclined to adopt soil conservation practices that lessen the impact of the on-site costs that result from soil degradation. However, where conservation activities influencing the level of off-site costs from soil degradation and water quantity/quality effects are concerned, there is little incentive for the private land owner to provide proper watershed management. Thus, economic analysis of the incentives for watershed protection must consider the factors that influence the decisions to internalise the user cost and external cost of land use practices.

In some regions, the existence of government subsidies for agricultural development may be directly influencing private landowners to cultivate on steep slopes, thus precipitating watershed degradation (Barbier, 1989, ch 4). Public ownership of forest lands as 'catchment' reserves is often decreed in an effort to ensure the integrity of critical watershed areas. However, the level of protection afforded to such reserves is often minimal due to budget constraints. Reserving land simply for the purposes of watershed protection is seen to be a luxury for some developing regions where arable land is scarce, population pressure is increasing and agricultural productivity remains low. In other countries institutional factors such as tenure systems and the prospect of compensation for land claims may lead settlers to invade virgin forest areas.

In considering potential solutions to the difficulties encountered in providing an 'optimal' social level of watershed protection it is important to include the full range of possible management regimes:

- private 'for profit' ownership of land
- private management by 'non-profit' organizations

- public ownership of land - ie. setting aside of protected areas

- mixed private/public system, ie private ownership with market-based incentives such as taxes, subsidies etc, that are designed either:
  - to promote water and soil conservation, or
  - to ameliorate perverse incentives for watershed degradation

- common property management by collective groups of watershed residents or land-owners

Management of a given watershed may clearly involve some combination of the above regimes.

No matter which of the above regimes is in place, the existence of some form of financial 'compensation,' or resource transfer, in return for the off-site benefits provided by watershed protection would greatly increase the likelihood of success. If the indirect benefits provided by the environmental function of watersheds could be directly marketed or otherwise exchanged in return for financial compensation, a clear incentive for the maintenance of watersheds would be developed. In particular, this would be the case if the opportunity cost of 'idling' land were not too high (ie poverty and population pressure coupled with land scarcity were not driving watershed degradation out of short-term subsistence needs) and if the majority of the benefits of watershed protection accrued 'off-site' in terms of affecting water supplies and quality by downstream users.

Under these conditions, it may be possible to devise incentive systems whereby downstream stakeholders provide financial compensation to those undertaking watershed protection or conservation activities in the upper watershed:

- 'polluter pays' - ie government action to penalise (eg, through taxes) upstream users

- contractual arrangements - ie agreement between watershed managers and downstream stakeholders

- development of property rights - ie national legislation of right to receive market compensation for generating environmental functions (water conservation) or their end products (water)

- development of marketable permits and transferable development rights - ie establishment of permissible local sedimentation or water yield limits (a hydrological 'bubble'), allocation of 'pollution' permits and development of a market in the permits.

There are several problems facing any proposed incentive scheme. First, a significant moral problem may exist in defining the baseline for some of the above schemes. Expectations regarding the onset of incentive schemes may lead landholders to 'pollute' more in the short-term in order to obtain a larger allocation of pollution permits or to strike a larger contract. Second, negotiation and agreement are never costless, particularly when unidirectional environmental externalities are involved. Recent economic evidence on international environmental agreements suggests that successful conclusion of agreements involving financial compensation depends substantially on the number and diversity of economic agents involved, the existence of conditions encouraging agents to free ride on agreements, and the degree to which agents face different costs of abating any harmful activities or of investing in conservation activities (Barrett, 1990).

Similar problems are likely to be faced by any local or national level incentive scheme for watershed protection management. A scheme is likely to be more successfully established if the watershed conservation activities are being undertaken by a single entity, such as a forestry commission or a large landowner, who in turn seeks compensation from a single downstream entity, such as a utility company owning a reservoir or a local government agency. On the other hand, it may be possible to 'pool' many upstream agents into a single representative body for negotiation purposes eg, the local farmers' association or cooperative on behalf of watershed farmers undertaking conservation. Many downstream users, such as municipalities, farmers' groups, etc, may also be collectively approached for negotiating and extracting compensation. However, if the costs of compliance are too high or the returns to compensation too low for many of the agents involved in the scheme, then the incentives for participation may break down, and those not participating may seek to 'free ride' on the scheme as much as possible.

