Market and Policy Incentives for Livestock Production and Watershed Protection in Arenal, Costa Rica

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Abstract

Conventional wisdom amongst environmentalists holds that the cutting of tropical forest for livestock production is not only bad business but also bad for the environment. In particular, it is thought that conversion to pasture leads to rising sedimentation of waterways and reservoirs, an increase in flooding and loss of dry season water supply. In the case of Lake Arenal, Costa Rica this conventional wisdom is stood on its head in an evaluation of the market and policy incentives guiding land use in the Río Chiquito watershed of the Arenal region of Costa Rica. The study suggests that ranching, dairy and associated downstream hydrological effects represent important values to the Costa Rican economy, values that significantly outweigh expected returns from options for reforestation or forest regeneration. Further, there appear to be no large market or policy incentives subsidizing livestock production or providing incentives for rapid deterioration of soil productivity. Thus non-hydrological externalities associated with changing land use from forests to livestock production, such as carbon fixation, biodiversity, ecotourism and existence values, are likely to be of minimal importance in Río Chiquito. Therefore the analysis suggests that there is little reason to encourage large-scale reforestation of the watershed or to purchase land for protection. Instead efforts should focus on how to maximize the complementary returns from livestock and water production.

Abrégé

Chez les écologistes, les idées reçues veulent que l'abattage des forêts tropicales au profit de l'élevage soit non seulement une mauvaise affaire (au sens économique), mais aussi un mauvais coup porté à l'environnement. On estime, en particulier, que la conversion en pâturages des terres concernées entraîne une sédimentation accrue des cours d'eau et des réservoirs, une augmentation des inondations et la perte de l'eau disponible en saison sèche. Dans le cas du lac Arenal, au Costa Rica, cette idée reçue est mise sans dessus dessous par l'évaluation des incitations proposées par le marché et par les instances politiques pour orienter l'utilisation des terres du bassin versant du Río Chiquito, dans la région de l'Arenal (Costa Rica). Le présent travail suggère que l'élevage en ranchs et la production laitière, ainsi que leurs retombées hydrologiques en aval, sont pour l'économie costa-ricaine d'importantes sources de valeur ajoutée. Cette valeur ajoutée excède de beaucoup les retombées favorables attendues d'options telles que le reboisement ou la régénération forestière. De plus, il ne semble pas qu'existent de substantielles incitations, qu'elles fussent dues au marché ou aux instances politiques, revenant à subventionner l'élevage ou à pousser à une rapide détérioration de la productivité pédologique. Enfin, l'évolution des périphénomènes non-hydrologiques (fixation du carbone, biodiversité, écotourisme et prix attaché à la seule existence de la forêt) due à la transition de l'utilisation des terres de la forêt à l'élevage, risque fort de n'avoir qu'une importance minimale dans la région du Río Chiquito. En conséquence, l'analyse effectuée suggère qu'il n'y a guère de raison d'encourager le reboisement à grande échelle du bassin versant ou d'acquérir des terres pour préserver la forêt. L'effort devrait plutôt porter sur le moyen de maximiser les recettes de l'élevage et de l'exploitation de l'eau, d'ailleurs complémentaires.

Resumen

Los ambientalistas comúnmente sostienen que la tala de bosques tropicales para la ganadería no sólo es mal negocio sino que también es nociva para el medio ambiente. En particular, se piensa que la conversión a terrenos de pastoreo conduce a un aumento en la sedimentación de las vías fluviales y represas y conlleva a un incremento de las inundaciones y bajas en las reservas de agua en épocas de sequía. En el caso del lago Arenal en Costa Rica, esta percepción convencional se ve cuestionada mediante una evaluación de incentivos de mercado y de política que guían el uso de la tierra en la cuenca del Río Chiquito en la región de Arenal. El estudio demuestra que la ganadería, la lechería y los efectos hidrológicos asociados a éstas representan valores importantes para la economía costarricence. Estos son valores que sobrepasan los ingresos esperados de opciones como la reforestación y de la regeneración forestal. Además, no parece haber incentivos de mercado o de

política que subsidien la ganadería, ni incentivos para un deterioro acelerado en la productividad del suelo. Dado que las externalidades no hidrológicas asociadas con la ganadería, tales como la fijación de carbono, biodiversidad, bioturismo y valores de existencia son tan poco importantes en Río Chiquito, el análisis sugiere que no hay razón para fomentar la reforestación de las cuencas a gran escala o adquirir tierra para protegerla. Los esfuerzos deben dirigirse, en cambio, a maximizar los ingresos complementarios provenientes de la ganadería y de las vías fluviales.

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Introduction

The conversion, fragmentation and disturbance of tropical forest ecosystems in developing countries over the last few decades are well documented. Increasing recognition of the role that the economic importance of intact forest ecosystems may play in providing incentives for the conservation of these forests has led economists to study the non-market benefits that are lost when intact forests are modified or converted. These benefits include locally consumed non-timber products, biodiversity prospecting, ecotourism, carbon sequestration, soil and water conservation, and option and existence values. This line of research serves to illustrate the importance of conservation and, ideally, to flag developments that although possessed of considerable market potential, will adversely affect the non-market goods and services provided by intact tropical forest ecosystems.

A second and, again, ideally complementary line of research has explored the "incentives" for and against deforestation, focusing on market, political and institutional forces that drive deforestation. This research typically also explores the means by which conservation values can be made explicit in decision-making, through a mix of the creation of new markets, institutions and enabling policies and legislation. Despite the emphasis on deforestation it is worth noting that the problem can be viewed from either of two perspectives: that of preventing forest degradation (i.e. protecting forest) or that of encouraging watershed protection once ecosystems are disturbed or converted to other uses (through soil conservation, reforestation or forest regeneration). The type of problem faced varies depending on whether or not the locale under scrutiny has reached a post-agricultural frontier stage in its development.

The role of tropical forests in watershed protection, or more specifically the maintenance of "normal" hydrological function (both on-site and downstream), is a prime example of this trend in environmental economics. Economists and environmentalists alike typically cite soil and water conservation as one of the most important of environmental services offered by tropical forests. Efforts to demonstrate this hypothesis rely on the integration of hydrological and economic analysis. The reality, however, is that there are few, if any, studies that attempt to look at both the on-site and off-site aspects of the problem, much less simultaneously incorporate valuation information into the design of incentives. With respect to the design of incentives the difficulty is that with no clear understanding of the economic costs or benefits of natural hydrological function, efforts to promote watershed protection may lead to poor choices with regards to incentives policies or the misallocation of project funds. In other words, incorporation of economic information into a holistic "watershed approach" remains an elusive objective. This picture is complicated by the increasing tendency to question the traditional belief that watershed protection values provided by primary forest are unambiguously positive.

This paper presents the results of an effort to assess the market and policy incentives that may affect the land use decisions made by landholders in an upland tropical watershed in Costa Rica. The principal objective of the paper is to identify the nature and extent of the watershed protection problem and to assess to what extent it is a result of market or policy failure. Through the use of valuation methodologies and cost-benefit analysis the impacts of current productive use of the land are contrasted with those likely to be generated by alternative uses, principally forest protection or production. By undertaking the analysis from both a private and an economy-wide perspective it is possible to illustrate how current economic policies and market failures affect land use and its economic consequences. This information subsequently feeds into a companion effort to analyse and develop incentives and institutional arrangements that might underpin a constructive and participatory effort to improve watershed management in the study area (Aylward and Fernández González 1998).

Selection of a site for the study was guided by a number of factors, principally the existence of a demonstrated watershed protection problem (from at least a biophysical perspective), a reasonable database on hydrological problems and the prospects for a fruitful, multi-disciplinary and collaborative research effort.[1] In particular, it was important to choose a site where the hydrological impacts of different land uses would be likely to be the most significant value generated by forest cover.

These conditions were all met in the case of Lake Arenal, which serves as the source of water for the country's largest hydroelectric facility and irrigation scheme. Previous analyses suggested an extreme mismatch between land use capability and actual land use. Measurement stations operated by the Costa Rican Electricity Institute (ICE) provided considerable data on the tributaries reaching the Lake. The Río Chiquito watershed is one of the three large, micro-watersheds that form the upper Arenal watershed and provide the majority of the water supply to Lake Arenal. The other two micro-watersheds are largely forested and in the hands of conservation organisations. As the only one of the micro-watersheds in the upper Arenal watershed that is largely converted to pasture, Río Chiquito has long been targeted for action by conservationists and remains an area of considerable debate and conflict in this regard. Given that Costa Rica is largely a post-agricultural frontier country, Río Chiquito exemplifies the incentives problem faced in the local context and was chosen as a focal point for the study.

The paper begins with an overview of the conceptual framework and methodology employed in analysing existing political and market incentive structures. The ensuing three sections explore in turn the methods and results of the analysis of livestock production, on-site erosion and productivity issues, and off-site hydrological impacts. This information is then employed in stepping through the conceptual framework and evaluating the market and policy incentives that influence the choice between livestock production and watershed protection in Río Chiquito. Conclusions of the study and topics for future research are summarised in the final section of the paper.

[1] A fundamental objective of the CREED Programme is the development of the collaborative partners' capacity to apply economic analysis to environmental and development issues.

Conceptual Framework and Methodology

The objective of the analysis is to identify market and policy incentives critical to land use decision-making in Río Chiquito, in particular those that lead landholders to continue livestock production on holdings currently dedicated to pasture. Ranchers are assumed to be profit maximisers, operating in an environment in which market and policy conditions may not always lead to decisions that consistently maximise both landholder profits and economic welfare.[2] The conceptual framework lays out the steps necessary to identify potential conflicts between these two viewpoints using a quantitative, cost-benefit approach. The conceptual framework consists of four steps divided into three phases, as presented below.[3] As a guide to the ensuing sections a brief introduction to the methodology employed in evaluating this framework is then provided.

Private Incentives

The first step is to determine the benefits and costs of ranching from the private perspective, B^P and C^P, that is, how ranchers view them. This yields the private returns to ranching:[4]

$$R^P = B^P - C^P \qquad \text{Eq. 1}$$

That these returns are expected to be positive is self-evident given that ranchers are actively engaging in ranching. Evidence to the contrary would require a substantial explanation.

Private and Societal Incentives Given Policy Distortions and On-Site Market Imperfections

In the next phase the net benefits of ranching are adjusted to reflect the removal of policy or on-site market imperfections (not including off-site environmental impacts) that factor into private decision-making. This phase involves two distinct steps: (1) standard economic project evaluation adjustments for distortions of input and output prices and (2) "environmental" adjustments to account for land-related distortions or imperfections.

Removal of Policy Distortions. 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 market prices and "economic" prices. This transformation of market prices to "economic" or shadow prices, B^E and C^E, involves examination of any distortions in input and output prices and the use of social rates of discount in place of private rates. The result is the standard economic assessment of net benefits in the absence of such distortions:

2 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)

3 For a detailed presentation of the framework and origins of the Arenal case study see Aylward, Echeverría and Barbier (1995).

4 As forestry and hydrological values are eventually included in this framework, the returns discussed here are discounted intertemporal returns, i.e. net present values.

$$R^E = B^E - C^E \qquad \text{Eq. 2}$$

If $R^E < 0$ then the allocation of private resources to ranching is maintained only as a result of the distortionary policies identified. Resolution of the incentive problem then becomes a matter of assessing the means of removing the distortions currently in place. Following the removal of such distortions, if ranching remains a profitable activity, then the analysis may move on to the next step in the analysis.

Removal of On-site Market Imperfections. There may also exist policy distortions and market imperfections that induce ranchers to ignore important land and environmentally related impacts of their land use on production and, thereby, fail to achieve a level of production that would be privately efficient in the absence of such failures. Four processes may be driving ranchers to make less than perfect decisions in this regard: (1) conditions in capital markets that lead to a private discount rate that greatly exceeds the social discount rate, (2) insecurity over land tenure, (3) a failure of the asset markets for land to accurately reflect land values and (4) a lack of information regarding the long-run impact on productivity of current land husbandry practices.

The existence of these conditions may lead to a disparity between the rate at which the rancher "uses" the soil resource and that at which society would prefer the resource to be "used." The intertemporal "loss" of productivity that results is called the "user cost" of soil erosion, and in practical terms reflects the value of lost future productivity incurred by ranchers who degrade their soil resource faster than is economically optimal, as measured with the social rate of discount.

If these failures exist, then by implication the economic returns to ranching are not adequately reflected in the previous equation. Instead it is necessary to also include the user cost of soil erosion, *UC*, into the equation. The economic returns to ranching then become:

$$R^{E'} = B^E - C^E - UC \qquad \text{Eq. 3}$$

If the addition of user costs to the equation leads to negative economic returns then it may be that there are conservation technologies that, if employed, would generate sufficient incremental returns to push overall economic profitability back into the black. Alternatively it may be that a reduction in the intensity of use (i.e. a reduction in stocking rates) would be a less costly method of achieving the same goal.

Societal Incentives and External Costs

If production remains profitable following the inclusion of user costs, the analysis proceeds to the incorporation off-site, or external, costs associated with current land use practices in the watershed. They are labeled as "costs" due to the prevailing notion that ranching leads to negative environmental impacts. Typically, the market provides no incentive to ranchers to incorporate these off-site impacts of their land use into their own profit-maximising framework. Thus, the external costs are included into the analysis of returns from society's perspective. The full specification of the economic returns from ranching must also include the external costs of ranching, *EC*:

$$R^{E''} = B^{E} - C^{E} - UC - EC \qquad \text{Eq. 4}$$

If the economic returns are negative, once external costs are included, then the market failure that precludes the internalisation of these external effects into the farmer's land use decision framework is causing net economic losses to society.[5] Additional incentive mechanisms for promoting watershed protection are thus justified as a means of moving land use towards more economically efficient land use alternatives.

In this paper the emphasis is on hydrological externalities, which were previously determined as the principal off-site value in the study area (CCT 1980 and ACA 1993). Conclusions regarding the overall “suitability” of ranching as a land use (based on this analysis) are limited as the full external costs of ranching have not been incorporated. To this end it is useful to consider the potential impact of land use on other environmental values in the study area.

Clearly, this is a fairly simplistic and utilitarian approach to what is a very complicated problem, including, as it does, socio-economic, biophysical and institutional components. Given the long history of land use change and conflict between stakeholders in the Arenal watershed it would be presumptuous at best to assume that a quick “economic” or “policy” fix could immediately, costlessly and perfectly align private incentives to maximise economic welfare. For this reason, Aylward and González (1998) take a second look at the problem in a companion paper, deepening the approach to identify specific physical measures, institutional arrangements and incentive mechanisms for improving economic efficiency and widening the design and evaluation process to include non-economic factors and participatory processes. Thus, the analysis developed below can be regarded as a first cut at the problem, intended to assess in a quantitative fashion the presence of major policy distortions or market imperfections that may be leading to poor land use allocation in Río Chiquito.

Methodology

The framework is evaluated using financial and economic cost-benefit analysis. As suggested above the objective is to evaluate the profitability of livestock production from the private and economic perspective, include user costs of soil erosion (if any) and assess hydrological externalities. A qualitative assessment of other externalities is included and potential returns from carbon sequestration and calculated to round out the analysis of externalities. For the purpose of investigating how the resolution of incentives problems would affect competing uses and how one use would be preferred over another, the costs and benefits of alternatives are also calculated. These alternative are primarily forestry options, including absolute protection (no use) and forest production.

All analyses are carried out over a seventy-year time horizon. This horizon is chosen in order that the analysis provide ample time for longer term effects to play out, specifically to allow a full cycle of harvesting for the natural regeneration scenario under the alternative of forestry production. The analysis relies on discount rates developed by Aylward and Porras (1998) as part of the CREED Costa Rica project. A private opportunity cost of capital of 10%, with an

[5] Note that as the discussion revolves around “external costs” the equation employs a minus sign, however this does not preclude the existence of “external benefits” which could be added in (with a plus sign) to the equation.

upper range of 13%, is used to discount income and expenditure streams based on estimates of the cost of capital in Costa Rica. In the economic analysis, a simple opportunity cost of capital approach is used with a constant negative exponential rate of 9%. A range of 7% to 11% for the discount rate is explored in the sensitivity analysis. The sensitivity analysis also contrasts the use of this approach with that of the consumption equivalents method. In the latter method cost and benefit streams are separated according to their capital or consumption shares. The consumption rate of interest (CRI) of 9% is used to discount consumption flows and the CRI weighted by the shadow-price of capital (1.037) is used to discount investment flows.

Livestock production data is recorded in Costa Rican colones as of January 1995. Given a gradual decline in the value of colones during the course of the study, other data obtained in colones is also adjusted to constant 1995 colones. Results are reported in dollars using an exchange rate of 165 colones to the dollar, effective in January 1995. The costs of electric power production are reported in dollars and, therefore, no adjustments are made to these figures in the evaluation of hydrological externalities.

Subsequent sections report on the analyses of livestock production, hydrological externalities and forestry options. As appropriate each of these sections reports on methodologies, data and results. A final section pulls the information together to assess the conclusions emerging from the analysis of the conceptual framework.

Financial and Economic Profitability of Livestock Production

In this section the financial and economic analysis of livestock production is presented (i.e. the first two steps in the conceptual framework). The first question is whether livestock activities are profitable from the private perspective and, if so, to what extent? The second question is whether these same activities profitable from an economy-wide perspective (before inclusion of environmental effects). The latter includes the shadow pricing of inputs and outputs. Consideration of the user cost of soil erosion and hydrological externalities is then taken up in subsequent sections.
The evaluation of livestock production addresses a number of subsidiary issues regarding the impact of economic realities, trends and policies on the incentives for livestock production in the watershed. These include:

- a comparison of the principal livestock activities in terms of relative profitability
- the role of family labor in farmer decision-making
- the role of future price changes in key inputs and outputs in altering livestock profitability in the watershed
- the impact of import duties (and other taxes) on factor inputs on production
- the impact of economic policies that affect the farmgate price of milk on the profitability of different livestock production activities

In addition, observations on the relationship between private profitability and variables such as farm size and spatial distribution within the watershed, may assist efforts to design incentive mechanisms for improved watershed management in the area.

A distinction that must be stressed at the outset is that the objectives listed above depend on the estimate of the expected returns to land from the various activities. The analysis should not be taken as, nor is it intended to be, a household level analysis. Instead returns to each of the holdings in the sample are calculated. As a result land is not included as a factor input except as its productive value may be expected to change over time. This issue is addressed through the discussion of the on-site costs of soil erosion.

First, a summary of the methods and data employed is presented, followed by a discussion of the results.[6]

Methods

The analysis is undertaken for the three livestock activities present in the Río Chiquito watershed. These are ranching, dairy and dual purpose (mixed ranching/dairy). The basic unit of analysis is a production holding as identified through a census of the watershed. Cost-benefit analysis is used in evaluating the costs and benefits on each holding. The net present value of production on holding *i*, depends on the extent of benefits generated from beef and

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[6] A full accounting of the methods and data is found in Aylward *et al*. (1998) .

milk production, B^B and B^M respectively and costs incurred (fixed costs, *FC*, variable costs, *VC*, and user costs, *UC*) in every period *t*, as follows:

$$NPV_i = \sum_{t=1}^{n}((B_{it}^B + B_{it}^M - UC_{it}) / (1+r)^t - (FC_{it} + VC_{it}) / (1+r)^{t-1}) \qquad \text{Eq. 5}$$

Discounting of intertemporal flows is achieved using a discount rate *r*, over a planning horizon of *n* years. It is assumed that fixed and variable expenditures occur at the beginning of the year while benefits (and user costs) accrue at year's end. For this reason, fixed and variable costs are not discounted in the first year and are discounted at *t-1* years in subsequent periods.

The net present value (per hectare) of each holding is derived from the net present value of the holding and the area dedicated to the activity, A_i. The average per hectare returns to each type of livestock activity is calculated by grouping together the per hectare returns for each type of holding, summing them and dividing by the number of holdings in the group, *I*.

$$NPV_{average} = \frac{\sum_{i=1}^{I} NPV_i / A_i}{I} \qquad \text{Eq. 6}$$

This result provides an indication of returns on the average holding under each type of livestock activity. Alternatively, it is possible to calculate the average productivity per hectare of land in the watershed by weighting the NPV of each holding by the number of hectares in that holding and dividing through by the total number of hectares.

<u>Levels of Analysis and General Parameters</u>

The private analysis is divided into a cash flow analysis and a private opportunity cost approach. The former takes into account only actual monetary flows. This analysis is useful for estimating the financial self-sustainability of a given activity, but should not be taken as indicative of the potential worth of an activity to the landholder or society. The costs and benefits included under the cash flow benefit are: (1) sale of milk (including its use in cheese production), (2) sale of animals, (3) purchase of fixed and variable inputs and (4) salvage value of fixed assets at the end of the planning horizon. As the analysis treats the production units as ongoing activities rather than as new projects, cash flows associated with the purchase of fixed assets occur only as necessary based on existing farm inventories and replacement needs (not all at once in the first year).

The private opportunity cost approach incorporates non-monetary flows that reflect conscious decisions regarding the opportunity costs of different inputs and outputs as made by the operator of the holding. For example, the use of unpaid family labour does not enter into the cash flow analysis. However, it is clear that the use of family labour implies a trade-off with respect to the opportunity cost of these economic inputs and outputs. In the private opportunity cost approach the following unpriced inputs and outputs are priced at their opportunity cost: home consumption of milk and beef, other uses of milk and unpaid family labour. The opportunity costs of fixed assets at inception are explicitly included based on the farm inventory. The same opportunity cost of capital is used as in the cash flow analysis.

In the economic cost-benefit analysis, economic prices (as opposed to observed market, or financial, prices) for input, outputs and the cost of capital are employed in order to assess the true resource costs and productive benefits to the Costa Rican economy of these activities. The types of adjustments necessary to arrive at economic prices for inputs and outputs are explained below with a brief indication of their relevance to the study and sources employed.

Adjustment for Transfer Payments: Input Prices.

In adjusting for transfer payments an attempt is made to re-acquire the undistorted market price, or economic price, that would have prevailed in the absence of such policies (Jenkins and Harberger 1986). In this study an effort is made to assess the likely impact of a range of taxes that apply to goods that are in theory tradeable. In such cases, the economic price at the border is the CIF price (cost on arrival including insurance and freight). Typically in the development of shadow prices, any import tariffs (and any other distortions) are simply disregarded in building the cost structure of the good to the point of sale (Jenkins and Harberger 1986). In the case of the current study, available data includes only information on observed market prices at point of sale and the policy instruments in place, not the actual CIF price. Thus, a second-best method is constructed as follows.

Costa Rica is a small country with a resource-based economy located relatively close to the United States, a country capable of producing almost anything of use in livestock production. In addition, Costa Rica's Central American neighbors are also heavily invested in livestock production and produce many of the items used in such production. Thus, it is assumed that practically all manufactured or produced goods are potentially importable. It also may be assumed that for reasons of political economy, import tariffs exist only where there exist local industries desiring protection and, most importantly, that these tariffs exist only when a real threat to the local producer exists. In other words, the overall assumption is that the duties placed on imports are necessary in order to raise prices of imports to such levels that they will be more or less on par with internally produced goods. Removing the impact of these duties will then identify the potential resource savings that could be obtained by eliminating the duties.

The importation of goods in Costa Rica is subject to four different types of tax as follows:

- import duties, *DAI* (*derechos arrancelarios de importación*),
- excise taxes imposed on selected imports, *SC* (*selectivo consumo*)
- a statutory taxed of 1% levied on practically all goods under Law 6946, *LEY*
- a sales tax levied on most goods, *IV* (*impuesto de ventas*).

In effect, the *DAI*, *SC* and *LEY* function as a single import duty. Goods produced in the country are subject to only the IV. The calculation of the amounts of each of these duties to be paid on an import with an arrival price in port of CIF is as follows:

$$DAI = CIF * t_{DAI} \qquad \text{Eq. 7}$$

$$SC = (CIF + DAI) * t_{SC} \qquad \text{Eq. 8}$$

$$LEY = CIF * t_{LEY} \qquad \text{Eq. 9}$$

$$IV = (CIF + DAI + SC + LEY) * t_{IV} \qquad \text{Eq. 10}$$

Where:

t_{DAI} = percentage DAI rate of tax (e.g. 40%)

t_{DAI} = percentage SC rate of tax, and so on

With this in mind it is clear that the in-country market price (excluding transport and marketing), or P^M for the good will be as follows:

$$P^M = CIF + DAI + SC + LEY + IV \qquad \text{Eq. 11}$$

As pointed out above the undistorted market price of the good, or economic price, is in fact its *CIF* (subsequently renamed P^E). Substituting P^E into the above equation and rearranging terms gives us the following equation for calculating the economic, or shadow price of tradeable goods based on the tax rates and the in-country market price:

$$P^E = P^M \,/\, (1 + t_{DAI} + t_{SC}(1 + t_{DAI}) + t_{LEY} + t_{IV}(1 + t_{DAI} + t_{SC}(1 + t_{DAI}) + t_{LEY})) \qquad \text{Eq. 12}$$

For the purposes of this document the denominator will be called the "shadow factor." It is worth pointing out that this method will be only more or less correct given the assumption that the denominator in the above equation can be applied to the actual market price found at the point of purchase. Typically, trading margins and transport costs would also need to be taken into account were they available. In other words the P^M that is actually used in this study overstates the amount to which the adjustment should be made and, thus, overstates the size of the adjustment. This implies that the economic price will be slightly biased downward, i.e. it will understate the economic price.

<u>Adjustment for Transfer Payments: Output Prices</u>

Beef has long been one of Costa Rica's principal export products. Thus, even though minor import duties are levied on the import of live animals, import regulations are not likely to be an important policy distortion in local markets for beef. Instead, export policy and production subsidies are potentially more relevant sources of distortion. As for production subsidies, the principal subsidy received by ranchers in Costa Rica has been the extension of credit to ranchers at below market interest rates. However, such policies have largely been eliminated in the 1990s. For this reason, no adjustment is made to the company or auction price in the economic analysis.

In the case of milk, import tariffs are combined with both producer and consumer level price controls. Fortunately, existing studies and data serve to enable a calculation of the actual extent of the difference between the economic price as represented by an imported substitute (in this case milk powder valued at its CIF price) and the retail market price. Assuming that milk processors are not collecting excess rents and are not inefficient producers it is possible to use this difference to draw conclusions about the economic price of milk at the farmgate. It needs to be stressed that this analysis relies on assumptions about the substitutability of reconstituted milk and "real" milk. It also raises questions about whether an economic price based on current international prices is itself useful when international markets are themselves heavily distorted by producer subsidies in exporting countries. Given uncertainty in this regard the base case scenario explored in the paper does not adjust the farmgate price of milk, rather it is left for the sensitivity analysis to explore.

Due to the complexity of the cost-benefit analysis a Microsoft Access® Basic program was developed to carry out the necessary computations. Given this capability, a number of numerical parameters are built into the structure of the analysis so as to enable sensitivity analysis of the results. In particular, the program may be used to assess variation in discount rates (both private and social), the impacts of projecting changes in real price trends of key inputs and outputs, and the relative importance of any uncertainty over the base prices of inputs or outputs.