Land Use Issues and Watershed Protection in Arenal, Costa Rica

In order to examine incentive issues of land use decision-making related to the economic evaluation of the watershed protection function, the project will undertake a case study in the Arenal watershed of Costa Rica. In this section we present a brief overview of the biophysical characteristics of the Arenal area, review the history of land use development in the watershed and summarise the incentive issues that the study will address.[7]

The watershed surrounding Lake Arenal is located in northern Costa Rica, lying just on the Atlantic side of the Guanacaste and Tilarán ranges that form the continental divide between the Atlantic and Pacific zones of the country. For the purposes of this study the watershed can be divided into (1) an upper watershed area comprising three micro-watersheds Río Chiquito, Caño Negro and Aguas Gatas; (2) the remaining micro-watersheds on the northern and western sides of the lake, and (3) the lake itself.

Surface area and respective slope classes of these areas are shown in Table 2. The table amply demonstrates the abrupt relief present in the watershed. Note that the terrain in the upper watershed area is proportionately steeper than that of the remainder of the watershed. Fully 90 percent of the upper watershed has a slope of greater than 25 percent, whereas the corresponding figure for the remaining area of the watershed is 67 percent. Slope is one of the principal factors in determining land use capacity from a biophysical standpoint. In particular, slopes of the severity found in Arenal are likely to play an important role in erosion, streamflow and sedimentation. In other words, the Arenal watershed - in particular the upper watershed area - is often considered to be of great importance in terms of its watershed protection value (CCT 1980; ACA 1993).

Other important biophysical factors that characterise the Arenal and upper watershed areas include soils and climate. The majority of the soils in the Arenal watershed are deep, sandy soils of volcanic origin possessing good natural drainage and of low fertility. In terms of climate, eight of the twelve Holdridge Life Zones are found in the watershed, with the predominant zone being premontane wet forest. Rainfall varies between 2,000 and 6,000 mm per year with R. Chiquito reporting average rainfall of 2,400 mm compared with 4,000 mm per year in the adjacent watershed of Caño Negro. Average annual maximum temperatures are around 28°C, with mean minimum temperatures of 19°C. Annual average relative humidity is about 80 percent. Wind is an important climatic factor at the northern end of the watershed where there is a natural saddle between the Pacific and Atlantic zones and on the exposed ridges of the upper watershed areas.

Data on land use capacity for the Arenal watershed (Table 3) suggests that the land in the region is primarily suited to production forestry and protection forestry. In the Arenal watershed none of the land is considered suitable for annual cultivation. The diagnosis is similar for perennial cultivation and pasture. For example, in the upper watershed area only

[7]For more detailed reports on biophysical conditions in the area see CCT (1980) and ACA (1993). The biophysical data in this section comes from these sources. For an overview of the Arenal area and its economy *see* Echeverría *et al.* (1994).

1 percent of the area is deemed suitable for either perennial cultivation or pasture. Meanwhile, 19 percent of the land is indicated for use in production forestry and 80 percent for complete protection - ie. no 'productive' use or at least no activities involving alteration of the forest.

The history of local land use in the Arenal area is clearly contrary to such recommendations. Kauck and Tosi (1989) identify four important events in the history of land use in Arenal. From 1880 to the 1950s, demographic pressure in Costa Rica's Central Valley - the site of most of the country's population and productive activities - led to the colonisation of lands in other regions of the country. Increasing production of coffee in the Central Valley meant the displacement of many other agricultural and livestock activities to the hinterlands - such as the Guanacaste region and Arenal.

The second event of consequence for the Arenal area was the major commercial expansion of cattle ranching from the 1930s to the 1970s. Low up-front costs, plenty of available (though forested) land and the ease of running cattle in areas without roads and other infrastructure made ranching a practical choice of land use to immigrants in frontier areas such as Arenal. The growth of the national market, technological improvements (including importation of improved breeds) and public policies favouring the development of the sector spurred stocking levels to ever higher numbers during this period. With the rise in export markets from the 1950s, Costa Rica entered a "Golden Era" of meat production. By the 1970s Costa Rica was the fourth largest exporter of meat to the United States.

The third and fourth events occurred in the late 1970s with the construction of the Arenal Hydroelectric Project and the creation of the Arenal Forest Reserve (later renamed the Arenal-Monteverde Protected Zone). In 1979 the Sangregado dam was completed with an operational level of 546 meters above sea level. The result was a 34 meter rise in the level of the old Arenal Lake and the diversion of water from the new Arenal Lake watershed over to the Pacific side of the continental divide. The discharge from the lake - roughly 100 cubic meters/second - is now used by three different power plants with a total generating capacity of 362 megawatts - 33 percent of Costa Rica's total capacity and up to 70 percent of its dry season capacity. The water is then passed on to the Arenal-Tempisque Irrigation Project where it currently services approximately 6,000 hectares of agricultural land. An estimated 54,000 hectares remains to be added to the total of irrigated land in subsequent phases of the project.

Displacement of local towns and ranching activities on to the higher and steeper slopes of the upper areas of the watershed accompanied the building of the dam. The declaration of the Arenal-Monteverde Protected Zone - which included most of the land not yet occupied in the Caño Negro and Aguas Gatas micro-watersheds introduced additional uncertainty in local land markets. Kauck and Tosi (1989) suggest that the principal effect of these two events - which persisted into the 1980s - was to promote land colonisation, registration and speculation in efforts to capitalise on the prospect of government land purchases in the protected zone.

In the 1980s, however, little land purchase took place in the upper watershed area - at least not by the government. Neither MIRENEM, the ministry in charge of parks and forestry, nor ICE, the semi-autonomous institution in charge of the hydroelectric scheme, purchased land (except lake-side property obtained by ICE). This, despite suggestions made by the Tropical

Science Center in 1980 (as part of its environmental study of the hydropower project) that the government should create a national park in the area in order to protect the hydrological resources of the area in perpetuity.

Notwithstanding, the Arenal area has recently seen an increase in land transactions for both conservation and tourism. In the late 1980s and early 1990s the Monteverde Conservation League, a local non-profit conservation organisation, began purchasing land in the Caño Negro and Aguas Gatas micro-watersheds forest area. Funds for land purchase have come mainly from foreigners and interested conservation organisations. In addition, Costa Rica's popularity as an ecotourism destination for foreigners has led to additional land speculation, though in the first instance, this has occurred outside of the upper watershed area, ie, in lake-side areas where all-weather roads exist. Local tourist attractions include the majestic Arenal Volcano to the southern end of the lake, fishing on the lake and other water sports at the northern end of the lake.

In October of 1994, MIRENEM and the Arenal Conservation Area (ACA) signed a declaration establishing the Arenal National Park.[8] In addition to the Arenal Forest Reserve which protects land around the Arenal volcano, the park incorporates 12,000 hectares at the lower fringes of the Arenal-Monteverde Protected Zone in the Aguas Gatas and Caño Negro micro-watersheds, as well as additional land adjacent to the volcano. Of this area ACA had purchased roughly half before making the declaration, leaving another 6,000 hectares in private hands (*La Nación* 1994).

The upper watershed area of Lake Arenal makes an interesting case study of the incentive issues connected with the watershed protection function of tropical forests. Together these three watersheds comprise 46 percent of the land area in the Arenal watershed and provide 48 percent of the water (CCT 1980). Measurement stations operated by ICE are situated at the mouths of these micro-watersheds and provide hydrological, sedimentation and meteorological data which may be used in the assessment of off-site costs and benefits of watershed protection. While livestock and agricultural uses in the remainder of the watershed have resulted in the conversion of almost all primary forest, these three micro-watersheds have large tracts of both pasture land and primary forest. As shown in Table 4, the Río Chiquito basin is largely deforested and under pasture for dairy farming and ranching (63 percent in 1992). The other two watersheds retain a large portion of land in primary forest - land used for production and protection forestry (roughly 80 percent in 1992). The comparison of land use data from 1980 and 1992 (shown in Table 4) reveals that during this period continued conversion of forest to pasture occurred in all three micro-watersheds. Most notably in the Aguas Gatas micro-watershed - the amount of protected primary forest fell from 87 percent to 63 percent. These numbers indicate that in Arenal, unlike in other areas in Costa Rica and indeed in Central America, pasture lands are not being abandoned.