Data

Of 156 land parcels identified in the Río Chiquito watershed a census survey gathered land use and production information on 137 of these parcels. Discarding those parcels not involved in livestock production and accounting for parcels operated as a single holding, leads to the identification of a total of 120 livestock holdings in the watershed.

A detailed questionnaire on inputs was simultaneously applied to a random subset of the holdings in the census. Unfortunately, the accuracy of the pre-survey list compiled by the research team from prior lists and interviews with key informants was limited and greatly over-estimated the number of holdings. Thus, although the team initially kept to the random sample, it became apparent once the survey was well under way that it would be necessary to simply apply the cost questionnaire in every subsequent interview. As a result it was not possible to obtain the intended number of random samples from each of the three production types. Nevertheless the questionnaire was applied to 69 holdings in the Río Chiquito watershed. The distribution of the census holdings and sample holdings are presented in Table 1. Given the difficulties in selecting a sample and the relative importance of each of the three livestock production activities it is not surprising that ranching has the highest absolute number of holdings sampled, followed by dual purpose ranching and dairy.

Table 1. Livestock holdings in the sample

	Census Total (N)	Sample Total (N)	Unusable (N)	Usable (N)	Usable Sample as % of Census Total
Ranching	69	37	12	25	36%
not linked to dairy[1]	49	28	4	24	49%
linked to dairy	20	9	8	1	5%
Dual Purpose	29	17	2	15	52%
Dairy	22	15	4	11	50%
Total	120	69	18	51	43%

Notes: [1] Holdings that are not linked to dairy farms which are located outside the watershed

The table also indicates the number of sample respondents that were dropped due to problems with either production or cost data. In total, three-quarters of the sampled farms produced usable data. The attrition rate was, in large part, due to the need to discard a large number of the sample holdings that were linked to dairy farms outside the watershed. As these holdings are used in a complementary fashion to these dairy operations they did not yield usable production data on "ranching." As a result only one of twenty holdings of this type was successfully sampled and they are excluded from further analysis. For the remaining holdings, approximately half the number of holdings surveyed in the census are successfully sampled.

Results

The mean net present values (and their standard deviations) for each of the production types are listed for each type of analyses in Table 2. Net returns under the cash flow analysis are positive for all three activities. Moving to the private opportunity cost analysis, returns fall as the effect of adding in private opportunity costs is negative on net. In particular, this reflects the large quantity of family labour used on the smaller farms. Economic returns rise slightly (over private returns) as the effective discount rate is lowered and inputs are priced at their lower economic prices. Returns are negative (and only slightly so) in the case of the private opportunity cost and economic analyses of ranching. Average per holding returns for the entire sample are close to \$500/ha. When viewed by production type, considerable variation is exposed with dairy having by far the highest net per hectare return over the sample (\$1,600 in economic terms). Returns to dual purpose are less than half those of dairy with returns to ranching lagging considerably.

Table 2. Livestock production mean net present values

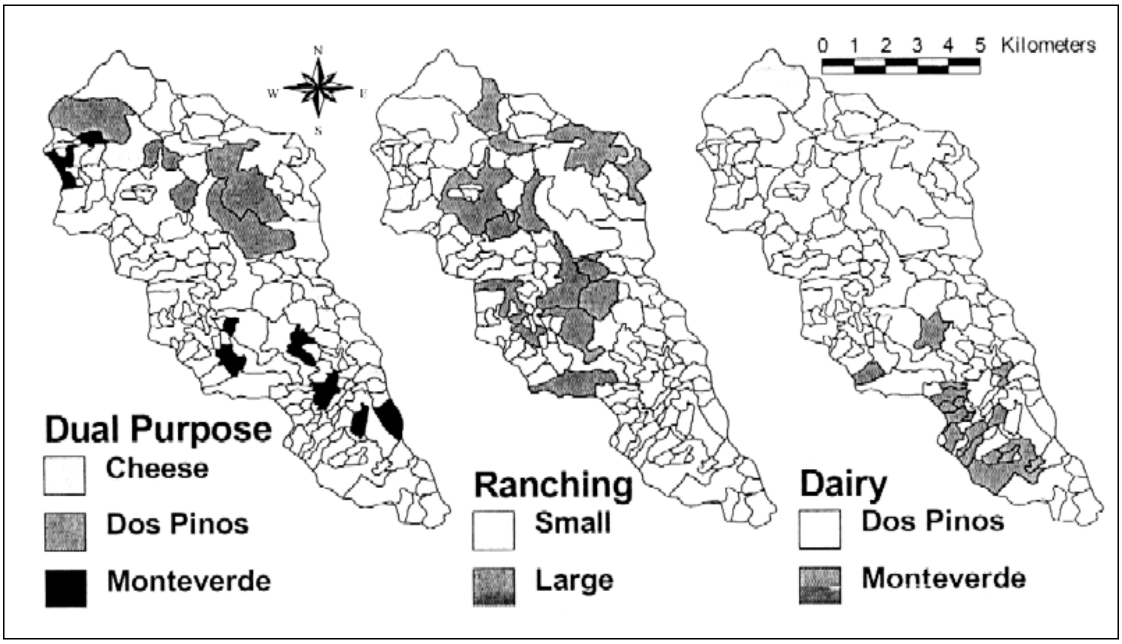
		Cash Flow			Private Opportunity Cost			Economic		
		Mean	Range		Mean	Range		Mean	Range	
(\$/hectare)	N		Low	High		Low	High		Low	High
Ranching	25	417	44	790	(129)	(558)	299	(43)	(511)	426
<80 hectares	16	166	(359)	691	(634)	(1,133)	(135)	(588)	(1,135)	(40)
>80 hectares	8	977	748	1,207	884	643	1,125	1,053	769	1,336
Dual Purpose	15	1,051	436	1,667	308	(473)	1,089	545	(341)	1,432
Dos Pinos	2	1,652	355	2,518	1,413	909	1,917	1,874	1,099	2,650
Monteverde	7	1,437	844	2,461	327	(1,296)	1,949	605	(1,212)	2,421
Cheese	6	402	(246)	1,050	(81)	(455)	293	33	(397)	463
Dairy	11	2,632	1,151	4,113	1,020	(794)	2,834	1,557	(577)	3,692
Dos Pinos	2	6,011	4,489	7,533	6,032	4,481	7,583	7,408	5,930	8,886
Monteverde	9	1,881	540	3,222	(94)	(1,391)	1,203	257	(1,303)	1,818
Total	51	1,081	618	1,545	247	(253)	748	475	(110)	1,061

Notes: Figures in parenthesis denote negative returns. One of the ranches in the sample is linked to a dairy holding outside the watershed and is excluded from the sub-groupings which represent "pure" ranching holdings. "Range" refers to the 95% (two-sided) confidence level.

The confidence intervals for the means, as shown in the table above, are quite large, reflecting the small sample size, the wide diversity of practices employed within each farm, the range of biophysical conditions and the disparity between large and small enterprises. Nonetheless, it remains clear that there does exist a fairly significant trend within the sample as noted above. In order to tease out these trends, the sample is further divided according to the factors most likely to be playing an important role in introducing such large variation within each activity type. The results of this exercise (as shown in Table 2) show that the returns to ranching vary considerably with the size of the operation. Economic returns for ranching holdings over 80 hectares in size are approximately \$1,050/ha, while returns to smaller ranches average are negative at -\$588/ha. The confidence interval for the under 80 hectare size group remains high due to three operations with large negative returns. Still, the trend in this group is marked as only four ranches generate returns in the black.

Segregating ranching operations in this manner leads to a relatively narrow confidence interval for large ranches firmly in positive territory (\$769/ha to \$1336/ha). This suggests that there may well be very important economies of scale in ranching. A factor that is potentially correlated with farm size, and thus that may limit the strength of the previous conclusion is the location and, hence, biophysical conditions experienced by each holding. A review of the spatial dispersion of the holdings reveals that the larger farms are, generally speaking, more likely to be located in the lower watershed, with a large number of the smaller, less profitable, operations being located in the interior of the watershed (see the figure below).

Figure 1. Map of livestock holdings in Río Chiquito



For the dual purpose and dairy operations, holdings may be divided into three groups: (1) large, mechanised operations selling to Dos Pinos; (2) smaller operations, mostly dairy, selling to Monteverde; and (3) medium-sized operations, mostly dual purpose, selling milk for cheese production or engaging in home production of cheese. With returns classified in this fashion, the farms selling to Dos Pinos produce superior returns, although the sample is small at two farms each for dual purpose and dairy. The advantage, in this case, is in favour of diary holdings which have economic returns of over $7,000/ha as compared with approximately $1,900/ha to dual purpose. Although the confidence intervals remain large for this category, holdings selling to Dos Pinos may be expected to produce returns of over $1,000/ha. This comparison is reversed in the case of Monteverde producers, in which case, dual purpose holdings generate just over $600/ha and dairy $250/ha on average. Two factors may explain this reversal. At the more marginal levels of net benefits obtained by the operations selling to Monteverde the returns from stock received by dual purpose holdings play an increased importance. It is also true that dairy operations are more input intensive, spending a proportionately larger amount on variable costs than in the case of dual purpose holdings. Finally, the results indicate that those dual purpose holdings involved in cheese production lag all other milk producing holdings with $33/ha in net economic returns.

If mean returns per hectare are considered in place of the mean returns per holding the profitability of livestock production comes to just over $1,200/ha (as opposed to just under $500/ha for the latter). This is due to the increased influence of the larger, more profitable ranches and farms in the sample. The figure also suggests that on average, land in pasture in Río Chiquito earns a considerable rate of economic return.

Sensitivity Analysis: Discount Rates

A number of discount rate scenarios were assessed in a sensitivity analysis. For both the cash flow and private opportunity cost analyses an upper range of 13% is compared with the best estimate of 10%. The results indicate that changes in the cash flow values shift downward by 26% to 35%. In absolute terms the changes ranged from -$148/ha for ranching to -$681/ha for dairy. Percentage changes were higher for the private opportunity cost scenarios ranging from -39% to -47%. The changes in absolute terms, -$394/ha for dairy and roughly -$140/ha for ranching and dual purposes, are comparable to those in the case of the cash flow scenarios. This rise in sensitivity reflects the drop in the best estimates that occurs in moving from the cash flow to the private opportunity cost analysis. In sum, changes to the discount rate scenarios under the cash flow and private opportunity cost analysis do decrease net returns to these activities, but not sufficiently to push average returns to dairy or dual purpose into the red.

Table 3. Livestock production net present values: Discount Rate Sensitivity Analysis

	Discount	Ranching			Dual Purpose			Dairy		
	Rate	Mean	Change		Mean	Change		Mean	Change	
Analysis		$/ha	$/ha	%	$/ha	$/ha	%	$/ha	$/ha	%
Cash Flow	13%	269	-148	-35%	776	-275	-26%	1,951	-681	-26%
Private	13%	(181)	-52	-40%	163	-145	-47%	626	-396	-39%
Economic	7%	43	86	200%	783	238	30%	2175	618	28%
	11%	-98	-55	56%	391	-154	-39%	1158	-399	-34%
	9%, CE[1]	-35	8	-23%	554	9	2%	1575	18	1%

Notes: [1]Refers to the Consumption Equivalents approach to discounting in which investment flows are adjusted by the shadow price of capital (1.037 in this case) and then added to consumption flows with the resulting sum discounted by the consumption rate of interest (9% in this case).

The use of discount rates of 7% and 11% (in place of 9%) display the same pattern as seen earlier: returns to ranching changing by less than $100/ha and returns to dual purpose and dairy changing by 15 to 40%. Use of the lower rate even raises average ranching returns into the black. With the higher rate, dual purpose and dairy remain profitable activities.

Comparing the use of a simple 9% opportunity cost of capital to the use of the consumption equivalent method in the economic analysis demonstrates that the difference between the results obtained with these two approaches and rates is very limited (at least in this case).

Sensitivity Analysis: Private Profitability

As suggested in the conceptual framework, a negative result for the private opportunity cost suggests that producers are not acting as rational profit maximisers. Yet the private returns are negative in the case of dairy farms selling to Monteverde, smallholder ranches and dual purpose producers linked to cheese production. The implication is that there are no net private returns to land as a factor in production. Since, the land does have resale value the decision taken by these landholders to remain in production represents somewhat of a conundrum. Three potential explanations exist. First, landholders may be non-rational in the sense that they are more concerned about returns to labour than returns to land. The positive cash flow values of all of these activities provide evidence that these activities do earn money with which the landholders may support their livelihoods. This also suggests a second explanation: that the real opportunity cost of family labour employed on the holdings is over-estimated in the modeling of private and economic returns. The final explanation concerns the role of landholder expectations regarding future price trends. Each of the latter two points is taken up below using further sensitivity analysis of the results.

As a sensitivity analysis, figures for own labour costs are halved and the private opportunity cost analysis is repeated to assess the changes. Significant changes to the overall returns to the three activities are observed. Ranching returns rise \$200/ha to become profitable at \$101/ha (see table below). Returns to dual purpose rise by \$350/ha and to dairy by \$850/ha.

Table 4. Livestock production Net Present Values: additional sensitivity analyses

	Private Opportunity Cost						Economic	
Parameter:	Own Labour Cost		Beef Price Growth Rates				Price of Milk	
Sensitivity to:	Reduction of 50%		Rise of 2%		Rise of 0.5%		Shadow Factor 1.45	
	Mean	Change	Mean	Change	Mean	Change	Mean	Change
Ranching	101	+230	656	+785	37	+166	na	na
<80 hectares	-291	+343	138	+772	-470	+164	na	na
>80 hectares	917	+33	1,740	+856	1,066	+182	na	na
Dual Purpose	665	+357	649	+341	381	+73	69	-476
Dos Pinos	1,527	+114	1,941	+528	1,525	+112	1,012	-862
Monteverde	852	+525	616	+289	388	+61	-107	-712
Cheese	160	+241	258	+339	-9	+73	na	na
Dairy	1,888	+868	1,216	+196	1,062	+42	-485	-2,042
Dos Pinos	6,032	na	6,223	+191	6,073	+41	3,965	-3,443
Monteverde	967	+1061	103	+197	-52	+42	-1,474	-1,731

Notes: All figures are mean values in \$/hectare, na is not applicable to the holdings in the sub-sample.

The changes by type of holding are even more pronounced. Reflecting their reliance on hired labour, the returns to large ranches and to dual purpose and dairy producers selling milk to Dos Pinos are affected only marginally. Returns to small ranches and dual purpose operations selling to Monteverde improve by \$300/ha and \$500/ha respectively with the former remaining negative (-\$291/ha). Both dual purpose producers selling milk for cheese production and dairy farms selling to Monteverde record positive returns under this scenario. The returns to the former (\$160/ha) shift only by \$250/ha. Returns to the latter, however, gain over \$1,000/ha with the halving of the imputed price of family labour. In other words, if the true opportunity cost of the family labour employed on these holdings is just half the going wage rate, private returns to land on such holdings would be positive at \$967/ha. As the results on these dairy holdings are extremely sensitive to parameters for family labour and as the base case is an upper estimate of the cost of family labour, the best estimate of -\$97/ha for the private opportunity cost analysis is probably an underestimate of returns.

An alternative explanation for the apparent irrational behavior of producers maintaining livestock production in face of negative private returns relates to uncertainty over relevant future price trends. Thus, an additional question emerging from the analysis is how sensitive are the negative results under the private opportunity cost analyses to changes in prices of outputs? As indicated earlier, the government sets the farmgate price for milk. This is

accomplished using a model of milk production costs. Typically, the farmgate price is revised twice a year, in line with the effects of inflation on the cost model. Thus, it is unlikely that farmers hold expectations that the real price will vary substantially in the future.
On the other hand, fluctuation in beef prices, principally in a downward direction over the last decade, may effect rancher expectations regarding future prices. The Chairman of the local Ranchers Association in Tilarán suggests that prices for beef move in ten year cycles, and that an "up" cycle is currently expected (Ruiz pers. comm. 1996). As a rough measure of the potential impact of upward changes in the price of beef changes in the real growth rate of the price of beef of 0.5% and 2% on the private opportunity costs analysis are explored.

A 2% change doubles the net present value earned by large ranchers ($1,740/ha) and pushes the return to smaller ranchers into positive territory ($138/ha). Dual purpose holdings involved in cheese production also move into the black ($258/ha). Interestingly, dairy farms selling to Monteverde are affected in a similar fashion ($103/ha) although the upwards movement in returns is more restrained. Overall, returns to dairy rise by roughly $200/ha to $1216/ha while dual purpose returns double to $649/ha. Under this scenario, however, the competitive advantage between dual purpose ranching and beef ranching tilts slightly towards the latter as overall mean returns to ranching rise to $656/ha. With a much smaller annual change of 0.5% in the growth of beef prices, the effect is muted considerably. Overall returns to ranching barely enter positive territory ($37/ha) while the effect on dual purpose and dairy is minimal ($381/ha and $1,062/ha respectively). Returns to each of the three groups of activities that had negative returns under the base scenario remain negative, although returns to small ranchers improve by almost $200/ha. In sum, landholder expectations regarding future price trends may be sufficient to provide an incentive to livestock producers to continue on in their respective activities. However, the expected rise is not insignificant; a 2% per year increase being necessary to move returns for all groups of producers into the black.

Sensitivity Analysis: Economic Profitability of Milk Production
Policy distortions that affect the market price of milk include price controls and import tariffs. Price controls exist on retail sales of milk with 2% milkfat ("lowfat") and the price paid to producers. The latter generally holds on milk sold to processing plants although it varies, as indicated above, due to transport costs, fat content and different payment methods (Motte and Billan 1994). One method of calculating the potential distortion introduced by price controls and import tariffs is to compare the retail price of lowfat milk with that of reconstituted milk powder imported from other countries. As shown in **Table 5**, Lizano (1994) calculates the cost of producing 2% milk in Costa Rica with powdered milk imported from Holland. The results of Lizano's analysis demonstrated that it would have been 18% cheaper to import and reconstitute powdered milk from Holland (cost of ¢57.54) than it was to purchase milk at the official price of ¢69.20. Taking the analysis one step further and subtracting the effect of the tariff (at the time) on the consumer price of imported, reconstituted milk leads to an economic price of ¢48.54 or the determination that it would be 30% cheaper to purchase abroad (see Table 5).[7]

[7] At the time of the analysis by Lizano (1994) tariffs were 20% on powdered milk and 10% in the case of liquid milk and imports of milk products required demonstration of an in-country shortage and a special license from the government. The import restrictions have since been removed however, the tariff on liquid milk, fresh cheese and milk powder of 109% as of January 1995 has the same effect of essentially prohibiting dairy imports.

If it is assumed that milk processing is no less efficient than milk production for processors would need to adjust the price paid to producers by a proportional amount to continue to compete with an open market for imports. This is equivalent to a shadow factor of approximately 1.45 (P^M/P^E). In other words the Dos Pinos producer price for milk employed in the study would be adjusted downwards from 53.75 colones to 37.07 colones and the Monteverde price likewise.

Table 5. Calculation of the economic cost of milk

	Units	With Tariff[1]	Without Tariff[2]
Price FOB	$/mt	1,800.00	1,800.00
Transport Cost	"	162.50	162.50
Insurance	"	21.59	21.59
Price CIF	"	1,984.09	1,984.09
Tariff	"	396.82	
Price CIF + Tariff	"	2,380.91	1,984.09
Internal Transport	"	13.75	13.75
Other Costs	"	8.00	8.00
Importer Price	"	2,402.66	2,005.84
Importer Margin (15%)	"	360.40	300.88
Processor Price	"	2,763.06	2,306.72
Hydrating Cost	"	300.00	301.00
Wholesale Price	"	3,063.06	2,607.72
Wholesale Price (5%)	"	153.15	130.39
Retail Price	"	3,216.21	2,738.10
Retail Margin	"	321.62	273.81
Consumer Price	($/mt)	3,537.83	3,011.91
	($/liter)	0.42	0.35
	(col/liter	57.02	48.54
Market Price		69.20	69.20
Savings from using Imports		18%	30%

Notes: Importer margin is 15%, wholesale margins are 5% and retail margins are 10%. Conversion factor for turning powdered milk into 2% milk is 8.5. Exchange rate is 137 colones/$. Source: [1]Lizano (1994), [2]Calculations based on Lizano (1994).

The overall effects of such a change are to drive dairy returns into the red (-$485/ha) whilst dropping dual purpose returns from $545/ha to $69/ha (see **Table 4**). Disaggregating by groups demonstrates that the effect is felt most heavily by dual purpose and dairy producers selling to Monteverde, with returns to the former falling from $615/ha to -$107/ha and for the latter from $276/ha to -$1474/ha. Meanwhile returns to producers selling to Dos Pinos (both dairy and dual purpose) are halved, but remain positive. Cheese producers are not affected as the change is applied to milk sold under the price controls to the two major processors. Such large changes suggest that the effect of opening up domestic milk markets to low-priced imports of milk powder would have the potential effect of driving smaller, non-mechanised milk producers out of business. Milk production would, however, remain economic at larger scales and with mechanisation.

As the impact of the tariff is excluded from the comparison of economic import price and price-controlled milk the change in tariff levels does not alter the conclusions.

Two problems with this re-assessment of the economic profitability of milk production limit its persuasive power. First, it can be argued that reconstituted milk of 2% milkfat is not a perfect substitute for fresh lowfat milk. Consumers would be likely to be willing to pay an additional amount above the ¢37.07 amount calculated above in order to have the additional satisfaction of drinking fresh milk. Whether consumers would be willing to pay over 40% more is doubtful. Price controls are often intended to ensure the widest availability of the product, given the goal of producing it locally. Nevertheless, the elimination of the duties and import restraints would clearly lower the market price of milk products and extend the benefits of their consumption to a much wider section of Costa Rican society.

Indeed, it can be argued that a large amount of those who would then be able to afford to consume (or increase their consumption) milk would be less concerned about the difference in taste between fresh and reconstituted milk. On these grounds it would appear that the import policies in place simply serve to protect the Costa Rican milk industry and its producers and represent a considerable cost to Costa Rican consumers. This, in turn leads to the contention that the fostering of a protected dairy sector in Costa Rica may have been an important contributor to deforestation in the country.

While the above argument holds from a strict view of economic well being as measured at the national level, it is suspect when viewed from a wider economic perspective. It is no secret that milk production in OECD countries is heavily subsidised. The US Department of Agriculture calculates that the producer subsidy equivalent for milk production (the total of transfers to producers divided by the value to producers) in the US from 1984 to 1992 averaged 49% while in the European Community it averaged 47% from 1982 to 1989 (USDA 1994). In other words, Costa Rica may be protecting its milk producers, but so do the principal dairy producers and exporters. It is, therefore, quite likely that the price for powdered milk from the Netherlands would be a poor indicator of a truly "economic" international price for milk. While it may be true that Costa Rica could profit from over-production in the North in the short run by importing cheap powdered milk, it is not clear that this would be the best long-run strategy.

For instance, were the US and the EC to slash support to dairy producers, other things being equal, dairy product supply would fall and the price would rise. It might even rise above the comparable cost of production in Costa Rica, given that support to northern producers is of the same relative magnitude as that indicated by Lizano (1994) and this study. Costa Rica would then find itself importing milk at a price above that at which it could have produced milk had it supported its dairy industry. For this reason, the argument that Costa Rica should not be producing milk (based on its cost of production) and, therefore, should not have deforested areas currently in dairy production is fallacious on purely economic grounds. As a result there is good reason to suppose that the use of such a shadow factor greatly overstates the economic argument in favour of liberalising milk markets in Costa Rica.

Soil Erosion and Livestock Productivity

Due to the deep, volcanic character of the soil resource in the watershed it is not expected that the intertemporal allocation of soil will have a significant impact on livestock production. Nonetheless, anecdotal reports of problems with soil fertility suggest that it is worth attempting to assess the veracity of such claims using the available data. To this end three efforts to establish the user cost of soil erosion are presented below: (1) a review of responses to relevant questions asked as part of the census, (2) a correlation analysis of age of pasture and production, and (3) the inclusion of age of pasture in a livestock production function.

Methods

The three analyses build on data gathered during the field survey. First, responses to questions regarding soil erosion and conservation obtained through the census are summarised. These questions provide an indication of to what extent soil erosion is perceived as a problem and to what extent efforts have been made to mediate such problems.

Second, simple correlation analysis is used to assess if a relationship exists between the relative level of soil exhaustion on holdings and their agricultural productivity. Although erosion maps were produced for the watershed, uncertainty over the geographic positioning of the holdings on the tenancy maps precluded using this data as a means of arriving at site specific data on soil exhaustion or cumulative erosion. Instead a proxy for cumulative erosion is utilised. Cross-sectional data on the age of the pasture of each holding is available for most holdings surveyed and provides a useful proxy for exhaustion, reflecting as it does cumulative years in use. In the absence of soil conservation measures specifically targeted at erosion, it would be expected that the older the pasture on a holding the lower would be its natural productivity. Correlation analysis between production and age is thus undertaken.

Of course, soil exhaustion may be high but its effects on production may be evaded through the use of increased levels of inputs such as fertilizers and feed. Thus, the third step is to develop a production function examining the statistical linkages between these variables. Production of milk or beef is assumed to be a function of a vector of monetised inputs, *X*, and the level of soil exhaustion, *S*:

$$Q = f(\mathbf{X}, S) \qquad \text{Eq. 13}$$

In order to document the existence of a tendency for yields to drop along with soil exhaustion a production function must be developed in which *S* plays a significant role and exhibits a negative relationship with *Q*.

Nonetheless, with a production function that includes a soil variable it is possible to explore whether farmers are choosing a less than optimal intertemporal path for soil erosion from a strictly private perspective. Such an analysis should incorporate not just changes in the intensity of use, but also the potential to maximise net returns through investing in soil conservation measures. The fact that actual behavior does not reveal considerable investment in such measures in the study area, implies that such options are uneconomic and makes data collection that much more difficult. The analysis is, therefore, limited to the first three steps suggested above.

Results

Census Data

As part of the census interviews, respondents were asked questions regarding the productivity of their land and conservation efforts undertaken. Half of the 120 holdings engaged in livestock production indicated that they felt the productivity of their land had declined over time. Respondents were then asked to be more specific about what type of decline had been observed. The most frequently cited problem was that of wind (and cold), particularly during the summer months with 27 responses (22.5%). Additional responses included increasing dryness or drought (18 responses or 15%), the growth of weed species (17 responses or 14%) and problems with soil fertility (15 responses or 12.5%). The latter category of response is somewhat open to interpretation with seven respondents making general remarks about soils and erosion, and another eight respondents simply suggesting that it was necessary to use (more) fertilizer.

A subsequent question asked of the respondents was whether they were undertaking any soil conservation measures. Slightly less than half (52 respondents or 43%) of those interviewed replied that they were engaged in such activities. Of these an overwhelming majority, 45 respondents (86%), said that they had installed live fences and/or windbreaks. Interestingly, eight respondents replied that they were protecting existing forest areas as a means of breaking the wind or protecting water supplies. A small number (4) indicated that they had installed drainage ditches. One response each was received suggesting that soil conservation had been achieved using a system of fenced pastures, limiting the stocking rate and re-seeding pasture to the African King Grass (*Cynodon nlemfuensis*).