In sum, studies of land use capacity and actual land use in the upper Arenal watershed

[8]ACA is one of the new Costa Rican conservation areas. It includes 204,000 hectares of land, including the Arenal watershed and adjoining lands. The composition of ACA is an amalgamation of different 'protected' areas (39 percent) and 'non-protected', or buffer zone, lands (61 percent). ACA also refers to an administrative body that is coordinating efforts to achieve sustainable development in the area with the help of the Canadian government and World Wildlife Fund - Canada.

indicate that a significant proportion of the watershed continues to be overused according to biophysical criteria. Principally, land that should be held as protection forest (according to its land use capability) is dedicated to pasture for dairy farming and beef ranching. Lake Arenal, meanwhile, sustains the country's largest hydroelectric facility and irrigation scheme. Over the years this situation has led to repeated suggestions from environmentalists that the benefits that would be provided by protecting the watershed would far outweigh the benefits generated by current land use practices. As a result, so the story goes, the initiation of a system of economic incentives for encouraging more appropriate land use in the area would be a win-win situation.

The principal objective of the project is to assess how changes in existing incentives can lead to the incorporation of watershed protection values - those received by both farmers and society-at-large - into land use decision-making. In this regard the upper watershed area provides a suitable site for evaluating two particular incentives issues. The first has to do with the issue of what, if anything, should be done about the use of land for pasture, particularly in R. Chiquito. More specifically, will dairy farming and ranching prove inefficient from an economic standpoint and, if so, what are the incentive issues that need to be addressed to ensure that private land use incentives produce a socially desirable pattern of land use. Second, given that there remain significant areas of primary forest under private control - in this case the lands purchased by the Monteverde Conservation League - how can the benefits they provide to society in the form of watershed protection be captured for use in funding operational activities?

The study will investigate the valuation and incentives issues associated with land currently under pasture. Nonetheless, the information developed by this study should provide an ample database for future work on the incentives issues linked to maintenance of existing primary forest in the Arenal area.

Conceptual Framework

In this section, the conceptual framework for isolating incentives issues critical to land use decision-making in the study area is developed. The framework presented below represents the flow of such an analysis for the case of land dedicated to pasture. Although it would involve scrutiny of a different set of benefit and cost streams and, hence, incentives issues, the analysis of the decisions facing private land-holders engaged in the protection of primary forest would proceed in much the same fashion.

In the discussion that follows we assume that the land-owner is the land use decision-maker. For convenience the term 'rancher' is used to refer to the land use decision-maker who may be either a rancher or a dairy farmer, or engaged in a combination of both activities. Finally, and again for convenience, we ignore the possibility of transfer of land-ownership to the state, an NGO or other producer, in the discussion of incentives, although clearly such possibilities will be taken into account in the study itself.

Phase 1: Private incentives issues based on actual private returns.

The first step is to determine the benefits and costs of ranching (B^R, C^R) from the private perspective, that is, how ranchers view them. This gives us the private returns to ranching:

$$NB_p = B^R - C^R$$

If $NB_p < 0$ we have a bit of a paradox as ranchers are still engaging in ranching despite it being a money-losing proposition. This might point to failures to accurately reflect ranchers' expectations about the future direction of important economic and institutional variables (ie, discount rates, land tenure, taxes and subsidies, the international price of beef) as well as the future likelihood of additional land purchase by conservation organisations. With these expectations taken on board NB_p would presumably become positive.

If $NB_p > 0$ we move to Phase 2

Phase 2: Private incentives issues based on hypothetical private returns - ie. without policy distortions and market imperfections but excluding off-site externalities

Next, we determine the net benefits of ranching from the private perspective, given the removal of any existing policy or institutional distortions that factor into private decision-making (ie. not including off-site environmental impacts). This phase involves two distinct steps: (a) standard project evaluation types of adjustments for distortions of input and output prices and (b) more 'environmental' adjustments to account for land-related institutional or policy distortions.