The responses to these questions indicate that there is a general perception that productivity is not what it could be, and that effort has been made to cope with this problem. However, it would incorrect to say that the results confirm that soil erosion is a serious problem limiting on-farm productivity. Just 12.5% of livestock holdings reported problems associated with soil erosion per se. As for conservation measures, the use of live fences and wind breaks are not erosion-specific measures. For one, the use of live fences is a low-cost fencing option and, thus, does not necessarily imply that a landholder is attempting to correct an erosion-caused productivity problem. Second, it is clear that the most frequently cited productivity problem is the wind (and cold) in the region. Live fence and windbreaks are arguably more appropriate measures to resolve these problems than they are for erosion.

Correlation Analysis

As part of the census questionnaire landholders were asked to indicate when their holdings were cleared of forest for pasture. This variable, henceforth denoted by "age of pasture" serves as a useful proxy for the cumulative effects of soil erosion on a given holding. The effects of erosion being felt through both a lowering of soil quality and depth. Other things equal it would be expected that the older the pasture, the more erosion that would have occurred and, therefore, the lower the natural productivity of the soil.

As a first effort to assess whether such a link holds in the Río Chiquito watershed the responses to the question regarding whether a decline in soil productivity had been observed (1=Yes, 0=No) were plotted against the age of the pasture. The results, as shown in Figure 2, demonstrate that, contrary to expectation, little discernible correlation between a yes answer and older pasture. For the 94 observations obtained for both variables, positive and negative answers to the question are spread evenly along the age axis.

Figure 2. Age of pasture versus observation of deterioration of land productivity

0 = degradation observed, 1 = degradation not observed

In a second correlation analysis, an attempt was made to correlate total productivity of each holding with the age of pasture. Again, the expectation was that the correlation would be negative, that is, the older the pasture the lower the productivity. Figures for annual production of both beef and milk were assembled for each holding. In order to put milk and beef into common units, the quantity of milk produced was divided by a factor of three to account for the relative prices of milk and beef. The sample for the correlation analysis consisted of those holdings with production figures that were complete and that had the age of pasture specified. For the entire sample (N=59) the correlation coefficient is 0.013. In other words the correlation was positive and extremely weak. Further to this analysis, the sample was divided by type of production. Interestingly, both dairy and dual purpose sub-samples (N=12 and 17 respectively) showed similar weak but positive correlation. However, the ranching sub-sample, making up by far the majority of holdings (N=32), showed a degree of negative correlation with a coefficient of -0.4.

Generally, the levels of agrochemical and feed inputs are much lower on beef ranches than on dairy or dual purpose holdings. The results of the correlation analysis suggests that in order to support milk production the former operations may be substituting other inputs such as fertilizer and feed for natural productivity, thus dampening any effect that age of pasture may have on productivity. Meanwhile, on extensive ranching operations such inputs are not used and thus a decline in productivity is noted over time. Given this suggestion the graphing exercise presented in Figure 2 was repeated for the ranching sample, however, once again no land productivity had been noted and the age of pasture (N=49).

Production Function Analysis

The discussion above highlights the difficulty of relying on a simple correlation analysis as relationship was observed between positive responses to the question of whether a decline in evidence of the role of age of pasture in soil productivity. If such a relationship holds it must hold on all holdings not just ranching holdings. In order to properly assess the relationship, then, the development of a statistical relationship between age, *S*, and annual production, *Q*, needs to account for other inputs, $\mathbf{X}_{\mathbf{i...N}}$, that may be affecting productivity. In order to accomplish this a production function was developed for estimation as follows:

$$Q = \beta_0 S + \beta_i X_i + \beta_{i+1} X_{i+1} \ldots \beta_N X_N \qquad \text{Eq. 14}$$

An intercept coefficient is not included as it is assumed that in the absence of inputs there would be no production of Q (although clearly there would be natural production of biomass). A simple additive, linear form is chosen given the exploratory nature of the analysis.

Due to the data requirements of the estimation, the sample is limited to those holdings that provided usable information on both the census form (production and age of pasture) and the variable input section of the cost questionnaire. A range of input variables was selected for their likely effect on productivity. These included labour (family and salaried), fertilizer, feed concentrate, molasses, salt, mineral salt and pecutrin (minerals). A number of other variables (including vitamins, antibiotics, de-worming medicines) were not included principally due to problems of standardising input quantities given the wide range of product brands employed. Correlation analysis was then used to assess which variables were likely to be good predictors of production and to assess multicolinearity between different inputs. Finally, a series of step-wise regressions were run to arrive at the best mix of independent variables. Choosing a significance level of $P<0.05$ for the independent variables led to the following production function:

$$Q = \beta_1 L + \beta_2 C + \beta_3 M + \beta_4 S + \beta_5 F + \varepsilon \qquad \text{Eq. 15}$$

Labour, feed concentrate, molasses, salt and fertilizer respectively, all had positive and significant coefficients (as seen in the figure below). As the p-value for fertilizer is only slightly over 0.05 it is left in the equation, lending the equation slightly more predictive power than without fertilizer. The R-Square measures indicate that despite having only 43 cross-sectional observation the production function explains approximately 80% of the variation in production.

Figure 3. Estimation of production function using variable inputs

Regression Statistics		ANOVA					
Multiple R	0.91		*df*	*SS*	*MS*	*F*	*Signif. F*
R Square	0.83	Regression	5	2,009,304	401,861	37	2.12E-13
Adjusted R Square	0.78	Residual	38	413,697	10,887		
Standard Error	104	Total	43	2,423,000			
Observations	43						

	Coefficients	*Standard Error*	*t Stat*	*P-value*
Labour	6.18	0.89	6.97	2.68E-08
Molasses	1.93	0.32	6.08	4.42E-07
Concentrate	0.35	0.06	5.43	3.46E-06
Salt	1.35	0.54	2.51	0.0165
Fertilizer	0.00	0.00	1.99	0.0537

Clearly, age of pasture does not appear in the final production function. Although its coefficient remained fairly stable at 0.81 during the early runs, the age variable is dropped from the estimation in the last step as it has the highest p-value (0.15). Nevertheless, there exists reason to attempt to reinstate it. A certain amount of colinearity between the age and fertilizer variables exists (0.37) and both fertilizer and salt can, along with age, be considered borderline variables as they also had a $P>0.05$ up to the final regression displayed above. When age is substituted for both fertilizer and salt in the above equation, age attains the required level of significance (see the figure below). As the estimated equation loses only a very minor degree of predictive accuracy the usefulness of the new estimation is essentially the same as the earlier production function. In fact, re-running the estimation with only labour, feed concentrate and molasses lowers the R-Square to 0.78 revealing that adding age (or salt and fertilizer) to the equation does not greatly improve its predictive accuracy.

Figure 4. Estimation of production function with age variable

Regression Statistics		ANOVA					
Multiple R	0.90		df	SS	MS	F	Sign. F
R Square	0.81	Regression	4	1,956,061	489,015	41	2.85E-13
Adjusted R Square	0.77	Residual	39	466,940	11,973		
Standard Error	109	Total	43	2,423,000			
Observations	43						

Variables	Coefficients	Standard Error	t Stat	P-value
Labour	5.61	1.07	5.23	6E-06
Molasses	1.99	0.34	5.88	7.5E-07
Concentrate	0.39	0.06	6.24	2.4E-07
Age	1.72	0.74	2.32	0.0256

Despite having developed a production function that includes age as a significant independent variable, the outcome remains the opposite of that expected. The coefficient of the age variable is positive (1.72). This result runs counter to the intuition that as pasture ages and soil depth diminishes due to erosion that productivity would also be gradually lost. Attempts to generate a production function solely for the beef ranches contained in the sample fail to produce an equation of predictive value and, even then the sign is positive for age. It should be noted that due to changes in the sample size, the groups of ranching holdings included in the regression analysis did not have the same strong negative correlation between production and age (it was -0.03 for the regression sample). Grouping dairy and dual purpose ranching together does lead to a useful production function that includes labour, feed concentrate and molasses. Age, however, is not a significant variable in the equation and is of positive sign in any case. Finally, as the mean age of pasture of the ranching and combined dairy and dual purpose samples are both 33 year, it is not possible to suggest that the results reflect some inherent difference in the type of land being used under the different livestock production types.

Despite the rather small sample, then, the development of a simple production function of relatively high predictive power is possible given the data collected for the cost-benefit analysis. Including age of pasture into the equation as a proxy for cumulative soil exhaustion does not, however, provide any empirical substantiation for the claim that soil erosion is affecting productivity yields on Río Chiquito livestock holdings. Thus, there is little basis for arriving at a user cost of soil erosion to impose on the straight-line projections of production employed in the cost-benefit analysis of livestock production.

Aside from stressing the productive potential of the deep volcanic soils that exist in the watershed, there are three other potential explanations of the positive relationship observed between older pasture and productivity. The first is that pasture often may take many years to be physically cleared of the remains of the deforestation process. Observation in the field and interpretation of aerial photography show that in areas of more recent colonisation in the mid- to upper watershed considerable debris remains on the land and pasture is punctuated with remnant trees and forest fragments. The length of time required to complete the clearing process might therefore affect the natural productivity and accessibility of pasture when examined on a per hectare basis (when the hectares include incompletely cleared areas). A related hypothesis is that the process of decay and liberation of subsoil organic matter (or above ground matter if the land is not cleared) may take more than just a few years. In this case farms and ranches entering into their second, third or fourth decade may still be in the process of consuming this natural stock of fertility. Arguing against this position is that owners of both old and newly cleared holdings report observing a deterioration of land productivity.

A second potential explanation is that the analysis suffers from a selection effect in that older sites that have experienced a severe loss in productivity and have already been abandoned. In that case they would not in the observed sample and the results would be biased. In this case the relatively small amount of watershed area in regeneration (7%) as opposed to area in pasture (56%) suggest that the potential for a selection effect is limited.

The more problematic criticism of the analysis is that the observed relationship is faulty since the best sites may have been selected and cleared first, thus biasing production results in the observed sample in favour of older sites. Given this latter point, the analysis should be taken as of an exploratory nature, rather than as conclusive evidence. Nevertheless, it is worth noting that no evidence was found that the effects of engaging in livestock production on steep hillsides in an area with a land use capacity rating for absolute protection or forestry production are as rapid nor as pervasive as expected. In sum, the analysis suggests that it is unlikely that massive, spontaneous abandonment of pastureland is likely to occur in the Río Chiquito watershed out of concerns over soil productivity. Achievement of a conservation agenda for the watershed is therefore likely to require intervention, whether of a regulatory or economic nature.

Valuation of Off-Site Hydrological Impacts

In this section the analysis moves beyond issues of production on the livestock holdings in Río Chiquito to the external effects of land use on downstream hydrology and, subsequently, downstream economic production. The objective is to assess the economic impact of these changes in hydrological function with respect to pasture as opposed to forest on both hydroelectric generation and irrigation, the principal downstream use of water originating from the Río Chiquito watershed.

Aylward *et al.* (1998) effectively debunk the notion that land use in Lake Arenal is likely to affect dry season production under the Arenal-Tempisque Irrigation Project (PRAT) in the foreseeable future. Current supply of water from the reservoir is far more than is required by the existing infrastructure. Nor is it clear whether or when PRAT will be completed. Second, it remains unclear why the national water services estimates of water requirements for the project are double those calculated by the CREED Costa Rica project (Allen 1995). Finally, and most importantly, the reliance on hydropower from Arenal during the dry season implies that the actual water release curve for the reservoir would coincide with that of the irrigation project. As there is little reason to believe that changes in land use in Río Chiquito and subsequent effects on water storage and availability in Lake Arenal would lead to changes in productivity of the irrigation project, consideration of externalities is limited to those that impact hydroelectric production.

The following section presents background material, a model for valuing hydrological externalities, an algorithm for calculating such externalities and the data and results of the application of this algorithm in Arenal and Río Chiquito.

Background

As an introduction to the valuation of off-site hydrological impacts, the linkage between land use and hydrology is summarised with reference both to the generalised views on the processes and effects involved and the available evidence regarding these effects in Arenal. The literature in which economic analysis is used to value these downstream effects is reviewed briefly, before turning to the valuation model used in the Arenal case.

Literature Review: Land Use and Hydrology

Disturbance of tropical forests can take many different forms, from light extraction of non-timber forest products through to wholesale conversion. Each type of initial intervention will have its own particular impacts on the hydrological cycle. In assessing the hydrological impact of land use changes many experts have also emphasised the importance of considering not just the impacts of the initial intervention but the impacts of the subsequent form of land use (Bosch and Hewlett 1982; Bruijnzeel 1990; Calder pers. comm. 1995).

For the purposes of this paper these hydrological impacts are divided according to whether they relate to water quality or water quantity. Using this typology, erosion, sedimentation and nutrient outflow are grouped together under the heading of water quality impacts; and changes in water yield, seasonal flow, stormflow response, groundwater recharge and precipitation are considered as water quantity issues. The general nature of the hydrological impacts of changes in land use and conversion of tropical forests can be summarised based on Bruijnzeel (1990), Calder (1992), and Hamilton and Pearce (1986):

- Erosion increases with forest disturbance, at times dramatically, depending on the type and duration of the intervention.

- Increases in sedimentation rates are likely as a result of changes in vegetative cover and land use and will be determined by the kind of processes supplying and removing sediment prior to disturbance.

- Nutrient and chemical outflows following conversion are generally negative as leaching and removal of nutrients and chemicals is increased.

- Water yield is inversely related to forest cover, with the exception of cloud forests where horizontal precipitation can compensate for losses due to evapotranspiration.

- Seasonal flows, in particular dry season baseflow, may increase or decrease depending on the net effect of changes in evapotranspiration and infiltration.

- Stormflow may increase if hill-slope hydrological conditions lead to a shift from sub-surface to overland flows, although the effect (flooding) is of decreasing importance as the distance from the site and the number of contributing tributaries in a river basin increase.

- Groundwater recharge is generally affected in a similar fashion to seasonal flows.

- Local precipitation is probably not significantly affected by changes in forest cover, with the possible exceptions of cloud forests and large basins (such as the Amazon).

Finally, the authors cited above generally agree that the type of management strategy employed may be even more important than the general category of use to which the land is put in the determination of the nature and extent of hydrological impacts (both spatially and temporally).

Literature Review: Hydrological Impacts in Arenal

In the case of Lake Arenal, the effects of the conversion of forest to pasture have largely been framed in terms of an increase in sedimentation of the lake. This topic has been discussed and debated at the national level for the past twenty years. During this time a number of studies have been conducted in an attempt to link land use practices in the Arenal watershed with sedimentation of the Arenal Dam (CCT 1973; CCT 1980; Matamoros 1988; ACA 1993). These studies supported the claim that the negative effects of land use in the upper Arenal watershed could have a substantial impact on the dam's capacity to generate hydroelectricity. The subsequent design and partial implementation of the Arenal-Tempisque Irrigation Project using the same water source raised additional concerns in this regard.

CCT (1980) suggested that erosion and sedimentation rates were on the order of 100 tons/ha/yr from pasture. Meanwhile, sampling data compiled by the Costa Rican Electricity Institute (ICE) report an average sediment delivery of 2.4 tons/ha/yr for the entire Río Chiquito watershed (see Hydroconsult A.B. 1993). Given that over half the watershed is pasture, these figures are very much at odds. More recently Hydroconsult A.B. (in collaboration with ICE) initiated a study in Arenal which included an in-depth re-examination of the issue of sedimentation in Lake Arenal. The results of this inquiry suggest suspended sediment yields for Río Chiquito of from 12 to 24 tons/ha/yr (Jansson 1996), which is up to ten times as much as originally estimated by ICE.

At the same time the CREED Costa Rica project sponsored a number of studies related to land use, erosion and sedimentation in Arenal. Vásquez and Rodríguez (1995) used the Universal Soil Loss Equation (USLE) to calculate surface erosion in the Río Chiquito watershed and conducted a field study to assess likely erosion yields (for comparison with the simulated figures from the USLE). In a study of rural watersheds in Costa Rica, Calvo and Quirós (1996) conducted a statistical investigation of the determinants of suspended sediment yields, later applying the resulting equation to generate estimates of sediment yields in the Arenal watershed. Building on the information developed in previous studies, Saborío and Aylward (1997) calibrated the CALSITE (Calibrated Simulation of Transported Erosion) program using data for the Río Chiquito and Caño Negro watersheds. The CALSITE program calculates simulated erosion using the USLE and then applies a sediment transport model to generate estimates of delivered sediments. Because it is integrated with IDRISI, a geographic information system software package, CALSITE enables the user to effectively determine the suspended sediments generated by each plot or cell in the watershed.

A review and comparison of these studies leads to the determination of the likely level of erosion and sedimentation in Arenal and the Río Chiquito watershed (Aylward *et al.* 1998). The studies can be grouped into three categories. First, the early TSC studies, providing an upper bound on erosion estimates. Second, the studies based on ICE data, providing a lower bound on sediment yield. And thirdly the subsequent set of results from the CREED Costa Rica and Hydroconsult work, supporting an intermediate estimate on both erosion and sediment yield. Thus, it should be expected that while not as extreme a problem as originally envisaged by the original TSC study, erosion and sediment yield from pasture are much more important than suggested by previous ICE reports (in the range of 40 tons/ha/yr and 20 tons/ha/yr respectively).

Thus, the process of continuing to work towards more rigorous estimations of erosion and sediment yield has led to a narrowing "confidence interval" as additional studies have been undertaken. It is also true that the degree of specificity has improved over time as computer applications have enabled analysts to distinguish between an increasing array of land use types. For the purpose of this paper, however, the most intriguing development is the ability to actually employ relevant biophysical information in determining the quantity of erosion generated by a particular plot that will actually arrive at the mouth of the watershed on an annual basis (Saborío and Aylward 1997).

With respect to issues of water yield and timing of delivery of such yield, there exists a general consensus (as cited above) that changes in vegetative cover from forest to pasture will lead to an increase in water yield, or annual run-off. The literature also reveals that, a priori, it will be difficult to specify what the effect of such a change in cover will be on seasonal flows and stormflow. Given the existence of a large reservoir at the mouth of Río Chiquito, the issue of the timing of water delivery during a single storm event is of little interest in this paper. Instead, the crucial water quantity issues involve an assessment of how much water will arrive in a year, and how that quantity will be divided up over the seasons, dry and wet. In particular, it is interesting to know the difference in water yield and regulation that would be obtained under forest and pasture, the predominant uses in the watershed. In addition, given the presence of significant areas of cloud forest in the watershed, an understanding of how evapotranspirative processes in these forests is also of interest.

Unlike the case with erosion and sedimentation, little previous work exists on these topics in the Arenal watershed (CCT 1980; Zadroga 1981). As part of the CREED Costa Rica project two separate studies were conducted on these issues. Calvo (1996) explored a number of issues related to water yield and regulation at the watershed level, but was unable to link land use change to changes in water yield or seasonality of flow given a lack of sufficient data. Nor is it clear that improved data would help given that the presence of both pasture and cloud forest in the upper watersheds of Arenal act to complicate the process of teasing out the effects of land use change or the extent of horizontal precipitation at the watershed scale. A change to pasture would typically be expected to add to water yield. However, in the upper watersheds of Lake Arenal this effect will depend very much on where the land use change occurs (in cloud forest or not) and in what manner it occurs (the clearing of all forest or the establishment of a fragmented forest).

Fallas (1996) carried out a yearlong field experiment aimed at detecting a difference in interception rates between different forest types in an area of cloud forest. The results suggest very large differences in interception rates between fragmented and non-fragmented forest, and between high and low forest types in the cloud forests of Río Chiquito. In the experiment, fragmented primary forest produced 800 mm more in net precipitation than the high primary forest (the amount reaching the ground). This amount represents a capture of 460 mm more in net precipitation than received in pasture. Thus, the results support the contention that cloud forest may capture considerable amounts of horizontal precipitation, however, they also indicate that intact primary forest is not the best collector in this regard.
In sum, experience suggests the difficulty of actually looking for evidence of water quantity effects (and by inference water regulation effects) at the watershed level, but points to important micro level processes that demonstrate the importance that land use may have on evapotranspiration and, subsequently, on run-off. Unfortunately, this implies that the only options for developing an understanding of how the pattern of land use change affects water quantity issues are to extrapolate from micro studies or use results obtained more generally (or from specific-sites). These approaches are explored further later in this paper in order to arrive at a quantitative estimate of the difference in evapotranspiration between forest and pasture in Río Chiquito.

Literature Review: Valuation of Hydrological Externalities

Conventional wisdom suggests that "deforestation" leads to a loss of the soil and water "conservation" services provided by natural forests. The downstream economic consequences are generally held to be costly sedimentation of downstream hydropower, water supply and irrigation facilities; an increase in flooding; and a decrease in dry season flow for agriculture and other economic activities. The summary of the hydrological literature provided above suggests that the relationship between land use, hydrological function and flooding is localised at best, and that the effect on dry season flow may be positive or negative depending on site conditions. Further, the hydrological literature clearly states that annual water yield, or run-off, will increase with the removal of forest cover.

Before turning to the economic literature on this topic, it is important to stress, that in all likelihood, it is not possible to generalise regarding the net effect of these hydrological impacts on economic activity. While sedimentation effects may be presumed to be largely negative a number of the downstream economic activities affected by land use change are likely to respond positively to increases in water yield (e.g. water supply, hydropower, navigation and productive land uses). Others will not be affected or affected negatively depending on site-specific characters of both biophysical impacts and economic activities. For example, if stormflow does increase in a particular site the extent of the local impact of flooding will depend on the concentration of vulnerable economic activities in the area affected. Similarly, if changes in land use lead to reduced infiltration opportunities that exceed gains in net precipitation the ensuing reduction of dry season base flow may lead to important negative local effects on agricultural production and water supply in rural areas.

With regard to off-site sedimentation costs in tropical moist forest environments a number of studies exist including (by country) as follows: Cameroon (Ruitenbeek 1989); Costa Rica (CCT 1980; Rodríguez 1989; Kishor and Constantino 1993); the Dominican Republic (Veloz *et al.* 1985); Ecuador (Southgate and Macke 1989); Malaysia (Mohd. Shahwahid *et al.* 1997); Panama (Intercarib S.A. and Nathan Associates 1996); Philippines (Briones 1986; Cruz *et al.* 1988; Hodgson and Dixon 1988); and Indonesia (Magrath and Arens 1989).[8] The valuation approach typically employed is a measure of productivity losses or damage costs. In a number of cases, significant changes in productivity or damage costs have been demonstrated. For example, in Ruitenbeek's (1989) valuation of the Korup Project in Cameroon, the benefits from watershed protection were estimated to be almost half of the direct conservation benefits.

More recently a number of studies have provided results at odds with these earlier studies. In summarising some of the difficulties in undertaking valuation of hydrological externalities, Enters (1995) reports on the case of Thailand were extraction of sediment from streams and canals suggests that sedimentation may also produce positive externalities. In Malaysia, Mohd Shahwahid *et al.* (1997) examined sediment effects on run-of-stream hydroelectric plants and treated water production. The results indicated that a program of reduced impact logging would have essentially no effect on water supply and would lead to only a minimal disturbance of hydropower generation. In other words, the gains from logging could easily compensate for the losses incurred by the hydroelectricity producer.

As compared with studies of the effect of sedimentation, few studies were found of the economic consequences of changes in land use on water yield or seasonal flows in the literature on developing countries. A temperate study from the United Kingdom, however, suggests the importance of such studies. Barrow, Hinsley and Price (1986) examined the effect of afforestation on hydroelectricity generation in the Maentwrog catchment in Wales and in 41 catchments in Scotland. Both analyses indicated that the increased evaporation under reforestation (in comparison with grazing) lead financially marginal sites (for forestry) to become financially sub-marginal once hydropower losses are included into the analysis. While there was some variation in results depending on site conditions, the example clearly shows the changes in productivity that afforestation may bring to a watershed in use for hydropower.

[8] For a review of these studies and more, particularly those examining downstream effects on large hydroelectric reservoirs, see Aylward (1998).

More recently, a number of studies undertaken concurrently with the CREED Costa Rica project have examined issues of water yield and regulation in economic terms. A study in Guatemala assessed the role of cloud forests in supplying dry season irrigation water in the Sierra de las Minas Biosphere Reserve of Guatemala (Brown *et al.* 1996). A number of studies in southern Chile examined the relationship between watershed management projects (including timber harvesting), and water yield and flood control effects (Alvarez *et al.* 1996; Vera *et al.* 1996). These studies have had mixed results. Brown *et al.* (1996) found that cutting of cloud forest would adversely affect dry season irrigation while Alvarez *et al.* (1996) suggest that thinning of forest in the Magallanes National Reserve would provide not only timber benefits but net hydrological benefits, principally due to increases in water supply.

Increasingly, then, analysts are beginning to question the general assumption that the returns to watershed protection are always positive. This development is also beginning to be expressed in more general analytical or review pieces (Chomitz and Kumari 1996; Enters 1998).

Methods for Valuing Hydrological Externalities

Estimation of the off-site costs of sedimentation have typically employed an approach to valuation known either as the “changes in productivity approach” or the “damage function approach.” These approaches involve both a biophysical and economic component. First the hydrological effects of the change in land use (or with- and without project scenarios) are quantified. These “changes in production” are then valued using the costs of alternatives, particularly thermal power. Application of these methods to the valuation of hydrological externalities in the case of large hydroelectric reservoirs suffer from three general limitations:

1. The studies are incomplete insofar as they typically consider only one of the hydrological functions that is affected by changes in land use and that may influence hydroelectric production (i.e. sedimentation).

2. The studies appear to be inconsistent in terms of how they confront the following issues: the relative importance of live and dead storage impacts; the definition of project life; and the method employed to derive the unit value applied to the productivity change.

3. The studies fail to consider the manner in which reservoir operation might be optimised over time in response to hydrological changes, in particular within the context of the local power generating system and under different hydrological conditions.

In addition a number of the studies suffer from poor or unclear assumptions. Most notably, none of the studies provide an explicit model that links land use change to welfare change. Typically, land use and hydrological impacts are specified according to the hydrological model employed. Occasionally a net benefit equation specifying the costs and benefits to be evaluated over the planning horizon is provided. However, the linkage between the two is typically inferred rather than made explicit. To counter these difficulties the following model and algorithm was developed by Aylward (1998) for valuing hydrological externalities.

A Model for Valuing Hydrological Externalities: Large Hydroelectric Reservoirs

The objective of the valuation exercise is to better understand the sign and magnitude of the hydrological externalities across different land units in the Río Chiquito watershed. There are four relationships that define the valuation problem in the case of hydrological externalities as they affect large hydroelectric reservoirs:

4. The relationship between land use and hydrological function (Land Use and Hydrology).

5. The relationship between hydrological function, water storage and water utilisation (Reservoir Operation).

6. The relationship between water, other inputs and hydroelectric power generation (Production Function).
7. The relationship between power generation from a given reservoir, alternative generating sources and the demand for hydroelectric power (Marginal Opportunity Costs).

Each of these is covered below in constructing a model that links land use to the marginal opportunity costs of generated power. Once this model is defined an algorithm may be developed for evaluating how changes in land use lead to changes in the marginal opportunity cost (MOC) of power generation.