<u>Phase 2a. Removal of Policy Distortions.</u> First, benefits and costs as currently perceived by ranchers are adjusted to account for market distortions introduced by policies that drive a wedge between actual and 'economic' prices. This transformation of actual prices to 'economic' or shadow prices, B^{R*} and C^{R*}, is likely to involve examination of any distortions

in beef or milk prices, the pricing of labour, and any taxes/subsidies on other inputs. The result is the *standard economic* assessment of net benefits in the absence of such distortions:

$$NB_{SE} = B^{R*} - C^{R*}$$

If $NB_{SE} < 0$ then allocation of private resources to ranching is maintained only as a result of the distortionary policies identified. The incentives issue then becomes assessing the means of removing the distortions currently in place. If the distortions can be removed, the implication is that ranchers will turn their land over to the next best use.

As a result, the study may then turn to a comparison of the potential value of alternative land uses such as protection forestry and production forestry. These alternatives could incorporate alternative scenarios for restoring forest cover, such as natural regeneration, enrichment and plantation activities. The study would then need to investigate the private and social returns of these uses and any incentive problems and solutions involved in order to reach the more economically preferable alternative.[9] At this point it would be important to consider not only the transaction costs of initiating land use change, but the actual nature of the benefits and costs incurred by replacing pasture with forest cover.[10]

If such first best solutions to achieving private efficiency through the removal of existing distortions are not feasible, we may choose to continue on to Phase 2b in the search for second-best alternatives.

In any case, should the analysis demonstrate that $NB_{SE} > 0$, then ranching remains a profitable activity following the removal of distortions we move immediately to Phase 2b.

Phase 2b. Removal of on-site Market Imperfections. In this part, we assess the land-related policy and institutional failures that may be inducing managers to ignore important land and environment related impacts of their land use and, thereby, fail to achieve private efficiency. There are two likely processes that may be driving ranchers to make less than perfect decisions in this regard: (1) failure in the asset markets for land and (2) failure to properly account for the on-site, or user, cost of soil erosion.

In the first case, institutional problems in land markets may be giving land owners unrealistic expectations regarding future land prices or may be giving owners the wrong signals with regards to how their current land use effects the asset value of their land. Note that if land purchase price expectations and tenure insecurity are distorting land markets, we may need to jump ahead and calculate the user cost of soil erosion. This would enable us to develop a measure of the opportunity cost of land under ranching to compare with actual (and

[9] Note that we may then be faced by the problem of incentives for internalizing off-site benefits under an absolute protection scenario - which would be identical to the analysis of incentives for the problem faced by non-governmental conservation organizations trying to manage forested land providing watershed protection benefits.

[10] It is must be born in mind that once policy distortions are removed in the analysis of ranching a level-playing field must be maintained henceforth. That is, subsequent analysis of different land uses must be conducted as if these distortions did not exist.

potentially distorted) land prices. If such problems are of sufficient severity to drive returns into the red, then the study will need to consider how to solve these incentives problems and to consider subsequent land uses and their incentives issues (as above).

Another more specific aspect of these land-related problems is that of the user cost of soil erosion (UC) - ie on-site productivity losses incurred by ranchers. This can be calculated by means of an erosion/yield relationship based on primary biophysical and economic data.

If UC » 0 then the on-site impacts of soil loss on productivity are not important and, thus, soil management is not an issue from the farmer's perspective. We thus jump to Phase 3 and a consideration of the off-site impacts of soil erosion from ranching.

If UC > 0 then the impacts of soil loss on productivity are significant and we must include this environmental cost into the economic evaluation of ranching profitability and assess farmer's responses to these user costs. In other words, we need to see what the returns to farmers would be with user costs included:

$$NB_{UC} = B^{R^*} - C^{R^*} - UC$$

If $NB_{UC} < 0$ we must address the question of whether ranchers are taking user costs into account.