Land Use and Hydrology

In the case of hydroelectric reservoirs, the hydrological impact of land use change is likely to be felt through three hydrological functions: sediment delivery, water yield and, potentially, the seasonal timing of water yield. For the purposes of identifying the relationship between a particular land use, *L*, and the production of these hydrological outputs it is necessary then to link individual land use units, *i*, within the watershed to the run-off, *R*, and suspended sediment, *SSY*, produced by means of the hydrological functions. A land use unit is simply an area on which the same land use is practiced.

The approach taken here is simplistic given the long chain of relationships that must be developed and the wide variety of methods for interpreting these relationships. The model focuses principally on the linkage between *L* and the hydrological outputs. Hydrological function in a given situation will vary with land use and with other site- and watershed-specific biophysical characteristics. The vector $\mathbf{X}_j$ denotes these characteristics, where *j* refers to the number of characteristics relevant to run-off. For example, run-off will be determined by precipitation and the infiltration rate at the site, as well as the topography and geology between the land use unit and the water channel. Run-off is then presented in general terms as a relationship between land use and this vector of biophysical characteristics:

$$R_{ij} = \mathrm{R}\left(L_i, \mathbf{X}_j\right) \qquad \text{Eq.16}$$

Suspended sediment yield is in turn determined not just by land use but by site- and watershed-specific factors such as erodibility, erosivity and topography. In addition, suspended sediment yield is itself largely determined by the quantity of run-off. One of the simplest methods for estimating *SSY* are sediment rating curves which relate differing levels of run-off to different sediment concentration levels. In fact, from a measurement point of view, suspended sediment yield is included in readings of run-off. As sediment concentrations are on average extremely low, this definitional problem is assumed away by taking R and SSY as separate volumes. The suspended sediment yield function can thus be specified as follows:

$$SSY_{ijk} = \mathrm{SSY}\left(L_i, R_{ij}, \mathbf{Y}_k, \alpha_i\right) \qquad \text{Eq. 17}$$

where the vector Y represents the relevant biophysical characteristics. The weight to volume conversion factor, α, allows SSY to be expressed in the same units as R (m^3). If compaction of soil occurs due to changes in land use, then α may also vary with land use: $\alpha_i = \alpha(L_i)$ subject to $\partial\alpha/\partial L < 0$.

The total run-off produced by a given watershed composed of n land use units and with m and o biophysical characteristics determining run-off and sedimentation, respectively during a given time period t would then be:

$$R_t = \sum_{i=1}^{n}\sum_{j=1}^{m} R_{it} \qquad \text{Eq. 18}$$

Similarly, the total suspended sediment yield delivered to the reservoir is:

$$SSY_t = \sum_{i=1}^{n}\sum_{j=1}^{m}\sum_{k=1}^{o} SSY_{ijkt} \qquad \text{Eq. 19}$$

However sediment delivery includes both suspended sediment and bedload. The bedload is that portion of the sediment that is not in suspension, but is instead traveling on the bottom of the stream channel pushed by the flow of water and pulled by gravity. Total sediment delivered to the reservoir at time t will be composed of bedload, B, and suspended sediment:[9]

$$SY_t = B_t + \sum_{i=1}^{n}\sum_{j=1}^{m}\sum_{k=1}^{o} SSY_{ijkt} \qquad \text{Eq. 20}$$

In sum, the modeling of the change in each of these hydrological functions may be conducted as a function of land use taking site- and watershed-specific biophysical factors into account. As noted earlier it is clear that the removal of vegetation tends to increase both sediment and annual water yield, while having an indeterminate effect on seasonal flows. The latter occurs as seasonal flows, in particular dry season baseflow, may increase or decrease depending on the net effect of changes in evapotranspiration and infiltration associated with the subsequent land use. Taking an increase in L as a change away from undisturbed natural vegetation allows a formalisation of the relationship between land use and sediment yield:

[9] It is assumed here that the principal effect of land use is on suspended sediment yield not bedload.

$$\frac{\partial SY}{\partial L} > 0 \qquad \text{Eq. 21}$$

In the case of run-off, while there is no general relationship between land use and dry season baseflow, it is clear that run-off will generally be an increasing function of land use when taken on an annual basis:

$$\frac{\partial R}{\partial L} > 0 \qquad \text{Eq. 22}$$

The exception to this rule might be when horizontal precipitation in cloud forests can compensate for losses due to evapotranspiration. If reference is made to flows over seasonal periods, the first derivative may be either positive or negative, although there is little experimental evidence of a negative relationship, even during the dry season.

In defining the portion of delivered sediment that comes to rest in the reservoir, S, the trap efficiency, β, is used to select out that portion of the suspended sediment that is not passed on through the offtake of the reservoir:

$$S = B + \beta SSY \qquad \text{Eq. 23}$$

Total sediment delivery in time t can be expressed as the sediment trapped in the dead and live storage, S^D and S^L respectively, and the suspended sediment that is not precipitated and continues downstream:

$$SY_t = S_t^D + S_t^L + SSY_t(1 - \beta_t) \qquad \text{Eq. 24}$$

If the parameter, κ, represents the portion of reservoir sedimentation that is deposited in the live storage, then the delivered sediment distributed to the live storage, and dead storage, can be derived as follows:

$$S_t^L = \kappa(B_t + \beta SSY_t) \qquad \text{Eq. 25}$$

$$S_t^D = (1-\kappa)(B_t + \beta SSY_t) \qquad \text{Eq. 26}$$

Reservoir Operation

This component of the model must simulate how changes in hydrological function affect discharge levels over time (and hence production). The dynamics of reservoir operation may be described by the following discrete difference equation, rearranged so as to isolate the productive discharge from the reservoir in time (Christensen and Soliman, 1988):

$$D_t = V_{t-1}^L - V_t^L + I_t - W_t \qquad \text{Eq. 27}$$

This equation reflects the intuition that the volume of water, V^L, stored in the live storage area of the reservoir at the end of the period t will be equal to the volume at the end of the last period adjusted for the inflow during the period from streams entering the reservoir, I, and outflows that occur either as discharges through turbines for power generation, D, or simple spillage, W (i.e. that does not generate power).[10]

Employing an overbar to represent maximum capacity, an underbar to represent minimum capacity and placing subscripts as appropriate for those variables that change over time, then the operational reservoir constraints are as follows:

$$\underline{V}^D \leq V_t^D,\ \underline{V}^L \leq V_t^L \leq \overline{V}_t^L,\ \underline{D} \leq D_t \leq \overline{D} \qquad \text{Eq. 28}$$

Land use is related to three of the elements that determine reservoir operation: inflows, live storage capacity and dead storage capacity. The inflow during a given period can then be defined to equal the run-off plus displaced dead storage capacity:[11]

$$I_t = R_t + S_t^D \qquad \text{Eq. 29}$$

The continued deposition of sediment in the live storage will have the effect of reducing the live storage capacity at the end of each time period:

$$\overline{V}_t^L = \overline{V}_{t-1}^L - S_t^L \qquad \text{Eq. 30}$$

At the same time, continued sedimentation of the dead storage causes the dead storage volume to decrease over time:

$$V_t^D = V_{t-1}^D - S_t^D \qquad \text{Eq. 31}$$

Production Function

Water destined for use in hydroelectric power generation at a given reservoir is a fixed factor of production. An increase in current expenditure on other variable inputs such as labour, equipment or maintenance does not have a significant impact on output in the short run. The marginal cost of production to the producer is zero (given that the producer typically does not pay for the water input). The production function for hydroelectricity from the reservoir, therefore, depends solely on the water input.

The physical relationships that govern the generation of hydroelectric power suggest that the two factors determining the production function are the quantity of water discharged, D, and the height of the discharge (otherwise referred to as the "head"). Where the fluctuation in reservoir height is limited in comparison to the head, this factor is of less significance and the production function for power, G, generated by an hydroelectric power plant is:

[10] For the purposes of examining the effects of changes in hydrological outputs on this equation, it is useful to clarify that in the above equation V actually refers to the live storage volume because a hydroelectric reservoir will never be drawn down lower than its outtake level. It is useful then to distinguish between the live and dead storage volumes using the respective notation: V^L and V^D.

[11] The sediment arriving in the dead storage in a given period can be interpreted as a gain in inflow during that period as it will displace an equal volume of water upwards into the live storage area.

$$G_t = \gamma D_t \quad \text{Eq. 32}$$

The parameter γ is the water conversion factor, which reflects the average power generation (kWh) per unit of water discharge (m^3). This factor is typically regarded as a constant for a given hydroelectric power (HEP) plant. It may be obtained from the technical rating of the equipment (turbines) in use or by estimating the relationship from observed data.[12]

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[12] See Christensen and Soliman (1988) for the functional relationship when power generation is a function of head (and therefore storage).

Marginal Opportunity Costs

Production of electricity from other sources is a perfect substitute for electricity produced by a hydroelectric reservoir. As a result all that is required in order to value the change in production is an understanding of the marginal rate of substitution between the two types of produced electricity and the cost of the appropriate alternative. Thus, the marginal opportunity cost of generated power will depend on the appropriate marginal rate of substitution, μ, the amount of power generated and the marginal cost to the system of producing this power, *MC*. The marginal cost will depend either on the costs to the system of meeting a reduction in power from the facility by employing alternatives, such as thermal power or other hydroelectric power sources; or the costs of unmet secondary power demand:[13]

$$MOC_t = \mu G_t MC_t \qquad \text{Eq. 33}$$

In the short run, the running costs may be taken as the relevant measure of marginal costs. In the long run both running and any capital costs will be the correct measure. In theory, the high short-run costs of any power shortages resulting from land use and hydrological change will be lessened by the introduction of new power supply capacity.

The model serves to illustrate that the effect of a change in land use on discharge, production and MOC will not be unambiguously positive or negative. With both the run-off and sediment yield functions increasing with land use, but having opposing effects on discharge, there is little grounds for assuming *a priori* that, for example, deforestation must have negative effects on hydroelectric production from a large hydroelectric reservoir.

The Algorithm

The model developed above links land use to power generation and the marginal opportunity cost to society of the power. In order, then, to calculate the welfare change due to hydrological change of alternative land uses, the change in MOC that follows on a change in land use must be evaluated.

Ideally, discharge should be optimised so as to minimise total costs incurred by a hydrothermal electric power system (HTEPS) given existing system design and demand, as well as expected expansion of system design and demand. If provided with information on the stochastic nature of inflows, this would be a problem in dynamic optimisation. The optimisation would then be undertaken under alternative land uses. The difference in system costs under two differing scenarios would represent the MOC of the change in productivity of the reservoir due to the change in land use. Apart from the complexity of such an approach, a practical limitation to the analysis of this problem is the likelihood of a lack of empirical data on the temporal pattern of run-off and, hence, suspended sediment yield, under alternative land uses. Sophisticated spatially distributed process models that model both run-off and sediment delivery might be used with historical precipitation records to generate stochastic data that is at least partially empirically based. A more straightforward approach, would be to simulate potential temporal patterns of run-off and sedimentation.

[13] Secondary power is produced only when base and peakload contracts are met and is, therefore, unreliable and contracted to particular customers without a guarantee of supply.

The latter intuition suggests a method for simplifying the problem further, while still capturing the essential variability in discharge and marginal opportunity costs that can be expected due to varied hydrological conditions over short and long-term frames. A deterministic simulation model, or algorithm, can be developed based on the simulation of the expected response of the state of the reservoir and the state of the hydrothermal electric power system to varying hydrological conditions. This analysis may be based on historical data and future expectations regarding how the HTEPS and reservoir are optimised under different hydrological conditions. As a result, it is not necessary to undertake a comprehensive evaluation of the model presented above over both land uses. Rather, once the hydrological conditions are known for a given period and the state of the reservoir and HTEPS is fully specified, changes in hydrological outputs can be used to derive changes in the reservoir discharge equation for that period. With these states clearly specified it is also, then, a straightforward matter to derive the marginal opportunity cost of changes in discharge and production.

The simulation is conducted on a yearly basis, *t*, with each year divided into a number of seasons, *s*. Here it is assumed that there are just two seasons: dry and wet. In order to find the appropriate welfare measure for changes in productivity, the planning horizon is further divided according to whether it is the short run or the long run. The algorithm consists of three steps repeated for each *st* of the evaluation period: (1) calculation of period values for the hydrological variables, (2) evaluation of changes in productivity and (3) calculation of the per unit marginal opportunity cost of this change in productivity. Once the algorithm is completed, stored values are summed and discounted to yield the net present value of the externalities for the particular *jk* combination of land use unit characteristics.

Hydrological Analysis

The first step is to acquire the expected change in run-off and sedimentation of the live and dead storage volumes with respect to the change in land use. In order to trace back the impacts to land units with different combinations of biophysical characteristics, the *j* and *k* characteristics that determine run-off and suspended sediment yield, respectively are made explicit below:[14]

$$\Delta R_{jst} = R^1_{jst} - R^0_{jst} = \mathrm{R}(L^0, L^1, \mathbf{X}_j) \qquad \text{Eq. 34}$$

$$\Delta SY_{jkst} = \Delta SSY_{jkst} = \mathrm{SSY}(L^0, L^1, R^0_{jst}, R^1_{jst}, \mathbf{Y}_k, \alpha^0_i, \alpha^1_i) \qquad \text{Eq. 35}$$

As noted earlier, bedload is assumed not to vary with land use, thus the change in live and dead storage sediment that occurs with a change in land use is as follows:

[14] In the equations that follow *Δ* is used to represent the change in variables that occurs as a result of moving from the initial land use to the subsequent land use, as shown in the first equation by denoting the former by a superscript 0 and the latter by a superscript 1.

$$\Delta S^{L}_{jkst} = \kappa\beta\Delta SY_{jkst} = \kappa\beta\Delta SSY_{jkst} \qquad \text{Eq. 36}$$

$$\Delta S^{D}_{jkst} = (1-\kappa)\beta\Delta SSY_{jkst} \qquad \text{Eq. 37}$$

From this point forward the *jk* notation is omitted to avoid unnecessary notational clutter. Implicitly, however, the algorithm serves to identify the MOC of productivity changes associated with particular *jk* combinations of the land uses under evaluation. Indeed, this remains the objective of the exercise.

As indicated above, hydrological variables affect reservoir operation in three ways. Continued sediment inflow to the dead storage area causes the dead storage volume to decrease over time implying the need to "check" the dead storage area at the end of each period. In the case of Arenal the large size of the dead storage area relative to sediment inflow precludes the possibility that this area might be filled over a reasonable economic time span.[15] Thus, in the case of Arenal it is necessary only to track the change in inflow and the change in the live storage capacity operating constraint.

The change in inflow caused by a change in land use is:

$$\Delta I_{st} = \Delta R_{st} + \Delta S^{D}_{st} \qquad \text{Eq. 38}$$

When comparing land uses at time *st* there are two measures of change in storage capacity: the cumulative change and the incremental change during the period itself. The cumulative change in storage capacity up to a given *st* is simply equal to the difference in storage capacities under the two land uses. This is equivalent to the negative of the difference in the sum of the sediment received up to that point under both uses:

$$\Delta\overline{V}^{L}_{st} = \overline{V}^{L,1}_{st} - \overline{V}^{L,0}_{st} = -\sum_{t}\sum_{s}\Delta S^{L}_{st} \qquad \text{Eq. 39}$$

This should be distinguished from the incremental change in sedimentation over the course of period *st* which is the difference between the change in live storage capacities under the two uses at the beginning and end of the period:

$$\Delta\overline{V}^{L}_{st\Rightarrow st-1} = (\overline{V}^{L,1}_{st} - \overline{V}^{L,0}_{st}) - (\overline{V}^{L,1}_{st-1} - \overline{V}^{L,0}_{st-1}) = \Delta\overline{V}^{L}_{st} - \Delta\overline{V}^{L}_{st-1} \qquad \text{Eq. 40}$$

The variation in the intertemporal path of the maximum storage capacity under the two land uses will alter reservoir operation, given that this capacity is a constraint on the reservoir operation equation. Whenever the reservoir rises to its changed capacity under the new land use, there is a corresponding change in volume that can be stored to the next period. This water must then be either discharged or spilled.

[15] Aylward (1998) extends the current algorithm to provide for the filling of dead storage.

Changes in Productivity
With the value of these current period changes in hand, as well as that of the change in live storage brought forward from the previous period, $\Delta \overline{V}_{st}^{L}$, it is possible to proceed to evaluate the reservoir operation equation in that period so as to determine any resulting productivity change. This requires the classification of each year according to the nature of its hydrological conditions and the subsequent state of the reservoir and HTEPS, as these values will determine how reservoir operation will be altered in the face of changes in hydrological variables.

Given the complexity and site-specific nature of the problem, only a general approach can be offered for determining changes in productivity. To this end Aylward (1998) takes a systematic approach to specifying the conditions that will determine how changes in hydrology lead to the choice between discharge, spillage and storage. This algorithm may then be combined with simulation of hydrological responses to land use change in order to generate a numerical estimation of the change in productivity. As the approach overlaps that of the process of determining the marginal opportunity costs of productivity changes, the discussion here is germane to both aspects of the problem.

The discharge equation suggests that the effects of changes in run-off and changes in delivery of sediment to the dead storage will be felt through a change in inflow. Given that a change in inflow may affect reservoir operation in one of three ways, the methodological problem is to establish the conditions under which this inflow is subsequently discharged, stored or spilled. As storage is one option, the methodology must also identify how changes in water storage brought forward affect reservoir operation. In the case of changes to the rate of sedimentation of the live storage capacity the effect is felt indirectly through a change in the amount of storage that is carried forward into the next period.

Essentially, then there are two discharge equations, one for each land use. The objective is to find the change in discharge and the change in water stored going forward. The change in discharge can then be converted into a productivity change while the water stored forward may lead to changes in discharge in future periods. In order to do this it is crucial to also identify what happens to spillage although such data is not of any subsequent use. In solving this problem, then, there are six unknowns and two equations. The intuition exploited by Aylward (1998) is to reduce the unknowns to two by clearly specifying the state of the reservoir and the state of the HTEPS at a given point in time. In this fashion it becomes possible to solve the equations. The simulation may then be undertaken for a particular reservoir by charting out the time path of hydrological conditions and specifying how these conditions affect system operation and the state of the reservoir.

The approach is presented with reference to whether the reservoir is in a balanced or blocked state at the end of each decision period. The state of a reservoir is blocked when it is either full or empty and balanced when it is anywhere between these two states. Annual reservoirs may be assumed to be blocked at the end of each season, however, interannual reservoirs may be either balanced or blocked at the end of a season, depending on hydrological conditions and the operation pattern of the reservoir. In the case of an annual reservoir, the reservoir is blocked at the end of each season so the evaluation of productivity changes will occur at the end of each season. This assumes that the annual reservoir is supplemented by an interannual reservoir that buffers the system. As a result, variation in inflows over time simply alters the seasonal discharge from the annual reservoir and it will be filled and emptied seasonally. If this were not the case and the annual reservoir serves as the buffer to the system, then it might not be fully emptied during wet years or filled during dry years. In the case of Arenal, the reservoir is operated in a similar fashion to an interannual reservoir in terms of the cyclical nature of blocked states.

As is clear intuitively, certain operational events that are linked to changes in hydrological outputs may only occur at certain blocked states (i.e. water can only be spilled when the reservoir is full). Meanwhile, balanced states may occur at practically any point in time. As blocked states typically occur only with a minimum periodicity equal to the passing of the seasons, there is therefore little advantage to employing an evaluation period shorter than that of a single season. In the discussion below, the effects on reservoir operation of changes in hydrological inputs is conducted for both balanced and blocked states at the end of each season.

The condition of the reservoir and the HTEPS prior to the change in land use must be specified, taking into account the distinction between the long- and short-run optimization (assumed) of the system. Thus, the analysis of welfare change turns on the simplifying assumption that prior to the change in land use the reservoir is operated in an optimal fashion.[16] A secondary assumption is that the change in hydrological outputs is a marginal one. In the algorithm developed here, there are then twelve possible states the reservoir and the HTEPS may be in at the end of a given evaluation period. These states correspond to the possible combinations of four conditions:

1. Whether the reservoir is blocked or balanced.
2. Whether it is the short run (SR) or long run (LR).
3. Whether the blocked states refer to the reservoir being full or empty.
4. What is the state of the system with regards to the use of alternative power sources and the status of secondary power.

With regards to the fourth condition, three potential states exist: (1) alternative thermal power (ATP) is employed by the system in the short run, (2) alternative power plants (APP) are deployed in the long run and (3) unmet demand for secondary power or power shortfalls exist (SP).

Each of the twelve possible states are discussed in Aylward (1998). As the mathematical manipulations of the discharge equation that underpin the results are straightforward, they are fully demonstrated here only for the first case. In each case the effect of changes in inflow and changes in the sedimentation of live storage are discussed. Given the need to identify whether, and when, a change in storage realised at a previous state alters discharge, it is also necessary to specify what happens to marginal increments in storage volumes from previous periods. Again, these may in principle be stored onwards, discharged or spilled.

[16] If this assumption cannot be made then it becomes very difficult to assess how reservoir operation subsequently will be modified to best accommodate the change in hydrology.

Case 1: Blocked/SR/Full/No ATP and No SP. If there is no unmet demand for secondary power and no ATP is generated and the reservoir is full, a change in inflow will not alter discharge from the reservoir. Any rise in inflow will not be demanded and will simply be spilled. Any decrease in inflow will simply result in a decrease in spillage, again, without affecting discharge. As a result the live storage will also be full under the new land use. The incremental change in sedimentation of the live storage must then also cause a corresponding increase in spillage. The difference between water storage going forward under the two land uses will, however, be equal to the cumulative change in storage capacity between the two land uses. This change in storage is equivalent to the change in water stored forward into the current period plus the incremental change in storage capacity during the period. As a result any change in water storage brought forward that originates from a change in inflows in previous periods must be spilled as well.

This can be shown by specifying the discharge equations for both land uses:

$$D^{0}_{st} = V^{L,0}_{st-1} - V^{L,0}_{st} + I^{0}_{st} - W^{0}_{st} \text{ with } W^{D}_{st} > 0 \text{ and } V^{L,0}_{st} = \overline{V}^{L,0}_{st}$$

$$D^{1}_{st} = V^{L,1}_{st-1} - V^{L,1}_{st} + I^{1}_{st} - W^{1}_{st}$$

If discharge in the two periods is equal, $D^{0}_{st} = D^{1}_{st}$, and the reservoir is also full under the new land use scenario, $V^{L,1}_{st} = \overline{V}^{L,1}_{st}$ then the equations can be combined to yield:

$$(W^{1}_{st} - W^{0}_{st}) = (I^{1}_{st} - I^{0}_{st}) + \left[(V^{L,1}_{st-1} - V^{L,0}_{st-1}) - (\overline{V}^{L,1}_{st} - \overline{V}^{L,0}_{st})\right]$$

If there is no change in sedimentation between land uses and a change in inflow is carried forward into period *st*, the portion of the above equation in square brackets reduces to this change in inflow. In other words this change in inflow from a previous period contributes to spillage, along with the change in inflow during the current period. If there is no change in inflow, spillage will equal the change in storage brought forward minus the cumulative change in storage capacity. If at the end of the previous period the reservoir was also full, the change in storage brought forward will be equivalent to the cumulative change in storage capacity at the end of the previous period. In other words spillage will be equal to the negative of the incremental change in storage capacity during period *st*. Although the change in storage capacity going forward is equal to the cumulative change in storage capacity, spillage increases by only the incremental change in capacity, previous units of displaced water having been spilled in previous periods.

The changes in discharge, storage and spillage in this case can be summarised as follow:

$$\Delta D_{st} = 0,\ \Delta V^{L}_{st} = \Delta\overline{V}^{L}_{st},\ \Delta W_{st} = \Delta I_{st} + \Delta V^{L}_{st-1} - \Delta\overline{V}^{L}_{st}$$

In sum, when in this state there will be no change in discharge or productivity, however there will be a change in the amount of water stored into the next period. The subsequent effect of this change in storage will depend on conditions that apply at the next (and potentially successive) evaluation period. If the three equations above are added together, the change in live storage capacity drops out, leaving the discharge equation as originally formulated. Thus, these equations capture the effect of changes in inflow and changes in sedimentation of the live storage on reservoir operation.

Continuing with the other potential cases it can be shown that the determination of whether discharge, spillage or water storage is altered by a given change in hydrological outputs depends on the sequencing of blocked and balanced states and the system conditions that apply during the period. Based on information about how a particular HTEPS system is optimised in the short and long run, it is then possible to simulate how different hydrological conditions will determine the state of the HTEPS and consequently arrive at the corresponding expected changes in reservoir operation (see **Table 6** and **Table 7**).

Table 6. Effect of change in hydrological outputs on reservoir operation

Conditions (Case Number)	Hydrological Outputs over the Short and Long Run ΔI_t	ΔV_{t-1}^L	$\Delta \overline{V}_{t-1\Rightarrow t}^L$
Blocked Full			
No ATP/APP and no SP (1&5)	ΔW_t	ΔW_t	ΔV_t^L
ATP/APP or SP (2&6)	ΔD_t	ΔD_t	ΔD_t
Blocked Empty			
No ATP/APP and no SP (3&7)	ΔV_t^L	ΔV_t^L	no effect
ATP/APP or SP (4&8)	ΔD_t	ΔD_t	no effect
Balanced			
No ATP/APP and no SP (9&11)	ΔD_t	ΔD_t	no effect
ATP/APP or SP (10&12)	ΔV_t^L	ΔV_t^L	no effect

Table 7. Change in reservoir operation variables under different system and reservoir conditions

Conditions	Short and Long Run ΔD	ΔV_t	ΔW
Blocked Full			
No ATP/APP and no SP (1&5)	0	$\Delta \overline{V}_t^L$	$\Delta I + \Delta V_{t-1}^L - \Delta \overline{V}_t^L$
ATP/APP or SP (2&6)	$\Delta I_t + \Delta V_{t-1}^L - \Delta \overline{V}_t^L$	$\Delta \overline{V}_t^L$	0
Blocked Empty			
No ATP/APP and no SP (3&7)	0	$\Delta I + \Delta V_{t-1}^L$	0
ATP/APP or SP (4&8)	$\Delta I + \Delta V_{t-1}^L$	0	0
Balanced			
No ATP/APP and no SP (9&11)	$\Delta I + \Delta V_{t-1}^L$	0	0
ATP/APP or SP (10&12)	0	$\Delta I + \Delta V_{t-1}^L$	0

Any change in discharge in a given period can then be directly translated from water units (m^3) into power units (kWh) once the conversion factor or equation is known. In the absence of a head effect, the change in generation that arises as a result of a change in discharge is:

$$\Delta G_t = \gamma(D_t^1 - D_t^0) = \gamma \Delta D_t \qquad \text{Eq. 41}$$

The use of the simple water conversion factor alleviates the need to model reservoir volume, all that is need is the change in discharge to arrive at the change in productivity. In the long run it is necessary to understand the capacity equivalent (in kWs) of a given change in production (in kWh) as discussed in the next sub-section.

Derivation of Per Unit Marginal Opportunity Costs

The previous sub-sections explain how a change in land use may lead to a change in hydroelectric power production. To complete the valuation exercise these changes in productivity must be "valued" by deriving the marginal opportunity costs that most appropriately reflect their effect on economic welfare. In order not to falsely overstate this value, an understanding of optimising behavior is essential. Otherwise the least cost alternative or the true marginal opportunity cost of the change in production may not be identified (Ellis and Fisher 1987). This is of particular importance in the valuation of hydrological externalities, given the increasing costs of alternative energy sources and the stochastic nature of inflows.

As with any good, the MOC of hydroelectricity will increase as it becomes relatively more scarce, which suggests that the MOC of the power generated by a reservoir will vary substantially over time as hydrological conditions change. Employing the concept of substitute goods, then, in the short run the unit MOC of the change in power is the difference in marginal costs between the alternative production source and production from the reservoir. Given that the marginal cost of generating another unit of hydroelectricity in the short run is zero, the unit MOC of discharged water will be the savings realised by not paying the marginal costs of production of the displaced alternative. In order to avoid confusion between marginal opportunity costs and marginal costs, the unit marginal costs of production are referred to from this point forwards as running costs (RC). In the long run the MOC will be derived from the running costs and capital costs (CC) of displaced alternatives. This accommodates the likelihood that the system may adjust its cost structure by making investments in power sources that have lower running costs. In this section, then, the algorithm for identifying the unit RC and CC relevant to particular time periods is derived.

While the model developed in this paper does not simulate optimising behavior directly, such behavior is implicitly incorporated into the assessment of the response of reservoir operation to changes in hydrological outputs. That analysis is based on an understanding of how the stacking patterns of hydrothermal electric power systems are optimised in order to minimise total system costs. These patterns subsequently determine reservoir operation under different states of the reservoir and the HTEPS; states produced by varying hydrological conditions. The objective is to find the unit RC or CC that corresponds to a change in power generation or power generating capacity. In order to accomplish this it is necessary to parse through the twelve combinations of reservoir and system states in a systematic fashion to ferret out the appropriate power source (or unmet demand) in a given situation. The marginal rate of substitution, running costs and capital costs associated with this source must then be estimated. Squaring with intuition, the method will potentially yield a different per unit MOC for a productivity change according to whether it is the wet or dry season, whether it is a wet or dry year and whether it is the long or short run.

In cases where no water is discharged and there is no productivity change, there is no relevant MOC. From **Table 7** above it can be seen that of the twelve cases only six actually result in a change in discharge and, hence, a change in productivity. These cases can be divided according to whether they occur in the short or long run. Treating the development of running costs and capital costs separately and then combining them in an appropriate fashion accommodates this difference. Running costs are treated first, followed by capital costs.

Running Costs The running costs of a change in reservoir productivity will depend on whether the system is incurring positive marginal costs in producing electricity during that period or whether unmet demand exists. The latter may include either a shortfall in meeting primary load demand or unmet secondary demand. The RC of a change in productivity will be the costs to the system of either the marginal power source or the unmet demand. The decision rule is to take as the RC for a given season, *s*, and year, *t*, the marginal costs of production of the power source (or unmet demand) with the highest cost to the system. In the case of power shortfalls, the cost is considered to represent the marginal costs of the shortfall, MC^{PS}, which reflect the penalty cost to the system of not meeting the primary load. The RC of unmet secondary demand will be the price of secondary power, P^{SP}. The decision then depends on a series of inequalities that serve to identify the marginal power source. The formulation below assumes that the marginal cost of alternative thermal power sources, MC^{ATP}, lies between power shortfall costs and the price of secondary power. The unit running costs for a given time period *st* in the short run are as follows:

$$RC_{st}^{SR} = \begin{cases} MC_{st}^{PS} \text{ when } G_{st}^{PS} > 0 \\ F_{st}^{P} \mu^{ATP,RC} MC_{st}^{ATP} \text{ when } G_{st}^{ATP} > 0 \text{ and } G_{st}^{PS} = 0 \\ \mu^{SP,RC} P_{st}^{SP} \text{ when } \overline{G}_{st}^{SP} - G_{st}^{SP} > 0 \text{ and } G_{st}^{PS} = G_{st}^{ATP} = 0 \end{cases} \qquad \text{Eq. 42}$$

The marginal rate of substitution, μ, will include adjustments required to equate power as it is delivered to the load center under both circumstances. In other words, μ reflects transmission costs and losses to load centers between the HEP plant and the alternative.[17] The substitution rate is not included in the MOC for power shortfalls as there is no substitution occurring. In the case of secondary power the substitution rate should reflect transmission costs and losses. The existence of unmet demand for secondary power is conditional upon the existence of a shortfall between secondary power generated, G^{SP}, in a given period *st* and the full period demand for secondary power, $\overline{G}^{SP}$. Further differentiation of this decision tree is, of course possible, particularly as regards the different types of thermal power that may be employed by a given system. The compounding factor, F^P is used to incorporate the expected growth rate of prices for the petroleum used as fuel in thermal power production.[18]

The corresponding long-run unit running costs are as follows:

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[17] This distinction breaks down the concept of the marginal rate of substitution into running and capital cost components, as mentioned by Barbour *et al*. (1985).

[18] The formulation is as follows:

$$F_{st}^{P} = (1+gr)^{(t-1)+l_s/365}$$

where *gr* is the growth rate in prices and l_s is the number of days from the beginning of period *t* through to the end of season *s*.

$$RC_{st}^{LR} = \begin{cases} \mu^{ATP,RC} \mathrm{MC}_{\mathrm{st}}^{\mathrm{ATP}} \text{ when } \Delta\overline{G}_{st}^{ATP} > 0 \\ MC_{st}^{HEP} F_{st}^{P} \text{ when } \overline{G}_{st}^{HEP} > 0 \text{ and } \Delta\overline{G}_{st}^{ATP} = 0 \\ \mu^{SP,RC} P_{st}^{SP} \text{ when } \overline{\mathrm{G}}_{st}^{SP} - G_{st}^{SP} > 0 \text{ and } \overline{G}_{st}^{HEP} = \overline{G}_{st}^{ATP} = 0 \end{cases} \quad \text{Eq. 43}$$

where the expected change in thermal power and HEP capacity under the long-run system expansion plan during period *st* are, respectively, $\Delta\overline{G}_{st}^{ATP}, \Delta\overline{G}_{st}^{HEP}$. Note, of course, that as the marginal costs of HEP are zero the middle line in the equation above will return running costs of zero.

Capital Costs A change in hydrological outputs in the long run will alter not only running costs (as in the short run) but also the investment and yearly operations and maintenance costs (O&M) associated with a power source that is added to system capacity. The role of secondary power is more problematic in the long run. System expansion may accommodate existing secondary demand in the long run, thus effectively eliminating the need to consider such demand. However, if the willingness-to-pay for secondary power remains very low into the future this is unlikely to occur, at least insofar as the price of secondary power is less than the costs of adding system capacity. In any case, the use of a change in productivity to generate secondary power remains a useful concept in the long run. In particular, for simulation purposes it provides a floor for valuing the change in productivity if there is uncertainty regarding the response of the expansion plan to the change in hydrology.

In a predominantly hydroelectric system the long-run MOC of a change in generating capacity during wet periods and seasons most likely will be related to the costs of hydropower generation. Thermal capacity during such periods is rarely utilised, much less at capacity. Instead, as wet period demand rises relative to supply, additional HEP plants are constructed to increase wet season capacity. In drier periods, the system would still be using thermal power, thus the relevant capital costs will be that of thermal power. In the long run it may be assumed that power shortfalls are eliminated, or effectively minimised, by system expansion.

The cost figures employed are intended to be annualised figures that are subsequently adjusted for the length of the seasons for which they are employed. The estimates then literally represent the cost of postponing or bringing forward the costs of the alternative power source over the number of days in the season concerned. The degree to which capital investments may be varied to suit interannual fluctuations in the change in productivity will depend on the system. Where plant investments occur only periodically, say once every five years, the valuation may need to adjust to this periodicity. For example, it may not be reasonable to take a benefit in one year and a cost in another in the case where the change in productivity is positive in one year and negative in the next. In addition, given marginal changes in productivity there is an implicit assumption that rather than postponing an investment it is possible to decrease or increase the scale of planned investments. In other words, a HEP project planned for year t may be scaled back in order to reflect increased HEP availability from the reservoir. Again, the usefulness and applicability of such assumptions will depend on the situation.

The calculation of long-run opportunity costs is therefore a complicated matter. In theory it requires knowledge of which plant would have been built or which plant would have been scaled up or down had the change in productivity not occurred. In addition there is the practical difficulty of manipulating the three cost figures, each of which are expressed in different units (capital costs in \$/kW, O&M in \$/kW/yr and running costs in \$/kWh). Although the calculation of a change in annual generation (in kWh) may be relatively easy to obtain it may be difficult to assess just how this translates into a reduction in fixed capacity (in kW). If such figures are available, one method of simplifying this problem is to simply use the long-run marginal costs of power expansion as provided by the system (in \$/kWh). Such figures may apply to the system as a whole or to particular types of energy production (i.e. thermal power, run-of-stream, etc).

A second method is to actually assess the KWs of expansion that can be postponed in a given year and then use the annualised capital cost component (*AFC* in \$/kW/yr) and corresponding

O&M costs to calculate the capital costs over the period *st*. The simplest method of arriving at the hydroelectric capacity equivalent of a change in productivity (in kWh) is to assume that the generation is distributed evenly across the period. Thus, for a season *s* of length in days of *d*, the change in generation capacity would be:

$$\Delta\overline{G}_{st}^{LHR} = \frac{\Delta G_{st}}{24 d_s} \qquad \text{Eq. 44}$$

The marginal opportunity cost of a per unit change in long-run power generation capacity would then depend on the following inequalities:

$$CC_{st}^{\Delta\overline{G}} = \begin{cases} \mu^{ATP,CC}\left(d_s/365\right)\left(AFC_{st}^{ATP} + O\&M_{st}^{ATP}\right) \text{ when } \Delta\overline{G}_{st}^{ATP} > 0 \\ d_s/365\left(AFC_{st}^{HEP} + O\&M_{st}^{ATP}\right) \text{ when } \overline{G}_{st}^{HEP} > 0 \text{ and } \Delta\overline{G}_{st}^{ATP} = 0 \\ 0 \text{ when } \overline{G}_{st}^{SP} - G_{st}^{SP} > 0 \text{ and } \overline{G}_{st}^{HEP} = \overline{G}_{st}^{ATP} = 0 \end{cases} \qquad \text{Eq.45}$$

In the case of alternative thermal power plants, the marginal rate of substitution will account for the lower mechanical reliability and flexibility of th*e ATP sources, as well as the hydrological availability of the HEP plant (Barbour *et al.* 1985). In the case of substitution by other HEP plants, the marginal rate of substitution is assumed to be one. As there are no capital costs associated with the supply of secondary power in this case, the capital costs are set to zero.

<u>Marginal Opportunity Costs</u>

In order to obtain the per period marginal opportunity cost, the per unit running costs and capital costs must be multiplied by the respective change in either production or generating capacity. As the relevant alternative power source for calculating running costs may vary from the short run to the long run, this must also be accounted for. The present value of the marginal opportunity costs of the productivity change over the planning horizon of *N* years, for each wet and dry season, *s*, employing a discount rate, *r*, is:

$$MOC = \sum_{t=1}^{N} \sum_{s=dry,wet} \frac{\begin{Bmatrix} RC_{st}^{SR}\Delta G_{st} \text{ when } t < \text{LR} \\ RC_{st}^{LR}\Delta G_{st} + CC_{st}\ \Delta\overline{\text{G}}_{\text{st}} \text{ when } t \geq \text{LR} \end{Bmatrix}}{\left(1+r\right)^{(t-1)+l_s/365}} \qquad \text{Eq. 46}$$

The MOC flows are assumed to accrue at the end of each period season, so that the discount factor must be modified to be proportional to period *st*. This is accomplished using the number of elapsed days from the beginning of period *t* through to the end of period *s*, as represented by l_s. This, as all money flows are assumed to occur at the end of seasons.

In sum, in order to establish a short-run measure that represents the welfare change occurring due to the downstream hydrological externalities, it is necessary to understand whether (and how) the change results in a change in the utilisation of other components of the system. If these components have a positive marginal cost, then the change in productivity will affect total system operating costs. In the long run, the welfare measure will include not just marginal costs but the fixed cost of capital improvements and O&M as they are affected by the productivity change. The diagnosis of the cost savings will, at its most complicated level, depend on an intertemporal assessment accounting for different hydrological years, the seasonality of run-off and storage, the system's structure, and the peculiarities of the reservoir, plant and watershed at the site.

It is important at this point to reflect on the theoretical limitations of employing these marginal opportunity costs as benefit measures. The benefits discussed above refer to the savings generated by avoiding the necessity of spending money on alternative generating sources. While an improvement over previous efforts, the methodology remains imperfect for a number of reasons. First, the method does not assess how changes in productivity might play through into changes in the demand for electricity or input markets for power production, with corresponding impacts on the costs and prices of the substitutes employed. If the change in hydrology is marginal, these concerns may be of limited practical impact.

Perhaps a more salient point is that the marginal opportunity costs employed under this method do not include any other non-priced externalities associated with the alternatives power sources. The cost of alternative power sources (fixed and marginal) are a reflection of competitively determined market prices for these alternatives, not their economic prices. In other words, if emissions from power plants are causing economic damage due to air pollution or climate change, then the economic value of avoiding these sources will be higher than the market price. Typically, it is implicitly assumed that hydroelectric power generation is environmentally benign, however, it may be important to consider the difficult-to-price but substantial social and environmental costs of dislocation and habitat loss associated with the construction of hydrostorage facilities. Estimation of the monetary impact of these factors ranges from the very site-specific (air pollution and social impacts of land use conversion to hydrostorage) to the unimaginably broad (climate change). Typically, the direction of such externalities would be to raise the marginal cost of thermal power and the capital cost of HEP, underscoring the conservative nature of these market-derived marginal opportunity costs as value estimates.

Data and Results

The data and assumptions underlying the application of the simulation model to Río Chiquito are presented in order starting with the hydrological analysis and continuing on with reservoir operation, production function and marginal opportunity costs. The results of the base case scenario are then reported, followed by the sensitivity analysis.

Hydrological Analysis

The objective of the hydrological analysis is to obtain the difference between expected suspended sediment and water yield under pasture and forest and to analyse the location of sediment deposition within the lake in order to arrive at the change in expected sedimentation of the dead and live storage volumes. All of this is to be accomplished with respect to the different combinations of land units in the study area. In the two succeeding sub-sections the analysis employed in assessing the changes in sedimentation and water yield are presented, as well as a brief discussion of how the seasonal timing of water yield is incorporated into the sensitivity analysis.

Sedimentation The response of sedimentation to land use change in Río Chiquito is simulated using the CALSITE software model developed by HR (Hydraulics Research) Wallingford (Bradbury 1995). The analysis builds on an application of the CALSITE Model to the three large micro-watersheds in the upper Arenal watershed (Saborío and Aylward 1997). The CALSITE program calculates simulated erosion using the USLE and then applies a sediment transport model to generate estimates of delivered sediments to the mouth of the watershed.

The CALSITE program has four stages. In the first stage the estimation of average annual soil erosion in tons per hectare for each cell *i* is conducted using the base maps for the USLE factors:

$$A_i = R_i K_i LS_i C_i P_i \qquad \text{Eq. 47}$$

The explanatory variables are as follows:

- *R* is erosivity of rainfall, determined by the maximum intensity over a 30-minute period and the kinetic energy of rainfall.
- *K* is an index of soil erodibility, calculated based on soil characteristics such as structure, permeability and organic material.
- *LS* is an index of length of slope and inclination of slope.
- *C* is crop practice factor or type of land use.
- *P* is type of conservation practice.

With erosion in hand the CALSITE model requires only the following three steps to arrive at suspended sediment yield:

1. Determination of a distribution index, DI_i, employing an empirical equation that relates the delivery of sediment to rainfall, surface flows and topographical features.
2. Calibration of simulated delivery with observed estimates.
3. Final estimation of suspended sediment (in tons/ha/yr) delivered from each cell in the watershed based on the product of A_i and $f(DI_i)$ where $f(\bullet)$ is the delivery ratio for the cell as calculated using the distribution index.

In keeping with the general model presented above, suspended sediment yield is thus defined as:

$$SSY_i = \alpha_i A_i f(DI_i) \qquad \text{Eq. 48}$$

where α is the weight to volume conversion factor. As bedload is assumed not to change with land use, the change in live and dead storage sediment, respectively for a given land unit, *i*, will be:

$$\Delta S_i^L = \beta\kappa\left(\alpha^P A_i^P f(DI_i) - \alpha^F A_i^F f(DI_i)\right) \qquad \text{Eq. 49}$$

$$\Delta S_i^D = \beta(1-\kappa)\left(\alpha^P A_i^P f(DI_i) - \alpha^F A_i^F f(DI_i)\right) \qquad \text{Eq. 50}$$

CALSITE is integrated with IDRISI, a geographic information system software package using raster technology. Consequently, CALSITE enables the user to effectively determine the suspended sediments generated by each cell in the watershed based on the USLE input maps and a digital elevation model for the watershed. In the base run of the CALSITE program the cell or pixel size employed is fifty by fifty meters and the base land use map shows 56.4% of Río Chiquito in pasture (Saborío and Aylward 1997). Total erosion generated on pasture in the watershed is close to 200,000 tons/ha/yr. The average sediment delivery ratio for pasture is 47% and the corresponding total for suspended sediment delivery is just over 90,000. Pasture is expected to produce 99% of the sediment delivered in the watershed under current land use.

In order to simulate the effect of different land uses on sedimentation within the CALSITE model the program may be run again using different land use input maps (Bradbury 1995). Thus, in order to simulate the effect of reforesting existing areas of pasture in Río Chiquito, a second map is generated that simulates sediment delivery under 100% forest cover.

In order to convert the estimates of suspended sediments produced by the maps (in tons/ha/yr) into sediment delivered to the live and dead storage areas of the reservoirs (in m^3/yr), the following four transformations are applied to both maps in sequence:

1. The weight of the sediment is converted to volume using the density of the sediments (α), 0.99 tons/m^3 for sediment arriving from pasture and 0.86 tons/m^3 for sediment arriving from forest (Aylward *et al*. 1998: 284).

2. Given the size of the lake and the distance to the outtake, it is fairly clear that all of the suspended sediment generated in Río Chiquito will precipitate out as sediment and justifies the use of a trap efficiency (β) of 100% (Mahmood 1987).

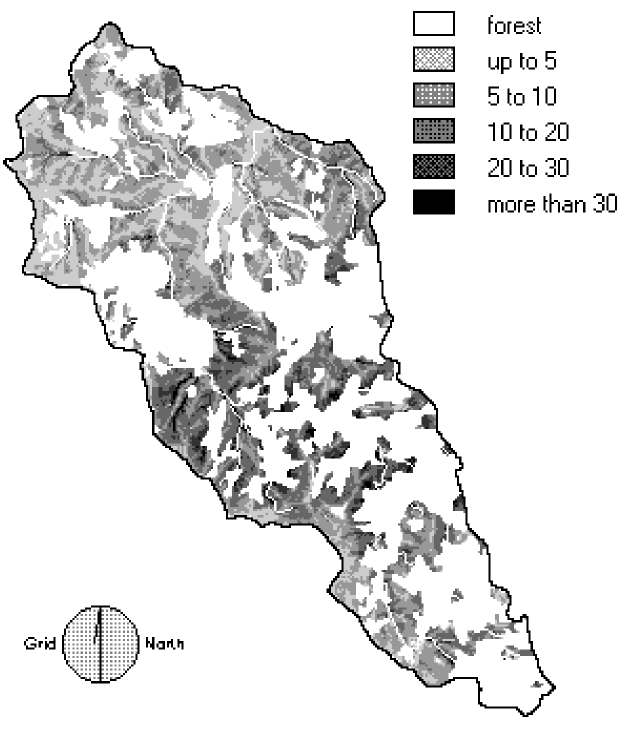
3. A bedload (*B*) of 9,000 m^3 of the suspended sediment load is assumed under both uses to arrive at total sedimentation. However, this figure is not included in the subsequent calculations (or maps) as the load does not have an identifiable source and is assumed not to be affected by land use.

4. Based on previous studies of sediment deposition in Lake Arenal, the sediment delivered is assumed to be distributed half to the live and half to the dead storage volumes (κ=0.5) (Aylward *et al*. 1998: 81).

At this point four maps are obtained for the original pasture areas in Río Chiquito: maps showing the amounts of sediments sent to the live and dead storage by each cell under the existing land use and the two corresponding maps under full forest cover. In order to arrive at the "extra" sedimentation produced by pasture, in comparison with forest, the maps (or more accurately the sediment values attached to each cell) representing live and dead storage of sedimentation under full forest cover are subtracted from the corresponding maps for pasture. The end result, then, is two maps of the additional sedimentation deposited under pasture, one map each for the live and dead storage areas.

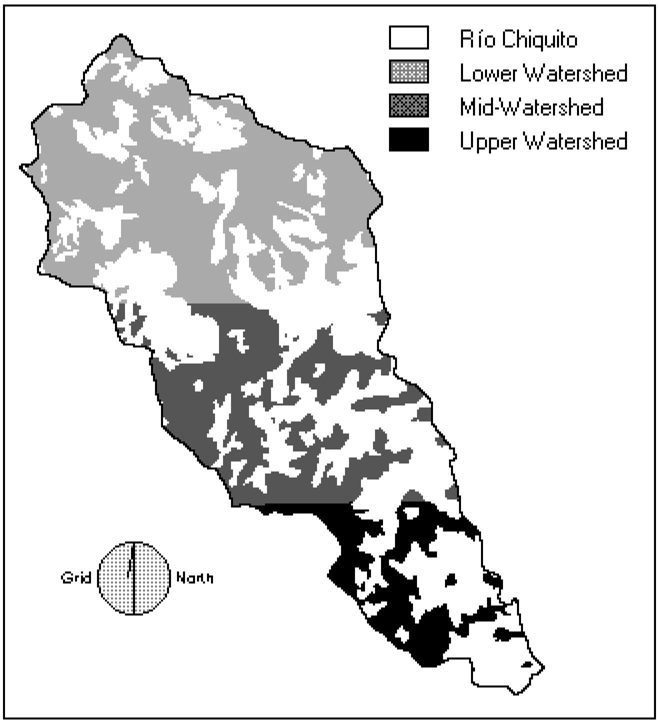
In **Figure 5** the final map for net sedimentation of the live storage is presented showing only those areas currently under pasture. As the proportion of sedimentation allocated to the dead storage is also 50%, the map for sedimentation of the dead storage is the same as **Figure 5** and, therefore, is not shown. The following discussion, thus, applies in equal terms to the sedimentation of the dead storage.

Figure 5. Map showing source of changes in sediment from Río Chiquito pasture that is deposited in the live storage of Lake Arenal



The annual sedimentation that is added to live storage due to the presence of pasture in place of forest is approximately 45,000 m^3/year. However, as shown in **Figure 5**, there are important variations in the per hectare rate of sedimentation across the watershed. In order to examine this variation the watershed is divided into lower, middle and upper sections as shown in **Figure 6**. The pastures located in the middle of the watershed clearly have a higher total sedimentation rate than those located in the lower watershed and the upper watershed. In the latter two sections of the watershed, practically all pasture areas deliver less than 20 m^3/ha/yr to live storage, whereas in the middle watershed some 18% of the area produces greater than 20 m^3/ha/yr of live storage sediment. Average rates of sedimentation of the live storage for these three regions of the watershed are 6.5 m^3/ha/yr in the lower watershed, 14 m^3/ha/yr in the middle sections of the watershed and 8 m^3/ha/yr in the upper watershed.[19] Similar figures, of course, apply to the change in sedimentation of the dead storage, and figures for total sedimentation are twice again as much.

Figure 6. Division of Pasture Areas in Río Chiquito Watershed into Three Sections



Water Yield In order to establish the change in water yield that can be expected under the two land uses, a variation of the water balance approach is used. Over the course of a year each land unit, *i*, has its own amounts for precipitation (vertical), P^V, evapotranspiration, *ET*, and change in soil water storage, ΔST. The precipitation that is not consumed by evapotranspiration or by adding to soil moisture is assumed to add to run-off:

$$R_i = P_i^V - ET_i - \Delta ST_i \qquad \text{Eq. 51}$$

Evapotranspiration can, in turn, be specified as the sum of precipitation that is intercepted and evaporated, *I*, and soil water that is transpired by vegetation, *T*, minus the input of horizontal precipitation, P^H, that is stripped from fog and clouds by vegetation:

$$ET_i = I_i + T_i - P_i^H \qquad \text{Eq. 52}$$

In many cloud forest studies only above ground measurements are taken, as water reaching the surface is collected in cloud forest and adjacent areas that have no forest cover. Leaving aside transpiration, then, these studies calculate net interception, which is simply interception minus horizontal precipitation, or the water collected in the open minus the water collected in

[19] Calculations were performed using the IDRISI™ geographical information systems software package.

the forest. In a non-cloud forest area net interception is positive as there is no horizontal precipitation. In a cloud forest, however, the amount of horizontal precipitation captured may make up for water intercepted and evaporated by vegetation and, thus, net interception will be less than in non-cloud forest. On occasion net interception may even be negative in a cloud forest, signifying that the forest is "producing" water (at the surface) that would not be produced in the absence of forest cover.

As the modeling exercise in this paper involves the use of average annual figures over an extended time horizon, it is assumed for simplicity that the annual change in soil water storage in Equation (50) does not change. In addition, the literature suggests that vertical precipitation will not change between one land use and another (at this micro-scale). However, the expectation emerging from both theoretical and empirical work in the literature is that the rate of evapotranspiration is higher in forest than in pasture. As a result, the problem of estimating the gain in water yield produced under pasture, ΔR, reduces to deriving the difference in evapotranspiration between pasture, ET^P and forest, ET^F. In, other words a comparison of Equation (50) under the two land uses reduces to the following equation:

$$\Delta R = ET_i^F - ET_i^P \qquad \text{Eq. 53}$$

If estimates of evapotranspiration are not available the above equation can be subsequently broken down into a determination of the changes in transpiration, interception and horizontal precipitation. Given the presence of cloud forest in the watershed the case study distinguishes between the change in run-off in cloud forest areas and non-cloud forest areas. The latter case is taken first.

Non-Cloud Forest Areas

In the case of non-cloud forest areas, a study by Calvo (1996) reported on the difficulty in determining linkages between land use and run-off at the watershed level in Arenal. The primary difficulty in estimating such relationships is that the data series on run-off (from 1975) does not overlap with the period during which most of the land use change occurred (1950s to 1980). As a result more general methods are required in order to assess the difference in evapotranspiration in these areas. This is accomplished by employing figures on the change in evapotranspiration between forest and pasture developed using the life zones methodology and by comparing these estimates with those available in the literature.