If the answer is yes, then we would expect UC to be at its optimal level. In other words there would likely be no soil conservation technologies that generate net incremental returns (ie. that increase C^{R^*} but not by as much as the ensuing decrease in UC). At this point (given that NB_{UC} is negative) we would need to return again to the general policy distortions involved in Phase 2a as it is these policy failures that are actually driving continued ranching activities (once the negative impacts of user costs perceived by ranchers are incorporated into the analysis).

If farmers are not taking productivity losses due to soil erosion into account they will be unlikely to be properly managing their soil assets properly (from their perspective). In such a case it may be that there are conservation technologies that, if employed, would generate net returns. Should these net returns be sufficient to make NB_{UC} positive we would need to focus not only on the distortions - such as tenancy issues, lack of information, etc. - that may be preventing farmers from incorporating user costs into their decisions, but assess any incentive measures necessary to ensure adoption of these 'ranching' conservation technologies. If no profitable 'ranching' conservation technologies exist then emphasis on dealing simply with the distortions is called for in order to aid farmers in coming to the realization that they would in fact be better off not ranching.

Again, if incentives issues can be solved we could then move to an analysis of subsequent land uses and related incentive issues. If the incentives problems cannot be solved - or in fact if the solutions have prohibitively high transaction costs - we may then turn to Phase 3.

If $NB_{UC} > 0$ then ranching remains economic even when on-site impacts of erosion are included. At this point we may (as above) want to consider if farmers are incorporating these impacts into their decision-making process and whether there exist any soil conservation technologies that would generate net incremental returns. This may be useful because any improvements of such a type will not only raise NB_{UC}, but will also be likely to lower the impact of the external costs of soil erosion quantified in the next step.

Phase 3: Societal incentives issues based on external costs

For the purposes of this project this phase involves both quantitative and qualitative evaluation of potential market failures - in particular externalities.

On the quantitative side, estimates of the off-site, or external, costs associated with the watershed protection function need to be developed and included into the analysis of returns from society's perspective. This is necessary because the market provides no incentive to ranchers to incorporate off-site impacts of their land use into their own decision-making framework. This can be accomplished by modeling how land use change leads to changes in water and sedimentation fluxes that affect dam storage capacity, with knock-on effects on hydroelectric generation and the availability of water for irrigation. As a result we include the external costs, EC, into our analysis of net benefits from society's perspective:

$$NB_S = B^{R*} - C^{R*} - UC - EC$$

If $NB_S < 0$ then the market failure that precludes the internalization of these external effects into the farmer's land use decision framework is causing net economic losses to society. Incentives for watershed protection are needed to move land use towards alternatives that provide positive rates of overall economic return. Note that at this stage we may want to verify that there are no improvements in ranching technology that through soil conservation would have a major impact on social returns. This is doubly important because any reduction in on-site erosion is likely to also reduce off-site sedimentation amounts. Failing possibilities in this regard, we may then proceed to assess incentives required for moving to alternative land uses such as those specified above.

If $NB_S > 0$ then current use of land for ranching remains efficient from an economic perspective, even once the external impacts of watershed protection are included in the analysis. If this is the case we may have demonstrated that an economic approach to land use capability analysis produces different recommendations to that generated through a purely biophysical approach.

However, this presumes that the full environmental and social impacts of ranching have been incorporated. We would need therefore to speculate the nature and magnitude of other environmental values, such as carbon and biodiversity values, that may be diminished by keeping land in pasture as opposed to primary forest. In addition, we may need to consider the impact of any observed differences in the private and social rate of time preference. If the addition of any, or all, of these factors is likely to change calculations of social profitability, then the above conclusion might well be premature. Further research on such impacts and evaluation of available incentive mechanisms would then be needed in such areas.

Conclusion

This paper reviews how economic information can be included into the land use planning framework and, in particular, how information on the economic value of environmental functions can be important in such a framework. In the case of watershed protection it is argued that it is important to include both the on-site impacts on farm productivity as well as the off-site impacts on economic activity into the analysis of 'optimal' land use. Such information is also useful in assessing the role of different stakeholders and the potential for different incentive schemes for promoting more efficient watershed management. The paper then goes on to a brief review of land use issues in the Arenal Watershed in Costa Rica. The paper closes by presenting the conceptual framework that will be used to analyse incentives for watershed protection in the upper area of this watershed.