As part of a study of water issues in Costa Rica, the TSC calculated the run-off that could be expected under pasture and forest using the life zones methodology (CCT and CINPE 1995). In Río Chiquito four life zones are present (see **Figure 7**):

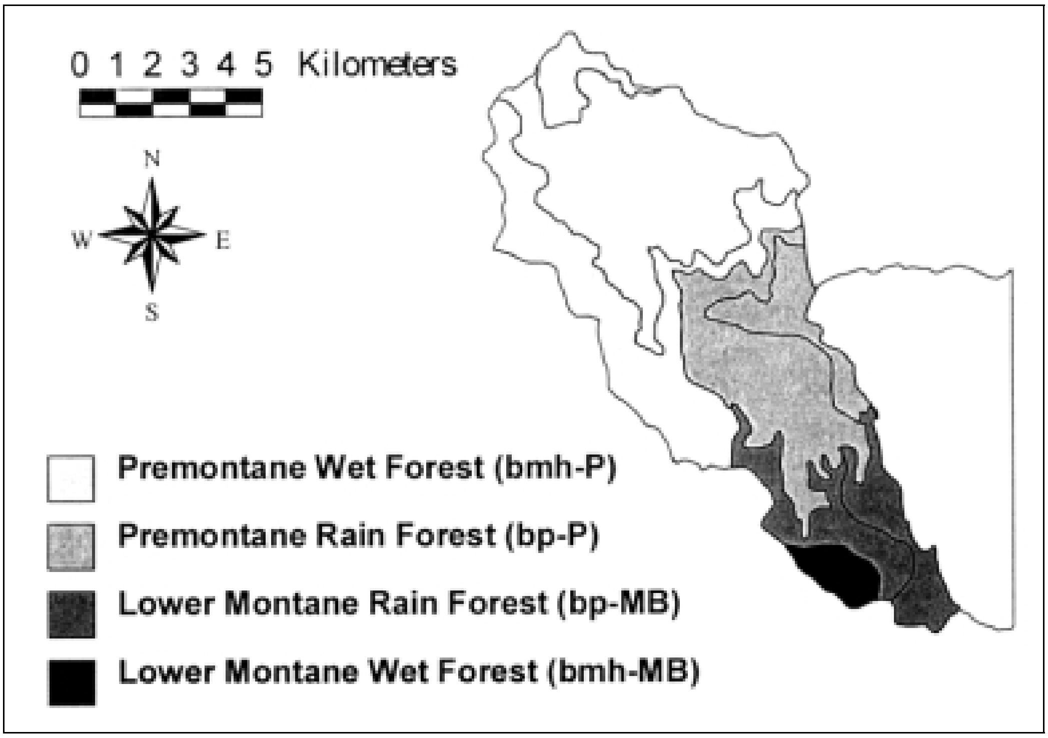
1. The lower watershed is Wet Premontane Forest (bmh-P).

2. The middle section of the watershed on the east side and to the interior are Premontane Rainforest (bp-P).

3. The upper watershed is predominantly Lower Montane Rainforest (bp-MB).

4. A small section of the upper watershed that borders the continental divide (to the west) is Wet Lower Montane Forest (bmh-MB).

Figure 7. Map of Life Zones in Río Chiquito



The figures cited in the TSC study suggest a gain of from 187 to 285 mm/yr in these life zones (see table below). In order to arrive at the extra production of run-off under pasture, that is in cubic meters of water per year, these figures need only be converted to meters per year and then multiplied by ten thousand square meters.

Table 8. Run-off under forest and pasture by life zone in Río Chiquito

Run-off (mm/yr)	bmh-P	bp-P	bmh-MB	bp-MB
Pasture	2,146	5,230	1,890	4,436
Forest	1,861	4,972	1,687	4,249
Gain from pasture	285	258	203	187
Gain in m^3/ha/yr	2,420	2,580	2,030	1,870

Source: CCT and CINPE (1995), Anexo 1. Notes: Figures are for climatic associations.

These results can be compared with estimates from the literature. In a statistical analysis of 94 studies in temperate and tropical watersheds, Bosch and Hewlett suggest that a 10% change in cover in a watershed populated by broadleaf forest would result in an increase of 25 mm/yr in run-off (Bosch and Hewlett 1982). Extrapolating the results to a full conversion for a given land use unit, i.e. a 100% change from pasture to forest, leads to an estimate of 250 mm/yr. Bruijnzeel (1990) reviews the results of paired catchment studies and empirical water balance studies conducted in the humid tropics and concludes that the removal of forest cover may result in a considerable initial increase in run-off (up to 800 mm/yr or more in areas of high precipitation). In addition, in the case of permanent conversion to pasture Bruijnzeel (1990) suggests that existing studies confirm that run-off levels will remain above that attained under forest cover. Studies in French Guyana, for example, observed changes of 270 to 325 mm/yr in lowland areas.

Consultation of the available literature suggests that the gain in run-off under pasture found by the TSC (187 to 285 mm/yr) is well within the bounds that have been established, and, considering local conditions may even be regarded as conservative. These figures, therefore, are employed to represent the gain in run-off derived from pasture (in place of forest) in areas that are not considered cloud forests.

Cloud Forests

In the case of cloud forest areas, a year-long interception study by Fallas (1996) was undertaken in 1995-1996 in the upper Río Chiquito watershed. Fallas found a gain in net interception of 194 mm/yr in the low primary forest and a loss of 318 mm/year in the high primary forest. If the difference in transpiration is assumed to be minimal, these results make it difficult to sustain the hypothesis that evapotranspiration in large tracts of primary cloud forest will be significantly different than that in pasture.

However, the results obtained by Fallas (1996) for forest fragments suggest that primary forest fragments that are intermingled with pasture will produce a water yield gain of 460 mm/yr over pasture. The yield gain then is associated with the opening up of the cloud forest and the exposure of an increased amount of surface area to fog moisture. For the purposes of water yield, then, neither end of the spectrum of land uses (100% pasture or 100% forest) is desirable. In a sense then the (incomplete) creation of pasture amongst cloud forest increases the ability of the forest to produce water. At some point, however, the continued expansion of pasture leads to a reversal of the effect as forest fragments that are prolific producers of water are cut.

The results provided by Fallas do not provide the detail required in order to estimate the form of this spatial relationship between fragment size and net interception. Nor indeed is the available information on land use in the watershed of such a resolution to enable a true estimate of fragmentation. Nonetheless, the sheer magnitude of the effect suggests that it is worth exploring its potential economic impact in a simplistic manner, particularly as a familiarity with the watershed and revision of aerial photographs suggests that the degree of fragmentation varies substantially across the watershed. Colonised first, the lower watershed consists of large open pastures with remaining forest concentrated on hilltops, near roads and in riparian areas. Progressively, the mid to upper watershed has less of a continuous pasture cover in cleared areas. For example, in the upper watershed many individual trees remain standing in pasture and there are scattered clumps of forest fragments dotting the landscape to the extent that the landscape appears to be an equal mix of forest and pasture.

The cloud forest areas within the watershed that are colonised lie along the continental divide in both the lower and upper watershed. Fallas (1996) studied the latter area. As Bolaños (1995, pers. com.) suggests that the lower watershed may receive even more horizontal precipitation than the upper watershed, the results from the study by Fallas represent a conservative estimate for these areas.

The only quantifiable indicator of the degree of fragmentation in these areas is the extent of forest as a percentage of pasture on livestock holdings. Average forest to pasture ratios for different livestock production types by elevation are as follows: 11%, 14% and 23% in the lower watershed; and 60% and 63% in the middle to upper watershed (Aylward *et al*. 1998: 86). These figures do not indicate the degree of fragmentation per se, as all of the forest on a given holding could be in a large contiguous block. As suggested above the degree of actual fragmentation in the middle to upper watershed, however, is quite high, falling somewhat in the lower watershed. A rough estimate of the proportion of fragmented forest to pasture, therefore, is 15% in the lower watershed and 60% in the middle to upper watershed.

Based on the research by Fallas and these estimates of fragmentation, the change in net interception expected from a hectare of pasture in cloud forest areas can be derived as follows. Consider a holding of 30 hectares that has two-thirds of the area (20 hectares) converted into pasture. There would be no net change in water yield for the pasture itself (as compared to if it were under forest cover). The forest fragment areas would produce 460 mm/yr more water than they would have in the absence of colonisation and fragmentation. The total water gain due to the off-site effect of clearing pasture would be 46,000 m^3/yr (10 hectares multiplied by 4,600 m^3/ha/yr). The gain in run-off per hectare of pasture would be 2,300 m^3/ha/yr (46,000 m^3/yr divided by 20 hectares). This is equivalent to simply multiplying the per hectare figure for water yield gain in fragmented forest (4,600 m^3/ha/yr) by the ratio of fragmented forest to pasture (10:20 or 50%).

In other words it can be expected that a hectare of pasture in the lower watershed is responsible for a gain of 690 mm^3/yr on average (15% of 4,600 m^3/ha/yr), while pasture in the mid to upper watershed produces 2,760 mm^3/ha/yr (60% of 4,600 m^3/ha/yr). Again, assuming the difference in transpiration to be minimal these figures are used in the simulations to represent the increase in water yield under pasture. Contrary to expectation, then, the conversion of cloud forest to pasture may not lead to a decrease in water yield, but instead an increase. The caveat, of course, is that the conversion be partial in extent and spatially distributed throughout the area. Although there is no previous precedent for this type of analysis or result, it has an intuitive appeal if the partial nature of deforestation is considered in a spatial context. Certainly, there is a conceptual linkage, however implicit, between this analysis and that used to justify the establishment of mechanical collection devices in areas of high cloud moisture content in the mountainous, desert areas of Peru, Ecuador and Chile (Schemenauer 1994 and Schemenauer and Cereceda 1994).

In sum the changes in water yield are based on the overlay of three factors: (1) life zones, (2) the range of cloud forests and (3) the spatial distribution of pasture and forest in cloud forest areas. For the sake of simplicity, it is assumed that net water yield is additional to the estimated change in total sediment delivered. In other words the change in run-off is assumed to be "clean," i.e. not to contain the sediment that is generated by moving from forest to pasture.

Seasonality of Water Yield Given the need to resort to fairly simplistic formulations of expected changes in water yield between forest and pasture it is not surprising that there is no quantitative information on which to sustain a hypothesis regarding the seasonal pattern of any changes in annual run-off. The baseline scenario in the simulation simply assumes that the gain is apportioned in proportion to historical averages of run-off from Río Chiquito. The water yield figures are divided into wet (January through May) and dry (June through December) season amounts based on monthly averages derived by Calvo (1996) for Río Chiquito run-off during the period 1975-1993. Run-off during the dry season is roughly 25% of total annual run-off. Thus, in the base scenario 25% of the gain in water yield is assumed to occur during this period and 75% in the rainy season. The same figures are used in apportioning sediment changes across the seasons.

Given that much of the horizontal precipitation in cloud forest is captured during the dry season when cloud moisture and high winds prevail, the proportional division of changes in water yield represents a conservative estimate for these areas. In pasture outside of cloud forest areas, other things being equal, it might be expected that infiltration rates are lower than they would be under forest cover. As a result the use of a proportional gain in water yield for the increase in dry season run-off might overstate the difference to be expected in comparison with forest. That is, it might be expected that with lower infiltration rates, proportionately more of the water yield gain under pasture would accrue during the wet season. This as the proportions are derived from data which reflect the watersheds current land use pattern and, therefore, incorporate the impact of lower infiltration rates.

Countering this argument is the fact that the watershed has a relatively unpronounced dry season. This suggests that during the dry season the higher evapotranspiration rate observed under forest cover will indeed lead to a decrease in water yield (as opposed to pasture) as rainfall during this period will be intercepted and evaporated away. For example, the meteorological station at the town of Río Chiquito, located in the middle of the watershed, shows that 22% (or 668 mm) of total precipitation falls during the dry season (CCT 1980). Given that dry and windy conditions can be expected to lead to higher interception and transpiration rates from forest (in place of pasture) during this period, it is unlikely that run-off during this period would actually fall under pasture unless there was a very pronounced decrease in groundwater storage at the end of the rainy season. However, given the generally high levels of rainfall in the watershed, the change in infiltration would have to be quite large to leave a "deficit" at the end of the rainy period under pasture (as opposed to forest). These factors suggest that this overstatement will be relatively limited in the case of Río Chiquito. Nevertheless, the simulation of seasonality of water yield considers the case where all of the increase in water yield observed under pasture occurs during the wet season. Further simulations are conducted to find the threshold at which the transfer of net water yield from the dry to the wet season would eliminate the expected positive effect of an increase in overall annual water yield.

Some limitations of the hydrological analysis as undertaken are worth noting at the outset. The substitution of pasture for forest, or forest for pasture is not instantaneous. In reality, given that pasture is already in existence, the true externality is that caused by not converting the land back to forest cover as of time zero. In theory it is possible to simulate the effect of changes in hydrology over time as the forest re-establishes itself. However, the effects of regeneration (or conversion) on erosion and evapotranspiration can be expected to occur rather rapidly in relationship to the time frame (70 years) of the analysis. Therefore, this complication is avoided by simply conducting a one-off comparison between pasture and forest and employing these results to compile the economic impacts over the full length of the time horizon.

A second limitation is that the methods for deriving the changes in water yield and sedimentation are not explicitly linked. The latter limitation is in part due to the manner in which the change in water yield is derived. No formal modeling process per se is followed, rather the change in water yield is based on the relevant literature and experimental evidence from sites in Río Chiquito. This also contributes to the difficulty of developing a model that can properly simulate the effects of variable rainfall levels on water yield and sedimentation changes over time. In other words, the changes in these hydrological outputs are assumed constant over time. Again, however, the need to properly simulate interannual variation is judged secondary to the importance of integrating both effects into a long-run economic analysis and subsequently simulating the effects of changing key variables.

Reservoir Operation

In this sub-section the effect of the aforementioned changes in hydrological outputs from Río Chiquito are explored in terms of the manner in which they affect reservoir operation over time. The presentation begins by evaluating the relevance of the operating constraints to the Arenal case. The change in hydrological outputs and the live storage capacity constraints are then presented based on the hydrological analysis. Background on the operation of the Costa Rican grid (or National Interconnected System, SNI) is covered before turning to the specification of the hydrological conditions and states of the reservoir and the SNI under these conditions.

Operating Constraints The dam holding back the Arenal River at Sangregado was finished in 1979. The dam greatly enlarged what had previously been a small natural lake at the headwaters of the Arenal River. The new Lake Arenal has a total storage of 2,416 million cubic meters (Hm^3) at its maximum height of 546 meters (Hydroconsult A.B. 1993). Between the maximum height and the minimum operational level of 520 meters the live storage volume is approximately 2,000 Hm^3 (Hydroconsult A.B. 1993). This leaves over 400,000 Hm^3 of potential dead storage. The land area within the watershed totals just over 41,000 hectares. In order to assess the possibility that the dead storage will fill over the planning horizon, a rough calculation of the sediment delivery required over one hundred years to fill the dead storage volumes is conducted. Even if 100% of the sediment delivered to the lake from all of its tributaries ended up in the dead storage, the annual per hectare sediment delivery rate would have to reach 68 m^3 for the dead storage to fill. Such an occurrence is extremely unlikely, if not impossible.[20] Thus, there is no need to check the operating constraints of the reservoir to see if the dead, or live, storage capacity will fill under either land use scenario. Dredging to remove sediment will not occur and the analysis may focus on evaluating the change in discharge and productivity expected due to the change in hydrological conditions.

Changes in Hydrological Outputs and Live Storage Constraint The next task in evaluating reservoir operation is to read into the algorithm the changes in hydrological inputs as they affect the reservoir and update the live storage capacity constraint. Based on the change in hydrological outputs from the previous section, the change in inflow and change in sediment arriving in the live storage can be calculated for each of the two seasons and each geographic land unit. The land units represent the valid intersections of the three spatial overlays represented by the location in the watershed, the type of forest, and the life zone. Location determines the sedimentation rate and the type of forest and life zone determine the change in water yield. As these parameters of the model do not change on a yearly basis it is not necessary to calculate them for every year. The figures for the valid combinations of these overlays are presented in **Table 9**.

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[20] The length of life of the dead storage is calculated to be anywhere from 168 to 1000 years by other authors, including respectively, Duisberg (1980) and Hydroconsult A.B. (1993).

Table 9. Change in hydrological outputs under pasture (in m^3/ha/yr)

Forest type	Non-Cloud Forest Areas				Cloud Forest		
Location	Lower	Middle	Middle	Middle	Lower	Middle	Upper
Life Zone	bmh-P	bmh-P	bp-P	bp-MB	bmh-P	bmh-P	bmh-MB
Sedimentation	13	28	28	28	13	28	16
Live Storage (ΔS_t^L)	6.5	14	14	14	6.5	14	8
Dry Season (ΔS_{dry}^L)	1.625	3.5	3.5	3.5	1.625	3.5	2
Wet Season (ΔS_{Wet}^L)	4.875	10.5	10.5	10.5	4.875	10.5	6
Dead Storage (ΔS_t^D)	6.5	14	14	14	6.5	14	8
Dry Season (ΔS_{Dry}^D)	1.625	3.5	3.5	3.5	1.625	3.5	2
Wet Season (ΔS_{Wet}^D)	4.875	10.5	10.5	10.5	4.875	10.5	6
Water Yield Gain							
Total (ΔR_t)	2,850	2,850	2,580	1,870	690	2,760	2,760
Dry Season (ΔR_{Dry})	605	605	645	467.5	172.5	690	690
Wet Season (ΔR_{Wet})	1,815	1,815	1,935	1,402.5	517.5	2,070	2,070
Change in Inflow							
Total (ΔI_t)	2,844	2,836	2,566	1,856	684	2,746	2,752
Dry Season (ΔI_{dry})	603	602	642	464	171	687	688
Wet Season (ΔI_{wet})	1,810	1,805	1,925	1,392	513	2,060	2,064

The next two inputs required by the valuation algorithm do change over time. First is the change in live storage carried forward from the previous time period, *st*. This input is simply read into the algorithm with the previous period's value. The initial value is assumed to be 0 ($\Delta V_{t=0,s=wet}^L = 0$). The values for the cumulative change in maximum live storage capacity is calculated as well based on Equation 39.

Operation of the SNI The SNI is predominantly hydroelectric, using a mixture of run-of-stream plants, water storage capacity (daily storage and Arenal) and geothermal plants to meet normal base and peakload demand.[21] Thermal power generation is used to supplement hydroelectric power during periods of peakload demand and low water storage, particularly during the dry season. As expected, the system rarely operates at 100% of capacity. As shown in **Table 10** the generating capacity of the system was 1122 MW in 1995, with 73% of the installed capacity being hydroelectric. The share of HEP production as a share of total production in 1994 roughly approximates this percentage at 76% (although due to the difference in years the numbers are not strictly comparable). Petroleum based thermal generation represents 22% of the system's generating capacity while new investments in geothermal provide 5% of capacity. All matters regarding electricity in Costa Rica are handled by the parastatal ICE. The private sector was recently given permission to produce a restricted amount of electric power. Nevertheless, ICE remains in charge of the SNI and continues to build and operate the bulk of the electric power generating capacity in the country.

[21] This section draws on material from ICE (1994) and ICE (1996).

Table 10. Electricity generation in Costa Rica

Type Plant	Generation (1994) (GWh)	(% of total)	Generating Capacity (1995) (MW)	(% of total)
Hydroelectric – Subtotal	3461	76.27%	819	73%
ARCOSA - Subtotal	1394	30.72%	362	32%
Cachí	613	13.51%	100	9%
Garita	169	3.72%	30	3%
Menores	221	4.87%	72	6%
Río Macho	585	12.89%	120	11%
Ventanas Garita	479	10.56%	96	9%
Other	Na	-	39	3%
Thermal Power – Subtotal	1077	23.73%	299	27%
Diesel and/or Bunker	688	15.16%	242	22%
Geothermal	389	8.57%	57	5%
Total	4538	100.00%	1122	100.00%

Source: ICE.

The Arenal hydroelectric complex is made up of the Arenal, Corobicí and Sandillal plants (ARCOSA). These three plants function in series using water drawn from Lake Arenal. Sandillal, the last plant in the series, has its own security reservoir as well. ARCOSA represents the most important component of the SNI with 32% of the installed electric capacity and 44% of hydroelectric capacity. In 1994, it produced 1,394 GWh of electricity or 31% of total production.

Power generation data for the period 1984 to 1993 reveals that up until 1991, hydroelectricity supplied practically all of Costa Rica's electrical energy. In 1992, thermal power generated about 15% of total electricity and in 1993, it generated approximately 10%. As summarised above, in the dry year of 1994 thermal power supplied 27% of electricity with 22% coming from petroleum based plants. Political discussions in the 1995 to 1996 period centered around rumors that ICE would be privatised, the failure of ICE to proceed with its expansion plan due to internal and financing problems and the potential for increasing power outages in coming years.

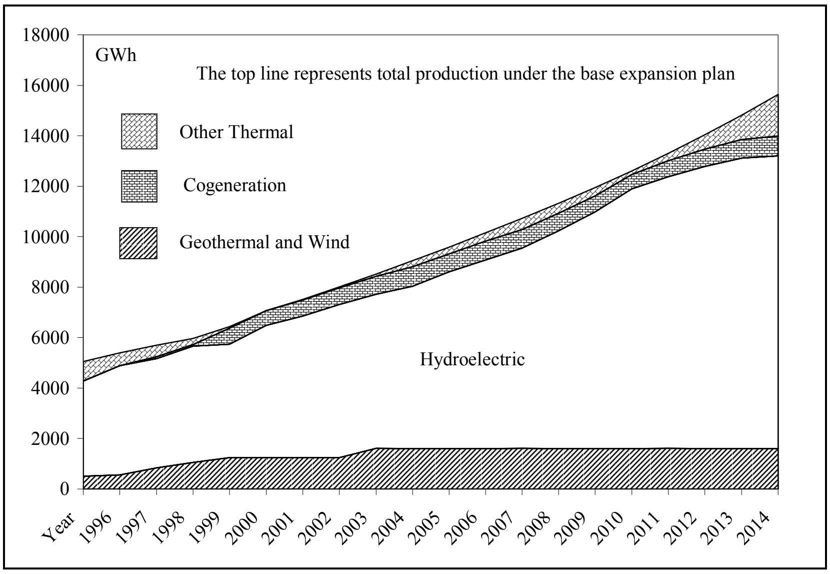
These supply issues arise in the context of healthy demand by Costa Ricans for electric power. Approximately, 93% of the country is electrified and the residential sector consumed 46% of electricity production in 1992. Industry accounted for an additional 30% of consumption in that year.[22] Demand for electricity is expected to grow rapidly as the country continues to industrialise. ICE (1994) projected demand growth of 5.5% in 1997, dropping to 4.6 by 2010. In an updated expansion plan ICE (1996) develops three scenarios for demand growth in electricity consumption. The scenarios begin from levels of just over five thousand GWh in 1996. By 2015, demand is projected to be 13,200 GWh under the low scenario, 15,400 GWh under the base scenario and 17,500 under the high scenario. Projected annual

[22] Ibid., 17.

growth in demand under the base scenario starts at 5.5% in 1996 and climbs to almost seven percent before gradually declining from 6.4% in 2005 back down to 5.6% in 2015.

As current exploitation of hydroelectric potential is just 11% of the country's potential, ICE's expansion plan for electric power generation in Costa Rica continues to center around hydroelectricity with increasing use of geothermal and wind. The base scenario for the expansion plan produced in 1996 confirms that hydroelectricity will continue to provide the bulk of the nation's electricity production through to 2015 (see **Figure 8**). Existing thermal plants that use bunker fuel will be retired from production once the Angostura hydroelectric project goes on line in 2000. Subsequent investments in thermal capacity will consist of cogeneration and modern gas turbines burning 100% diesel fuel.

Figure 8. SNI electric power expansion plan, 1996-2015



Source: ICE (1996)

Principal additions to power generation capacity under the base plan are listed in **Table 11**. A number of important run-of-stream and hydroelectric storage projects will be implemented during the 1996-2015 period. Siquirres and Pacuare are the only two large hydrostorage projects that will be built during this period. They will not come on line until late in the period. ICE suggests that only two sites remain with sufficient live storage volumes to be important for both annual and inter-annual purposes, Boruca (six times as large as Arenal) and Talamanca (over three times as large as Arenal). Neither of these projects appears as part of the projections to 2015 (only Boruca was included in the list of potential projects). In any case, both of these are probably off-limits, as they are mega-reservoir projects that involve the flooding of rare, protected wildlife habitat and the lands of (equally rare) indigenous peoples.

The expansion plan includes a number of projects that are sufficiently well underway to be considered fixed. These include projects up through and including the hydroelectric plant Pirrís in 2003. This suggests that the first available adjustments to the long-run capital expansion plan can be made beginning with the Tenorio geothermal project in 2004 and the Guayabo hydroelectric plant in 2006. This is roughly ten years from the initial date, 1996, employed in the projections of the expansion plan. Up until this point, changes in production from the ARCOSA complex will affect only short-run adjustments in running costs. Beyond 2003, such changes may be incorporated through changing the mix of projects or their implementation date. Thus the long run for the analysis is set to begin after ten years, i.e. in year eleven of the analysis.

Table 11. Costa Rican power generation expansion plan, base scenario 1996-2015

Year	System		Generation Projects					Costs		
	Energy GWh/yr	Capac. MW	Name	Capac. MW	Energy GWh/yr	Type/Fuel	Storage[a] Hm^3	Capital $/KW	O&M Fixed $/KW/yr	Running $/kWh
1996	5,021	903	Daniel Gutiérrez	20	89	Hydro	0.0	2,017	12.88	-
			Toro I (2nd unit)	12	46	Hydro	0.0	1,779	12.88	-
			Toro II	66	249	Hydro	0.2	858	12.88	-
			Tilarán[b]	20	94	Wind		na	na	na
1997	5,295	952	Various							
1998	5,623	1,011	CNFL Phase I	72	505	Cogen.		1,137	25.22	$0.034
			Miravalles II	55	389			2,280	27.14	
1999	6,007	1,080	CNFL Phase II	36	252	Cogen.		1,137	25.22	$0.034
			Miravalles III	28	205	Geotherm		3,344	27.14	-
			Tejona	20	94	Wind		1,430	1.35	7.44
2000	6,421	1,155	Angostura	177	869	Hydro	11.0	2,112	12.88	-
			Older thermal plants are retired: Colima, Barranca, S.A. Vapor and Moín Piston							
2001	6,850	1,232	Various							
2002	7,306	1,315	Various							
2003	7,792	1,404	Pirris	128	549	Hydro	42.6	2,240	12.88	-
2004	8,305	1,497	Tenorio	52	389	Geotherm		2,280	27.14	-
2005	9,373	1,690	None Listed							
2006	9,373	1,690	Guayabo	234	1,166	Hydro	1.6	2,190	12.88	-
2007	9,932	1,791	None Listed							
2008	10,513	1,895	Los Llanos	84	398	Hydro	1.5	1,433	12.88	-
			Laguna Hule	67	270	Hydro	0.0	1,493	12.88	-
			Moín Gas[c]	72	94	Diesel		614	8.00	$0.056
2009	1,113	2,003	Pacuare	158	761	Hydro	135.0	2,190	12.88	-
2010	11,737	2,115	Ayil	127	614	Hydro	33.0	1,956	12.88	-
2011	12,395	2,233	Siquirres I	206	1,334	Hydro	515.0	3,007	12.88	-
2012	13,091	2,357	None Listed							
2013	13,826	2,489	Siquirres II	206	226	Hydro	515.0	682	12.88	-
			Moín Gas	108	140	Diesel		614	8.00	$0.056
2014	14,603	2,627	Moín Gas	108	140	Diesel		614	8.00	$0.056
2015	15,424	2,774	Moín Gas	144	187	Diesel		614	8.00	$0.056

Source: ICE (1996).
Notes: Only the major projects are listed. Na refers to not available. [a]Storage capacity is live storage. [b]A private power generation project. [c]Energy production for new Moín Gas plants suggests a 15% capacity utilisation, however the original Moín Gas units are said to operate at 70% of capacity.

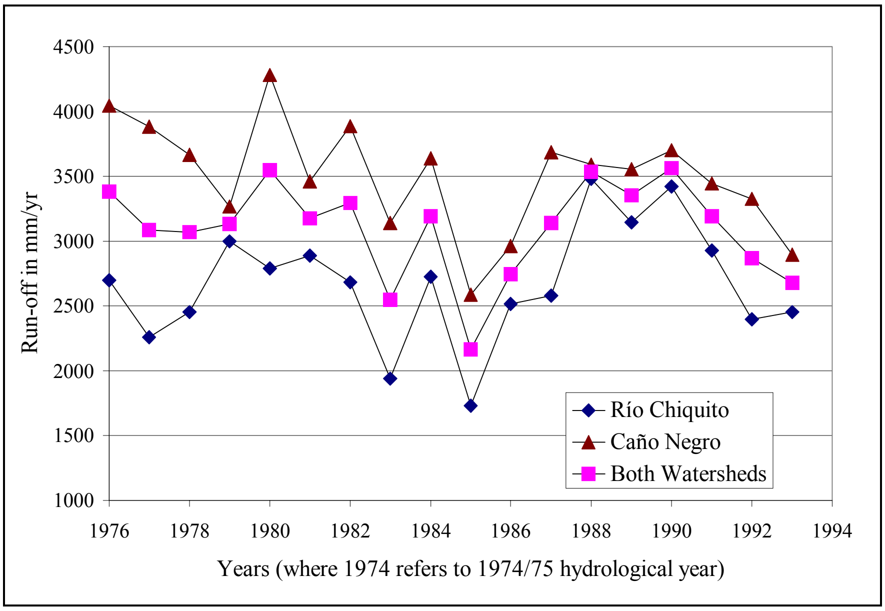
In the next sub-section the scenarios for the hydrological conditions are developed, leading to the corresponding specification of reservoir and SNI states.

Hydrological Conditions and the States of the SNI and Lake Arenal Calvo (1996) estimates the annual inflow of water to the lake to be 1,387 Hm^3. Interestingly, this figure is quite close to the figure (1,344 Hm^3) for the "live storage" recorded for the Arenal and Corobicí power plants in ICE's expansion plan (ICE 1996). Given that the true physical live storage capacity is closer to 2,000 Hm^3 (as reported earlier), this figure probably reflects ICE's estimate of the average annual inflow, thereby, confirming Calvo's estimate.

In order to operate at full capacity, ARCOSA requires a flow of 100 m^3/s. If operated at full capacity year-round this flow would total 3,154 Hm^3. Once filled, the reservoir appears to have been drawn down to below the 530-meter level only once, during the drought of 1994-95. By the end of the 1996 rainy season the dam had still not re-filled. The relationship between the storage capacity of the reservoir, the annual inflow and the installed capacity of ARCOSA confirms that the Arenal reservoir is an inter-annual storage reservoir, perhaps more by necessity than choice. The inflow received every year is not sufficient to support a yearly oscillation between the maximum and minimum operation levels. For reasons having to do with topography the capacity of the dam is quite large compared to the size of the surrounding watershed and the resulting annual inflow. Thus, ARCOSA must serve the dual purpose of regulating annual flows and providing inter-annual security. The next largest reservoir in the SNI is Cachí, which is a daily reservoir with a live storage capacity of just 38 Hm^3.

The manner in which hydrological conditions are hypothesised to play out over time will depend on assumptions regarding the serial dependence of river flows: in other words, whether wet periods are correlated with other wet periods or whether wet years are simply interspersed with dry years. Little (1954) argues that there are important and significant positive serial linear dependencies in river flows. Bolaños (1982) analyses run-off data at the Arenal and Boruca dam sites for the hydrological years 1957/58 to 1978/79 and finds evidence that successive dry and wet seasons are correlated, although this relationship is not as strong in the Arenal case. These sites are on opposite ends of the country and on different sides of the continental divide. Consequently, this demonstrates that wet and dry conditions are likely to be consistent across the country and affect all hydroelectric plants in a similar fashion. However, a review of water yield data compiled by Calvo for the Río Chiquito and Caño Negro watersheds over the period 1976/77-1993/94 does not display a marked interdependence as seen in **Figure 9**.

Figure 9. Annual Water Yield for Río Chiquito and Caño Negro, 1976/77-1993/94



Source: Calvo (1996)

A rough projection of the periodicity of hydrological periods can be pieced together by reviewing Río Chiquito and Caño Negro flow records and the history of water levels at Lake

Arenal. During the first six years of operation of the lake there was only one dry year, 1982, when lake levels dipped down to approximately 535 meters. Although data was not obtained on lake levels from 1985/86 through 1989/90 a review of flow data during this period (as shown in **Figure 9**) suggests that this was a very wet period. Consequently, it is likely that the dam was operated at full blocked states in successive wet seasons during this period. Decreasing flow levels in the 1991/92-93/94 period were followed by extremely dry conditions in 1994/95 and 1995/96. Anecdotal evidence suggests that rains were restored to normal in 1996/97, but that El Niño led to drought conditions again in 1997/1998. Data obtained from ICE on water levels for calendar years 1990 through 1995 shows that water levels started dropping off in the dry season of 1992. In 1994 and 1995 the lake level dropped below 530 meters for the first time since it was filled. As mentioned earlier, in 1994, petroleum based thermal power sources accounted for 22% of production, underscoring the critical nature of this drawdown.

Essentially the data series on run-off and reservoir patterns is of too short a duration to predict historical patterns regarding hydrological conditions in the case of Arenal. This is at least in part a result of the change in monitoring sites once the dam was built. In addition, the reservoir pattern reflects the influence of demand and supply conditions, and not merely hydrological conditions. Given the need of modeling these effects over a very long planning horizon, the distinction between hydrological regimes is therefore simplified to two periods, one dry and one wet. In order to effectively simulate reservoir operation it is necessary to stipulate the length and order of wet and dry periods, as well as the status of the reservoir at the end of the each season. As the reservoir has just recently been subject to a major drawdown under the drought conditions of 1994 and 1995 and 1998, it is assumed that the wet period will occur first in the modeling sequence. The length of the wet and dry periods is based on the observation that the drawdown accompanying the dry spell that began in 1991/92 lasted approximately five years. Thus, the base case for the modeling exercise is a fifteen-year cycle in which the wet period is ten years and the dry period is five years. Given the expected variability in the value of water in these different periods the sensitivity analysis examines three different scenarios. In the first two scenarios the cycle is shortened to ten years with either a three or five year dry period. In the third scenario, the effect of changing the order in which the wet and dry periods occur in the base case is explored.

It is assumed that the reservoir is operated in the following manner over these hydrological periods. The reservoir begins (t equals zero) in a full state at the end of the wet season. Across a string of wet years the reservoir is assumed to operate with a rhythm similar to that of an annual hydrostorage facility, except that it does not empty. It produces more electricity in the dry season than in the wet season and as a consequence is drawn down during the former (but not all the way) and filled up during the latter. In other words, ARCOSA consumes the annual inflow during these years (more or less). In these years the reservoir is filled by the end of the wet season (i.e. is in a blocked state) and at the end of the dry season it is in a balanced state.

During a dry period, the observed pattern is for a gradual drawdown to occur as the period is prolonged. During such a period, the dam will be in balanced state at the end of both seasons. For the purposes of the modeling exercise the dry period is assumed to include both the drawdown and refill periods, i.e. a cycle from full to empty and back to full. Hence, in a five year dry period the dam is emptied at the end of the third dry season, that is midway through the five year period. During the last two and a half years the reservoir is assumed to slowly recuperate and will be at balanced states at the end of each season, ending up full at the end of the five year cycle. The latter half of the dry period then includes years that that are more normal in terms of precipitation, but during which the dam is still refilling. For dry periods of different duration, the assumption of emptying at the midpoint may be maintained.

Having specified the hydrological conditions and the corresponding status of the reservoir, the remaining information required to implement the change in productivity algorithm is to specify the response of the SNI to these hydrological conditions. As seen below the potential of ARCOSA to supply secondary power is primarily an issue in the short run. However, as reported by ICE the opportunities to market secondary power within Central America are currently limited due to the "weak" degree of the present interconnection between country systems (ICE 1994). Current exchanges are limited and are often a matter of alleviating emergency situations. A number of initiatives to upgrade the Electrical Interconnection System of Central American Countries (SIPAC) and connect SIPAC to Mexico are in development and may spur the development of larger hydroelectric projects in the future. As pointed out above, however, these mega-dam projects may be too environmentally and socially risky to be built, and in the absence of these projects it is not clear how strong the impetus will be for this additional interconnection infrastructure. In the absence of these projects the likelihood that significant amounts of secondary power will be exchanged is low. The analysis thus assumes no unmet demand from secondary power.

As previously stated the short-run analysis will be conducted over the first ten years of the simulation. During the short run it is assumed that there is likely to be no power savings realised by increasing water supply to the Arenal reservoir during the wet season of years with normal or excessive rainfall. This, as HEP will be sufficient to meet demand without calling on thermal sources. In the dry season of wet years, it is assumed that thermal power generation will be required to supplement HEP. Low-cost, continuous power sources will be drawn upon and placed over the run-of-stream plants in the stacking pattern. This implies using the older plants that burn Bunker Coal or a mixture of Bunker and Diesel, until the cogeneration plants being built by the National Power and Light Company (CNFL) come on-line in 1998-99. These plants are twice as expensive to build as the newer gas turbine plants at Moín (GT) plants but cost almost half as much to run (see **Table 11**).

During short-run dry periods, it is assumed that changes in production from Arenal in the wet season will be supplied by the modern gas turbines at Moín. At such times all of the older plants would already be running flat out. It is assumed that in the dry season of a dry period power shortfalls will be met by power shortage in the short run. This, given the delays of the past few years in implementing the expansion plan and the lessons of the 1995-1996 dry period experience.

In the long run the system can be assumed to adapt supply to produce lower variable cost electricity and avoid shortfalls. Nevertheless, the current expansion plan demonstrates that demand for electricity in Costa Rica is expected to grow strongly for some time to come. As demand will increase in the wet, as well as dry, seasons continued investment in hydroelectricity projects is likely. As stated earlier and is evident from the expansion plan, there is considerable untapped potential available. Therefore, it is improbable that a change in supply from ARCOSA would simply alter the amount of spilled water during the wet season of wet periods. Instead, such a change in supply would have an impact on the level of additional investments in hydroelectricity.

The base expansion plan shows that gas turbines are added rapidly to fulfill demand towards the end of the evaluation period. While this may reflect a lack of other project ideas in the pipeline, it may also reflect the increasing attractiveness of new gas turbine technology that can be built quickly and to any scale. In any case given the lack of other investments it is assumed that changes in the wet period, dry season production from ARCOSA will be met by adjustments in the scale of gas turbine investments at Moín. Similarly, during dry periods, the expansion plan suggests that these same gas turbines will be the alternative source of capacity. Again, these plants are particularly suited for meeting peakload demand in the wet season of a dry year and as a last resort may be run full out in the dry season of a dry year.

Table 12 summarises the reservoir status and the power sources that can be expected to substitute for changes in production from ARCOSA for each combination of season and hydrological period across the short and long run. With the hydrological cycle and these expected values specified it is possible to derive the changes in discharge and storage going forward using the algorithm presented earlier.

Table 12. Relevant alternative power sources for changes in production from ARCOSA

	Wet Period		Dry Period	
Season	Short run	Long run	Short run	Long run
Reservoir Status				
Wet Season	Blocked/Full	Blocked/Full	Balanced	Balanced
Dry Season	Balanced	Balanced	Blocked/Empty	Blocked/Empty
Alternative Power				
Wet Season	None	Hydro	Gas Turbine	Gas Turbine
Dry Season	Cogeneration	Gas Turbine	Shortfall	Gas Turbine

<u>Production Function for ARCOSA</u>

Following on the formula for potential energy it can be expected that discharge used to generate electrical energy will produce energy consistent with its mass and the height from which it drops (the "head"). In the absence of a pronounced "head" effect the relationship may be modeled by using discharge as the single independent variable in estimating a water conversion factor. Echeverría *et al*. (1997) and others explored these relationships using a data set from the three ARCOSA plants that consisted of 1098 days with half-hourly readings of water flows, height and power generation levels. Preliminary analysis eliminated height as a potential explanatory variable. The water conversion coefficients that are subsequently estimated are highly significant (above the 99% level) and the estimation equations succeed in explaining a considerable amount of the variance in generation rates. As the plants operate in sequence, the coefficients may be added together to arrive at a water conversion factor of 1.225 kWh/m^3.

Table 13. Regression analysis of water and hydroelectric power generation

	Arenal	Corobicí	Sandillal
Regression Coefficients (kWh/m^3)	0.535	0.604	0.086
Multiple R^2	0.743	0.686	0.975
R^2	0.551	0.471	0.951
Mean Value (kWh/m^3)	0.574	0.660	0.086
Observations	1,098	1,098	1,098
Standard Error of the Estimate	0.004	0.005	0.0002
Correlation Index	0.838	0.827	0.975
t-Statistic	127.94	123.82	403.48

Source: Echeverría *et al*. (1997).

Marginal Opportunity Costs

At the end of the section on reservoir operation the sources of power and long-run additions to power capacity that are expected to substitute for any short-run or long-run changes in production from ARCOSA were identified (**Table 12**). In this section the cost estimates used for each one of these power sources and capacities are developed.

The marginal costs that are used in both the short- and long-run analysis are based on the running costs as estimated by ICE and shown in **Table 11**. The running costs are $0.34/kWh for cogeneration and $0.056/kWh for gas turbines. In the expansion plan ICE employs a power shortfall cost of $1.20/kWh, a measure adopted here. In the long run, other HEP projects are marginal substitutes for power from ARCOSA, however, their running costs are, of course, zero. ICE develops an intertemporal scenario for the running costs of thermal plants based on World Bank projections of increases in fuel costs. The analysis presented here employs the same rate of growth in the real price of fuel inputs, calculated to be 1% based on the ICE (1996) data.

For the long-run analysis only rough estimates of capital and O&M costs are developed. Abstracting from the expansion plan presented in **Table 11**, a figure of $2,000 per KW of capacity is used to reflect the average cost of hydroelectric expansion. The cost of gas turbine expansion is $614 per KW as derived directly from the plan. A simple back of the envelope calculation and a 10% discount rate suggests annualised values for the capital costs of $200 per KW/yr and $61.40 per KW/yr.[23] ICE estimates the fixed costs of O&M as $12.88 per KW/yr and $8.00 per KW/yr for hydroelectricity and gas turbines respectively. Totaling the annual opportunity costs of postponing additions to system capacity for each type of power, and rounding off, yields $213/KW/yr for hydropower and $69/KW/yr for gas turbines.

As stated in Chapter 4, these cost figures are used as annualised figures and applied to the change in generating capacity as determined in each time period. This change in capacity is itself calculated by spreading the change in power generation expected during the period evenly across the days and hours of the period. In other words, the change in productivity (in kWh) is converted into an instantaneous potential (in KW) by finding the minimum capacity (KW) level at which the increase in power could result in a constant stream of power over the relevant season. This figure is then multiplied by the marginal rate of substitution for the alternative power source and the appropriate investment and O&M cost as derived above.

The marginal rate of substitution is simply a way of accommodating the differences that may exist between an HEP plant and a thermal plant in terms of transmission costs and losses, mechanical reliability and flexibility, and hydrological availability (Barbour *et al*. 1985). For the gas turbines at Moín and the new hydroelectric projects transmission losses are assumed to be equal to those from ARCOSA. ARCOSA is located at a comparable distance from the Central Valley, where the majority of electricity is consumed. The rate of substitution may, therefore, be assumed to be 1.0 in the case of other hydroelectric plants. It remains to estimate the difference between ARCOSA and the thermal power sources in terms of differences in mechanical reliability and flexibility, as well as hydrologic availability. Typically, these effects work in opposite directions and when accounted for in net may constitute up to a 35% increase in the effective cost of alternative thermal power (Barbour *et*

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[23] The capitalized value equals the annual 'rental' value divided by the interest rate.

al. 1985). As no information of this type is available and given the presumed efficiency and flexibility of these new plants the rate of substitution is set at 1.15.

The parameters for running costs, capital costs and marginal rates of substitution are summarised in **Table 14**.

Table 14. Cost and substitution parameters

Season	Wet Period Short run ($/kWh)	Wet Period Long run ($/KW/yr)	Dry Period Short run ($/kWh)	Dry Period Long run ($/KW/yr)
Alternative "Power" Source				
Wet Season	None	Hydro	Gas Turbine	Gas Turbine
Dry Season	Cogeneration	Gas Turbine	Shortfall	Gas Turbine
Running Costs				
Wet Season	na	0	0.056	0.056
Dry Season	0.034	0.034	1.20	0.056
Capital Costs				
Wet Season	na	213	na	69
Dry Season	na	69	na	69
Marginal Rates of Substitution				
Wet Season	1	1	1	1.15
Dry Season	1	1.15	1	1.15

Externalities

The results for the hydrological externalities are presented in **Table 15**. The first column in the table lists the different simulations undertaken as explained in the preceding sections of this chapter. The results under each scenario are organised in rows corresponding to the externalities associated with water yield gain, dead storage sedimentation, live storage sedimentation and total externalities. Reading across the table, the first column of results indicates the output of the simulation model when it is run for a change of a single cubic meter of the respective hydrological output. In other words, under the first scenario the present value of the externalities caused by an increase in water yield of one cubic meter is $0.403. An increase in live storage sedimentation of one cubic meter causes negative externalities of $5.74. Succeeding columns present the results for each of the seven land units that result from the overlay of the three maps: forest type, location and life zone.

The first observation that can be made about the base case results is that positive externalities are recorded for all of the different land units, with per hectare present values ranging from $250 to $1,100. While sediment has a much larger impact per cubic meter than does water yield, the superiority of water yield changes in terms of volume swamps the effect of sedimentation when converted to a per hectare basis. The cost of live storage sedimentation does not exceed $100/ha under any of the land units, while the benefit of water yield gains range from $275 to $1,150/ha. The positive impact on production from the sedimentation of dead storage is, as expected, fairly marginal in comparison to the other impacts, never exceeding $6/ha in total.

Table 15. Hydrological Externalities: Results of Simulations

		Non-Cloud Forest Areas				Cloud Forest		
Location		Lower	Middle	Middle	Middle	Lower	Middle	Upper
Life Zone		bmh-P	bmh-P	bp-P	bp-MB	bmh-P	bmh-P	bmh-MB
Land Unit		A	B	C	D	E	F	G
Sedimentation (total)		13	28	28	28	13	28	16
Water Yield Gain		2,850	2,850	2,580	1,870	690	2,760	2,760
	($ per m^3)	($/ha)	($/ha)	($/ha)	($/ha)	($/ha)	($/ha)	($/ha)
Base Case								
Water Yield Gain	0.403	1,149	1,149	1,040	754	278	1,112	1,112
Dead Storage	0.403	3	6	6	6	3	6	3
Live Storage	-5.74	-37	-80	-80	-80	-37	-80	-46
Total		1,114	1,074	965	679	243	1,038	1,070
Sensitivity 1: Switch order of hydrological periods so that dry period is First								
Water Yield Gain	2.07	5,900	5,900	5,341	3,871	1,428	5,713	5,713
Dead Storage	2.07	13	29	29	29	13	29	17
Live Storage	-5.63	-37	-79	-79	-79	-37	-79	-45
Total		5,876	5,850	5,291	3,821	1,405	5,663	5,685
Sensitivity 2: Switch length of wet period and dry period to seven and three years respectively								
Water Yield Gain	0.991	2,824	2,824	2,557	1,853	684	2,735	2,735
Dead Storage	0.991	6	14	14	14	6	14	8
Live Storage	-11.5	-75	-161	-161	-161	-75	-161	-92
Total		2,756	2,677	2,410	1,706	615	2,588	2,651
Sensitivity 3: Switch length of wet period and dry period to five years each								
Water Yield Gain	1.55	4,418	4,418	3,999	2,899	1,070	4,278	4,278
Dead Storage	1.55	10	22	22	22	10	22	12
Live Storage	-8.92	-58	-125	-125	-125	-58	-125	-71
Total		4,370	4,314	3,896	2,795	1,022	4,175	4,219
Sensitivity 4: Change percent of inflow accruing in dry season set to 0% (i.e. 100% in wet season)								
Water Yield Gain	0.283	807	807	730	529	195	781	781
Dead Storage	0.283	2	4	4	4	2	4	2
Live Storage	-5.74	-37	-80	-80	-80	-37	-80	-46
Total		771	730	654	453	160	705	737
Sensitivity 5: Total externalities go to zero when percent of change in inflow gain accruing in the dry season is -56% and -48% for Land Units A and E respectively.								
Sensitivity 6: Discount rate set to 7%								
Water Yield Gain	0.580	1,653	1,653	1,496	1,085	400	1,601	1,601
Dead Storage	0.580	4	8	8	8	4	8	5
Live Storage	-9.64	-63	-135	-135	-135	-63	-135	-77
Total		1,594	1,526	1,370	958	341	1,474	1,528
Sensitivity 7: Discount rate set to 11%								
Water Yield Gain	0.296	844	844	764	554	204	817	817
Dead Storage	0.296	2	4	4	4	2	4	2
Live Storage	-3.75	-24	-53	-53	-53	-24	-53	-30
Total		821	795	715	505	182	769	789

Notes: All figures are present values over the one hundred-year planning horizon.

In the case of non-cloud forest areas, the externalities tend to decrease moving from the lower to the upper watershed areas (from $1,100 to $700/ha). Changes in sedimentation rates are lower and water yield gain is higher in the lower watershed. The pattern is reversed for pastureland use units located in cloud forest areas. The size of the hydrological externalities increases moving from the lower to the upper watershed. This primarily reflects the greater level of fragmentation assumed for the cloud forest in the upper watershed. The variation in the total value of hydrological externalities in cloud forest areas runs from $250 to $1,100/ha.

Examining the results for the base case in more detail, **Table 16** presents summary data on a number of relevant non-monetary indicators, as well as the breakdown of the externalities by running and capital cost components. To simplify the presentation only four of the land units are reported on in the table, selecting those with the greatest range in results from cloud and non-cloud forest areas. The first row indicates the cumulative loss in live storage capacity up through the year 70 of the analysis. As the sedimentation rates do not vary over time, the loss in live storage capacity is fairly straightforward and ranges from 455 to 980 m^3/ha.[24] The dominance of the water yield gain effect is witnessed by comparing this loss of storage in year 70 with the annual gain in water yield. For all but one of the land units, this water yield gain exceeds 1,870 m^3/ha/yr. In other words, even in year 70 the cumulative effect of live storage sedimentation is still outweighed by the water yield effect.

Table 16. Hydrological externalities: base case results in detail

		Non-cloud Forest Areas		Cloud Forest	
Land Unit		A	D	E	G
Changes in:					
Live Storage Capacity	(m^3/ha)	(455)	(980)	(455)	(560)
Discharge	(m^3/ha)	178,629	117,960	43,629	173,120
Spillage	(m^3/ha)	(11,538)	(24,850)	(11,538)	(14,200)
Storage	(m^3/ha)	21,781	14,900	5,581	21,200
Productivity	(kWh/ha)	218,820	144,501	53,445	212,072
Max Generation Capacity	(KW/ha)	0.65	0.60	0.24	0.66
Present Values					
Running Costs	($/ha)	861	501	176	823
Capital Costs	($/ha)	254	178	67	248
Total	($/ha)	1,115	680	244	1,070

Note: Total present values may differ slightly from those in **Table 15** due to rounding

Comparing the summary data on discharge and spillage suggests that only 10% as much of the gain in discharge is spilled. In other words, the bulk of the additions to water yield are used to produce electricity. The figures for storage simply represent the summation of changes in storage due to live storage sedimentation over the length of the analysis. As the units of water stored forward are subsequently either discharged or spilled, this figure simply indicates the extent to which the effects of hydrological change are lagged.

The summary data on changes in productivity suggest that a net gain of from 50,000 to 220,000 kWh of electricity may be garnered from a single hectare of pasture over the 70-year period. The figure for generating capacity represents the maximum value encountered over the planning horizon of generating capacity that may be postponed one period due to the influence of one hectare of pasture. The trend in these values actually varies greatly with the season. For example, in the dry season of the first year (in the long run) the value for land unit A is 0.21 and in the wet season it is 0.54. Over time the dry season value declines until at year 70 it is 0.08, while the wet season value increases until it reaches the maximum of 0.65 in the last year. This pattern reflects the fact that the negative impact of the

[24] Note that over a 100-year interval the losses would have varied from 650 m^3/ha to 1400 m^3/ha. This reflects the grossing up of the effect of yearly sediment delivered to live storage (6.5 m^3/ha/yr and 14 m^3/ha/yr respectively) for land units A and D, respectively.

accumulating change in live storage capacity is actually felt during the dry season of wet periods as an increasing loss of discharge. Meanwhile in the wet season, the loss of storage capacity actually raises discharge as no water is spilled in the long run, but rather is used to postpone the addition of new HEP. This increasing divergence of the capacity values occurs only during wet periods. During dry periods the effect of a change in live storage capacity is not felt in terms of a change in actual live storage as the reservoir is either balanced or empty at the end of each season.

With regard to the source of externalities as either running or capital costs, the share of running costs is roughly 75% of the total externality value. As indicated earlier, running costs for thermal plants are inflated at one percent per year in real terms, while capital costs are assumed to be constant over the evaluation period. However, only a small portion of the externality differential between running and capital costs, results from this assumption. If the model is repeated for land unit A without this rise in the price of fuel, the running cost component falls by just over 15% (i.e. by \$125/ha) and running costs remain the main contributor to the externalities.

These results may be used to derive rough figures for the total externalities associated with the use of pasture in Río Chiquito under the base case scenario. This is possible as the change in sediment and water yield is constant over time and thus the results can be expressed per unit net present values of sedimentation and water yield. If total sediment generated from pasturelands in Río Chiquito that goes to the live storage is roughly 45,000 m^3/yr, the present value cost of this continued flow of sediment is \$258,000 (45,000 m^3 multiplied by \$5.74/m^3). Similarly, taking roughly 5,000 hectares of pasture producing on average a 2,000 m^3/ha/yr gain in water yield nets a total positive externality of \$4 million for the watershed.[25] The total hydrological externalities associated with pasture in the watershed would be positive, generating net present benefits of roughly \$3.75 million.