Land use allocation will probably never be 'optimal' as defined by a top-down biophysical or economic approach based on extensive research studies.[11] Be that as it may, this study is attempting to demonstrate that the addition of economic evaluation - in particular that of the environmental services provided by the watershed protection function - can play an important role in providing information that can lead stakeholders to consider the benefits of changing existing land use allocations. However, only by a careful, step-by-step evaluation of the costs and benefits, and their distribution across members of society, can this sort of analysis expect to capture the richness of the relationships between biophysical and economic impacts and the incentive systems governing land use.

[11]Examples of participatory approaches to watershed issues are available from a collaborative research project entitled *New Horizons* being conducted by IIED's Sustainable Agriculture Programme and research partners in developing countries.

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Table 1 The Total Economic Value of Tropical Forests

Use Values*		Non-Use Values
Direct Uses	**Indirect Uses**	**Existence Values**
Extractive	Nutrient Cycling	Endangered Species
Timber	Watershed Protection	Charismatic Species
Rattan	Air Pollution Reduction	Threatened Habitats
Bamboo	Microclimatic Function	Cherished Landscapes
Medicinal Plants	Carbon Store	
Biochemical Resources		
Genetic Resources		
Other NTFPS		
Non-Extractive		
Ecotourism		
Recreation		
Scientific/Research		
Education		

Source: Based in part on Pearce (1990)

Table 2: Arenal Watershed - Area and Slope Categories

		Slope Categories						
	Class	1	2	3	4	5	6	Total Area
	Slope	0-6 %	6-12%	12-25%	25-50%	50-75%	+75%	Hectares
			(figures represent percent of land area)					
1.	Upper Watershed Area	1.4	1.3	7.4	41	34	15	19108
	a. R. Chiquito	1.9	0.9	9.3	40	35	13	9,136
	b. R. Caño Negro	1.3	2.0	5.4	38	34	20	7,248
	c. R. Aguas Gatas	0.0	1.2	5.8	54	31	8	2,724
2.	Remaining Land Area	3.2	6.1	23	43	21	3.3	22,226
3.	Subtotal - Land Area	2.4	3.9	16	42	27	9	41,334
4.	Lake Area	-	-	-	-	-	-	9,304
5.	Total - Arenal Watershed							50,638

Notes: Lake area at 550 feet above sea level - ie full

Source: CCT (1980)

Table 3: Arenal Watershed - Land Use Capability

		Land Use Capability Class				
	Land Area	Perennial Cultivation %	Pasture %	Production Forestry %	Protection Forestry %	Total Hectares
1.	Upper Watershed Area	0.5	0.5	19	80	19,108
	a. R. Chiquito	1.1	0.0	23	76	9,136
	b. R. Caño Negro	0.0	1.3	17	81	7,248
	c. R. Aguas Gatas	0.0	0.0	10	90	2,724
2.	Remaining Area	3.3	2.3	55	40	22,225
3.	**Total**	2.0	1.5	38	58	41,334

Note: Zero percent of the land is classified as suitable for annual cultivation.

Source: CCT (1980)

Table 4: Upper Arenal Watershed - Actual Land Use 1977 and 1992

Land Area	Annual Cultivation %	Perennial Cultivation %	Pasture %	Production Forestry %	Protection Forestry %
	Land Use Class				
1. 1977	0.1	0.0	35	12	53
a. R. Chiquito	0.2	0.1	57	20	22
b. R. Caño Negro	0.0	0.0	15	7	78
c. R. Aguas Gatas	0.0	0.0	13	0.6	87
2. 1992	0.6	0.0	41	12	46
a. R. Chiquito	0.3	0.0	63	19	18
b. R. Caño Negro	0.8	0.0	19	2	79
c. R. Aguas Gatas	1.2	0.0	22	14	63

Source: 1977 data from CCT (1980); 1992 data from a preliminary assessment from maps accompanying ACA (1993)

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