Continuing with this rough estimation for the watershed, the net effect of such levels of sedimentation and water yield gain would be to produce a yearly increase of approximately 12.5 GWh of electricity in year fifteen of the analysis.[26] Judging from the generation expansion plan this is equivalent to about 0.01% of national demand in fifteen years time. The wet and dry season figures for the change in power generation capacity, 0.16 and 0.38 KW, represent 0.4% to 1% of the installed generating capacity of ARCOSA. Thus, the change in production due to land use in the watershed can be regarded as marginal relative to total supply or demand. Nonetheless, when translated into per hectare values these externalities are very significant.

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[25] As the changes in evapotranspiration estimates are not converted into map form, there does not exist a comparable watershed total for the change in water yield due to pasture. The estimate of 2,000 m^3/ha/yr is justified considering that a large portion of the non-cloud forest pasture occurs in the lower watershed where the difference can be expected to be 2,420 m^3/ha/yr. and that the ranges for water yield change in cloud forest areas suggests an average figure of just under 2,000 m^3/ha/yr.

[26] These figures are derived by repeating the simulation for a water yield gain of 2000 m^3/ha/yr and a total sediment yield of 20 m^3/ha/yr. The latter figure is based on a total sediment yield of roughly 100,000 m^3/yr spread over 5,000 hectares of pasture. The result is a productivity change of 2,500 kWh/ha in year 15.

Sensitivity Analysis

The sensitivity analysis begins with an examination of the responsiveness of the results to the hydrological/meteorological assumptions regarding the order and length of the wet and dry periods used in the analysis. The effect of switching the first hydrological period to a dry period, but maintaining the five-year dry and ten-year wet period cycle, increases the hydrological externalities by a factor of five or more. In the base case, shortfall costs are not incurred as the first dry period occurs only in the long run. However, by switching the order of the periods around, shortfall costs are incurred. In addition, as the shortfalls are incurred in the first few years they are not heavily discounted. This sensitivity analysis, highlights then, the importance of water yield gains during dry periods.

In the second sensitivity analysis the length of the full wet and dry period cycle is shortened to ten years, consisting of a seven-year wet period followed by a three-year dry cycle. Externalities are more than doubled as shortfall costs are incurred again in the short run. The increase in externalities is less in this case, however, as shortfalls are incurred for a shorter period of time and further into the planning horizon. Switching the cycle to a five-year wet period followed by a five-year dry period leads to even higher externalities, although not as pronounced as those observed by switching the order in which these periods occur.

The results of these analyses suggest that by putting a long wet period as the initial hydrological period, the base case provides fairly conservative estimates of the positive externalities. Altering this assumption raises the externalities significantly. The costs of sedimentation rise, but such small increases are overwhelmed by the large returns from avoiding power shortfalls in the short run.

The next two sensitivity analyses assess how responsive the results are to the assumption made regarding seasonal flows. Historical data show that 25% of water yield in Río Chiquito enters Lake Arenal during the dry season. The water yield gain is thus apportioned 25% to the dry season and 75% to the wet season in the base case scenario. This data, however, reflects the effect of existing land use patterns in the watershed. Were the effect of having forest in place of pasture to shift water from the wet season to the dry season this might lessen the positive externalities observed under pasture in the base case. In order to simulate the potential effects of such a change in the seasonal timing of run-off the simulation is run assuming that 100% of the water yield gain accrues in the wet season under pasture. That is, none of this gain arrives during the season when it would be most valuable. Because of this change, the positive externalities are reduced from 35% to 50%. Nonetheless, pasture still produces significant positive externalities.

Of course, in the extreme case, having pasture in place of forest may result in a net shift of run-off from the dry season to the rainy season as dry season baseflow diminishes. In a second sensitivity analysis, it was assumed that the entire increment in water yield accrued in the wet season under pasture. An iterative process was then followed to find out how much more of the dry season run-off would need to be switched to the wet season (under pasture) in order to arrive at externalities of zero. In the case of land unit A the results suggested that this switchover point occurs at -56% of the water yield gain, or roughly 160 mm/yr.[27] Recalling from **Figure 9** that average yearly run-off totals for Río Chiquito are around 2500 mm/yr suggests that such a redistribution is significant, representing one-quarter of existing dry season run-off (160 of roughly 625 mm/yr). This analysis is then repeated using land unit E, the one that showed the lowest initial externalities. The results suggest that total externalities go to zero with only a shift of 33 mm/yr in dry season run-off (-48% on a water yield gain of 690 m^3/ha/yr).

Thus, while the size of the shift required will vary, in most cases an exceedingly strong seasonal shift would be necessary in order for the net hydrological effect of the land use change to be ambiguous.

The results show considerable responsiveness to the use of different discount rates. An increase of from 40% in the externalities is observed by lowering the discount rate to 7%. Although an intuitive result, it is still of interest to note that the lower discount rate increases the positive externality associated with pasture. Note that due to the limited nature of the sediment impact, the externalities over the planning horizon continue to increase as successively lower discount rates are applied. A reduction in the externalities observed for the different land units of 26% of their original values results from raising the discount rate to 11%. This primarily reflects the lowered present values for the externalities associated with water yield gains.

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27 As a volume change of 1,000 m^3/ha/yr is equivalent to a point measure of 100 mm/yr the percentage figure applies to the 285 mm/yr which is that annual yield gain.

Evaluation of Existing Market and Policy Incentives

In this section the results from the livestock and hydrological analyses are combined in order to evaluate the conceptual framework presented at the outset of this paper. The first sub-section clarifies how and which results are extracted from these analyses and how they are integrated into the conceptual framework. Each of the three phases of the framework are then evaluated based on this information and conclusions drawn regarding policy and market incentives for watershed protection in Río Chiquito.

Methodology

The analysis of private returns is based primarily on the cost-benefit analysis using the results for the private opportunity costs scenario, with occasional reference to the cash flow analysis as well. The attractiveness of land use alternatives to landholders is limited to the private analysis of returns to production forestry as derived from Bolaños *et al.* (1996).

The economic analysis of livestock production is divided into three parts under the framework. First, economic returns to livestock production are reviewed given the removal of policy distortions affecting input and output prices. In the second step potential private inefficiencies in production brought on by market or policy failures are examined. The analysis of the user costs of soil erosion presented does not support the contention that such costs are significant (at least within the time frame under evaluation) and thus no user costs are explicitly included in the quantitative analysis. In the final step, environmental externalities are considered. The hydrological impacts of sedimentation and changes in water yield are included in the evaluation at this point.

As suggested earlier the evaluation of externalities implicitly involves a comparison of two land uses in Río Chiquito: livestock production and forest protection. As the costs of ensuring a lasting conversion and protection of forest cover are of a significant order of magnitude these are also included in the analysis based on studies of the direct costs of park and reserve protection in Costa Rica. The quantitative assessment of net economic returns is considered then in light of a qualitative discussion of other potentially significant environmental externalities in the Río Chiquito area. As carbon fixation values are expected to be the most significant of these externalities, rough estimates of their potential value are provided.

The presentation of results is accomplished with reference to a fairly simple spatial representation of the values involved. Due to the lack of reliable geo-referencing of the land holdings, a spatial overlay of returns is not conducted by holding. Instead the spatial concentration of each of the seven types of holdings identified in the cost-benefit analysis of livestock production is assessed in deriving approximate locations within the watershed for these holdings (based on **Figure 1**). Returns to forestry production, hydrological externalities and the direct costs of forest protection are then incorporated using overlays that reflect the principal variation in these values across the respective land use units in the watershed as described below.

For the assessment of private returns the different types of livestock holdings are used to represent returns to producers. Assessment of forestry production values are differentiated by life zones, thus the life zones (see **Figure 7**) are overlaid with the production types to obtain relevant land use units. Due to the relatively small area of the upper watershed that is classified as Lower Montane Wet Forest (bmh-MB) no values are reported for this life zone in Bolaños *et al.* (1996). Given its proximity to the Lower Montane Rainforest (bp-MB) the values for this life zone are used for Lower Montane Wet Forest areas. In total, nine combinations of production type and life zone are evaluated under private returns.

The evaluation of economic returns is accomplished using the seven types of land use units developed in the analysis of hydrological externalities as a base. This reflects the integration of three spatial overlays: (1) the classification of values produced by the CALSITE sediment yield map according to whether the holding is in the lower, middle or upper watershed (see **Figure 5** and **Figure 6**), (2) the classification of water yield gains from pasture in non-cloud forest areas by Life Zone (**Figure 7**) and (3) the classification of production types in cloud forest areas by the expected water yield gain from horizontal precipitation in forest fragments. The latter two overlays are themselves based on the geographic positioning of livestock holdings and life zones in Río Chiquito (see **Figure 1**). The direct costs of protection are also included in this analysis but are assumed to be equal for all land use units. Values for livestock production under the seven original production types are then combined with the respective hydrological externalities and direct costs of protection to arrive at the economic assessment of the evaluation framework. A total of thirteen different combinations of land use units are selected to illustrate the range of results obtained.

Private Returns

The analysis of private returns to livestock production demonstrates that large production units involved in ranching, dual purpose ranching and dairy generate substantial cash flow and private returns to land. The private returns to these larger units range from $884/ha for ranching to $1,413/ha for dual purpose ranching and up to $6000/ha for dairy. As shown in **Table 17** and in **Table 18**, the potential returns to forestry production options are unlikely to be competitive with livestock production on such holdings. The only forestry production option showing a positive return is that of natural regeneration. Yet even so, it can easily be seen that a switch to such production by a large rancher would entail a loss of $862/ha on average. Admittedly, the returns to forestry production as calculated exclude the effect of taxes and incentives, which can be expected to be positive on net. Nonetheless, it is clear that the prospects of obtaining reforestation incentives net of taxes and other regulatory costs in the $800/ha range are unlikely.

Turning to the smaller production units, returns to small dual purpose operations selling milk to Monteverde are modestly positive at $327/ha. Meanwhile, smaller ranching units (less than 80 hectares), dual purpose holdings involved in cheese production and small dairy units selling to Monteverde produce net negative private returns, although the returns to the latter two production types are close to the break even point. As explored in the sensitivity analysis a lower than expected implicit opportunity cost of family labour and rancher expectations regarding upward trends in future beef prices may resolve the apparent anomaly of these negative private returns to land. Certainly, the minimal nature of the expected returns to forestry production options provides a measure of explanation for why those land-holders that have negative returns are not switching in large numbers to forest production. However, the existence of incentives programs has engendered an interest, particularly on the part of smallholders residing in or near the watershed, in reforestation initiatives (as described in the next section). Finally, the effect of significant and positive cash flows and the difficulties of actually selling and relocating to other employment opportunities may also weigh heavily on the decisions of producers who are failing to produce positive returns to land.

Table 17. Private analysis of ranching and dairy

(all figures NPVs in $/ha)	Ranching			Dairy	
Production Types	Small	Large	Large	Dos Pinos	Monteverde
Life Zones	bmh-P	bmh-P	bp-P	bmh-P	bmh-MB
Livestock	(634)	884	884	6,032	(94)
Forestry					
Natural Regeneration	1	1	22	15	16
Management	(217)	(217)	(39)	(52)	(52)
Plantation	(554)	(554)	(429)	(429)	(429)

Table 18. Private analysis of dual purpose ranching

(all figures NPVs in $/ha)	Cheese Producers	Dos Pinos	Monteverde Producers	
Life Zones	bmh-P	bmh-P	bmh-P	bp-MB
Livestock	(81)	1,413	327	327
Forestry				
Natural Regeneration	1	1	1	16
Management	(217)	(217)	(217)	(52)
Plantation	(554)	(554)	(554)	(429)

On-Site Economic Returns

The analysis of economic returns begins with a number of production types of only borderline profitability in private terms. The ensuing analysis of policy distortions reveals only a number of relatively minor taxes imposed on inputs. The result is to actually raise economic returns above the level of private returns. As can be seen from the three tables that follow the implication is that from an economic point of view the only production type yielding negative returns are the smallholder ranches. The large negative size of the returns (almost -$600) to these holdings does suggest that these production units might serve the economy better if they were devoted to other uses, whether forest regeneration or a move into another form of livestock production. Of course, the subsequent analysis of hydrological externalities leads to another conclusion.

Before turning to externalities, however, it is important to stress that the analysis of user costs was unsuccessful in finding any systematic linkage between the age of pasture on livestock holdings and production levels. As a result it appears that there is little leverage available in terms of the manipulation of market failures or policy distortions that affect the landholders' intertemporal profile of soil usage.

As suggested earlier, there are four potential villains that might lead ranchers to adopt a more than economically optimal rate of soil erosion. The first of these referred to a potential divergence between the private and social discount rates. Aylward and Porras (1998) shed considerable doubt on this possibility by demonstrating the convergence of the consumption rate of interest and the opportunity cost of capital in Costa Rican capital markets. Insecurity over tenure is another potential villain. In the case of Río Chiquito tenure insecurity appears, by and large, to be absent. Although formal compliance with all the legal requirements related to land ownership remains somewhat delayed in Río Chiquito, the field survey did not turn up any evidence of tenure insecurity. Indeed, there was ample evidence of the ability of local landholders to control the future of their land. A relatively mature market for land is also found in the area further adding to the relative feeling of stability with regard to tenure issues in the watershed. As much of the colonisation of the area occurred by the 1970s and there are no unclaimed lands (*tierras baldías*) remaining in the watershed this is not a surprising result.

The third potential problem regards the ability of asset markets for land to accurately reflect land values. This is an important issue in Río Chiquito, particularly as it relates to non-livestock production values. Land speculation for ecotourism is one possibility that is often mentioned. Clearly, the value of pasture lands for this purpose is largely unknown and thus essentially any transaction is largely speculative. However, as regards the productivity of land for livestock production, it can be assumed that buyers are reasonably aware of soil conditions in the area and that this aspect of land purchase is incorporated in the price. A final concern regards the level of information regarding the long-run impact on productivity of current land husbandry practices. Given the age of some of the landholdings under production in Río Chiquito and the pattern of handing down parcels from one generation to the next, it may be assumed that information on this topic is relatively good. That is, however, not to say conclusively that it may not be a problem.

Thus a consideration of the potential underlying causes that might be leading to an excessive intertemporal pattern of soil erosion in the watershed fail to illuminate any forces that may be driving such a problem. Given that the quantitative analysis failed to turn up evidence in support of the contention that landholders may be incurring large user costs of soil erosion there is little basis on which to support this contention (or actions designed to ameliorate it). This is not to say that soil erosion is not and has never been a problem in the watershed. The consultations carried out under the project with ranchers in the watershed revealed that in certain places significant land degradation has occurred. In this regard, the issue of soil loss and the last two concerns mentioned above are taken up again in the next section of the report. However, it does not appear that the problem is of such a magnitude as to warrant a significant realignment of land use patterns in the watershed. More likely it is a matter of confronting the problem on specific sites within particular holdings.

Societal Incentives and External Costs

Prior to considering externalities, then, for the most part the existing patterns of livestock production in Río Chiquito show either marginal or considerable economic returns to land. Due to the large and positive magnitude of the hydrological externalities associated with livestock production, inclusion of these externalities simply increases the economic returns to land for each of the land use combinations displayed in the tables. For the sake of consistency, the costs that would be incurred by protecting a regenerating forest on lands currently in livestock production are also included as positive benefits accruing to livestock production. Adding in the cost of actually protecting such a forest (as a benefit in this case) simply ensures that all of the relevant external costs and benefits are accounted for in the analysis. Under this assumption, in order to guarantee the regeneration of forest cover society would have to pay the $160/ha required to protect and maintain the area (Mejías, pers. com 1997).[28]

As suggested earlier, the intuition here is that in valuing externalities, particularly those caused by changes in environmental function, the analysis must explicitly trade off the level of environmental function under the existing land use (pasture) and a land use that is less degrading. In the case of the analysis of externalities in Río Chiquito, hydrological function under pasture is compared to that under primary forest. However, to actually realise these hydrological changes certain economic implications persist. If pasture is maintained, the net benefits of livestock production are generated and if forest is regenerated the direct costs of forest protection are incurred (as well as hydrological and other benefits). Of course the land could simply be abandoned and left without protection, however, the susceptibility of the land to invasion and squatting implies that this course of action would not lead inevitably to the regeneration of forest cover and restoration of the hydrological function of the forest.

The analysis originally presaged in this research project implicitly presumed that economic returns would be positive before externalities were considered. Externalities, in turn, were expected to be negative and to outweigh the production benefits. In such a case, it would have been clear that if forest protection costs are significant a simple negative result once the externalities are subtracted from the returns to production is not sufficient to demonstrate that livestock production is inefficient in economic terms. Rather, it is imperative that the change in land use (from livestock to forest) generate sufficient returns to more than pay for the direct costs of forest protection. Otherwise the net value society gains by dropping livestock production in favour of improved hydrological function will be a marginal benefit that is less than zero. Note that this also is in advance of consideration of potential transaction costs in moving from pasture to forest.

Table 19. Economic analysis of ranching

(all values NPVs in $/ha)	Small Holdings			Large Holdings	
Externalities Classification	A.	E.	F.	A.	C.
Livestock	(588)	(588)	(588)	1,053	1,053
Hydrological Externalities	1,114	243	1,038	1,114	965
Costs of Protection	160	160	160	160	160
Net Economic Benefits	686	(185)	610	2,327	2,178
Location and Life Zone	Lower, bmh-P	Lower, West bmh-P	Mid, West bmh-P	Lower, bmh-P	Mid, Interior bp-P
Sediment and Water Yield	13 / 2850	13 / 690 cf	28 / 2760 cf	13 / 2850	28 / 2580

[28] The $160 figure is a rough indication of the net present value of protection costs (in perpetuity) based on studies of the Guanacaste Conservation Area and the Monteverde Cloud Forest Preserve.

Table 20. Economic analysis of dual purpose ranching

(all values NPVs in $/ha) Externalities Classification	Cheese Producers A.	 B.	Dos Pinos A.	Monteverde Producers E.	 D.
Livestock	33	33	1,874	605	605
Hydrological Externalities	1,114	1,074	1,114	243	679
Costs of Protection	160	160	160	160	160
Net Economic Benefits	1,307	1,267	3,148	1008	1,444
Location and Life Zone	Lower, bmh-P	Midwest bmh-P	Lower, bmh-P	Lower, West bmh-P	Mid, Interior bp-MB
Sediment and Water Yield	13 / 2850	28 / 2850	13 / 2850	13 / 690 cf	28 / 1870

Table 21. Economic analysis of dairy farming

(all values NPVs in $/ha) Externalities Classification	Dos Pinos Producers A.	 E.	Monteverde Producers G.
Livestock	7,408	7,408	257
Hydrological Externalities	1,114	243	1,070
Costs of Protection	160	160	160
Net Economic Benefits	8,682	7,811	1,487
Location and Life Zone	Lower, bmh-P	Lower, West, bmh-P	Upper, West, bmh-MB
Sediment and Water Yield	13 / 2850	13 / 690 cf	16 / 2760 cf

Notes for **Table 19**, **Table 20**, and **Table 21**: Sediment and water yield figures are in m^3/ha/yr. The "cf" after water yield indicates increased yield from adjacent cloud forest on the holding.

Given the calculations of net economic returns for the different land use combinations presented above, it is clear that private and societal incentives are generally consistent, favouring livestock production as it currently exists. The exceptions occur in the case of smallholders with private returns that are in the red even once externalities are incorporated. Given the variation in economic returns to livestock it can be argued that in the case of small ranchers, cheese producers and dairy farms selling to Monteverde there is an important market failure reflected in these holders' inability to capture the externalities. Their failure to capture these benefits will cause them to be more likely to abandon their use of the land (than if they could capture them). This would presumably lead to an inefficient land use allocation and endanger hydroelectric production levels. This argument may of course be extended to other holdings that have below average returns to production but still generate positive hydrological benefits. In such cases an incentives program aimed at ensuring continued livestock production might be justified on economic grounds.

An alternative and perhaps more efficient scenario is also suggested by the results. Under this scenario holdings producing superior returns (principally large-scale ranching and mechanised dairy production) would be encouraged to buy smaller, less productive holdings. Not only would average holding productivity be expected to rise, but the need to extend incentives to marginal producers would be largely eliminated. Clearly this alternative is of low feasibility given its lack of attention to practical concerns of socioeconomic equity. In addition, it should be noted that few large producers are found in cloud forest areas. To the extent that a takeover of these areas by large scale ranching and mechanised dairy might lead to increased conversion of remaining forest on these holdings, there would be a corresponding drop in water yield and hydrological benefits. In general, it might be expected that micro-management of the spatial pattern of forest/pasture for the capture of horizontal precipitation would be best accomplished on smaller, family operated farms. In this regard it is worth nothing that the small dairy producers in the upper watershed (that sell to Monteverde) have arrived coincidentally at the land use allocation that leads to one of the highest gains in water yield of all the production types.

To this point the analysis of the evaluation framework begs the question of what are the non-hydrological externalities of livestock production in Río Chiquito. Of the thirteen examples presented in the tables, the net economic returns are over $600/ha for all but the small ranchers case discussed in detail above. Further, as the sensitivity analysis has demonstrated, the use of the lower bound for the discount rate will only increase the value expected from livestock production and hydrological externalities. Employing the upper bound in turn drops both sets of returns by only 15%. In other words the results are robust over a reasonable range of uncertainty regarding the discount rate employed in the analysis. Implicitly then, the value of other negative externalities must exceed $600/ha (or more) in order to justify the development of large-scale incentives programs for altering land use in Río Chiquito.

Aylward *et al.* (1998) consider a range of potential on-site and off-site benefits that might be consistent with restoring original hydrological function under primary forest in the Río Chiquito area. A number of these benefits are on-site direct uses of the forest (hunting and biodiversity prospecting) and thus, are not true externalities nor are they fully consistent with full forest protection. If biodiversity prospecting can nonetheless, be temporarily included in the list of externalities (its benefits in the form of the development of new products can be considered off-site benefits) the list would include biodiversity prospecting, carbon fixation, tourism/recreation, and existence values. Of these potential externalities only carbon fixation appear to be of potential significance to Río Chiquito. The existence of large areas of primary forest adjacent to Río Chiquito that are already preserved, suggests that biodiversity, existence and ecotourism values associated with an increase in forest cover would be minimal. As for the value of carbon that would be fixed by natural regeneration in Río Chiquito, estimates by Aylward *et al.* (1998) suggest values in the range of $200 to $300 per hectare. Certainly, to argue for the conversion of existing land practices in the watershed on this basis would be difficult given the livestock and hydroelectric value added that is produced by these pastures.

Conclusions

Prior analyses of land use in Río Chiquito had concluded that livestock production was causing severe and lasting damage to the hydrological resources of the area. On this basis several proposals for a drastic change of land use in Río Chiquito have been mooted over the years. Given the decline in international beef prices, such concerns are currently often qualified by the statement that "in any case ranching is not even profitable." This report suggest that these positions must be seriously reevaluated, particularly in light of their potential for misleading policy and conservation programs for the Arenal area.

First of all, dairy and dual purpose ranching are not as dependent on international beef prices as is pure cattle ranching. Thus, there is a misstatement of the problem when the profitability of the land use in Río Chiquito is associated solely with the prospects of ranching. Nor is it clear that beef prices will necessarily remain low over the long run. Thus, there is the additional danger of basing such an analysis of long-run policies on a fairly volatile market variable. Second, the analysis of hydrological function and the resulting externalities suggest that the net effect of having pasture in place of forest (and of having pasture interspersed with cloud forest) are to cause an increase in hydroelectricity production. Given the two results it is clear that the analysis has shown that there does not appear to be a significant incentive problem, be it a market failure or a policy distortion, that impedes efficient land use at the watershed scale.

An exception to this case would be the case of those holdings (primarily smallholdings) that have negative private returns. As seen in the aggregate, the nature of input price distortions actually depress private returns, leading them into the red. Thus, one recommendation would be to continue the process of trade liberalisation in the country, in order that these distortions do not dissuade producers from engaging in productive activities that generate positive returns to the economy. In a similar vein, aggregate negative returns to livestock production on a number of production types (typically smallholdings) in the watershed suggest that finding mechanisms to internalise the hydrological externalities may also be important in sustaining these producers and the economic returns that they generate. Finally, the consideration of additional externalities suggests that in certain cases of borderline private profitability the potential of incentives for reforestation and carbon fixation may be a useful motivating factor given the potential economic gains that such uses may provide. Clearly, any efforts towards reforestation will need to address their expected effect on water yield. This is of particular importance in cloud forest areas, where the spatial distribution of reforestation efforts can be expected to have a very significant effect on the capture of horizontal precipitation in the dry season.

At a fairly aggregate scale the evaluation of the incentives framework suggests that there may be little need for gross changes in land use or the proposition of large-scale incentives or changes in existing policies. At a more disaggregated level, that of the holding or of sections of a holding, the issue of land use, economic returns and incentives programs may still be of great importance. This point may be argued based on the great variability of livestock production technologies, the economic returns of these production technologies and the underlying socioeconomic and biophysical conditions found in the watershed. For example, particular attention should be devoted to cloud forest areas, where cooperation between landholders and ICE has potential for yielding significant joint benefits. This emphasis on sub-holding level dynamics of land use management is explored at greater length in the companion paper by Aylward and Fernández González (1998).

Future research on this topic in Arenal might fruitfully explore two important information needs not resolved by in this paper. First, field-level empirical investigation and analysis of evapotranspiration and run-off under different land uses and vegetative covers, with particular regard to the seasonal timing of run-off, would greatly strengthen the existing conclusions. A specific point worth emphasising in this regard is the relationship between the degree of "patchiness" of a forest/pasture landscape and increments/decrements in water yield. Second, for the purpose of improving the ability to utilise the existing dataset in the prediction of the hydrological and economic consequences of actions and measures undertaken by landholders an accurate mapping of the geographical position of land holdings in the watershed is essential.

As with any case study it is tempting to over-generalise regarding the results obtained. However, given the variance that exists from one watershed to the next in terms of biophysical and socio-economic characteristics it is risky to suggest that the conclusions and recommendations of this study should be extended *carte blanche* to other watersheds. Nevertheless, it is clear that the methods and results are largely (and easily) applicable to the remainder of the Arenal watershed. Given similar constraints and opportunities the results should certainly be considered as relevant to the Central American context and perhaps to other tropical countries where livestock and hydroelectricity (from hydrostorage) are produced in the humid tropics. This is particularly the case for sites that have volcanic soils and significant areas of cloud forest. While considerable variability can be expected in applying the valuation analyses to other sites and conditions, at a minimum this case study suggests the benefits of a comprehensive and thorough approach to the valuation of hydrological externalities in the case of large hydropower reservoirs. Additional case studies and more general theoretical work would assist in the development of a defensible consensus around rules of thumb and shortcuts in such analyses that would contribute to better policy and project formulation. Such guidance seems necessary given the current reliance on partial analysis and outdated conventional wisdom of the benefits of watershed protection in the humid tropics.

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