A Knowledge and Assessment Guide to Support the Development of Payment Arrangements for Watershed Ecosystem Services (PWES)

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Table of Contents

Introduction

Statement of the problem:

Increasing value is being placed on services provided by watershed ecosystems. However, these services are often poorly defined and inadequately assessed, which impedes the development of market and institutional arrangements needed to pay for watershed conservation, so as to insure their continued provision. Given the inherent complexity and uncertainty of watershed processes that support service provision, not to mention the site-specific nature of these processes, responses to watershed degradation may also require a complex and flexible mix of market and institutional arrangements, if adaptations are to be made to changing conditions. However, in the general absence of ecological and economic assessments, initiatives to develop Payments for Watershed Ecosystem Services (PWES) are often based on myths about land and water relationships that may lead to partial or inappropriate solutions, which not only fail to solve problems of watershed degradation but may actually exacerbate them (Kaimowitz 2001) (UN FAO 2002). Although the perception that science can provide certainty is equally misleading, with a careful identification and prioritization of information needs, it should be feasible to obtain better approximations as to the magnitude and direction of impacts of management activities and significant causal factors, as well as to verify these through monitoring, and thereby, over time, improve information needed for decision-making.

Modeling and monitoring complex biophysical characteristics are challenging tasks. The same is true of the development and implementation of economic and institutional arrangements needed to control access to resources that have public good and common pool characteristics. Putting PWES in place usually requires that both these challenges be overcome, thereby assuring that environmental services are actually received in exchange for payments. If biophysical characteristics are not modeled and monitored accurately and if suitable economic and institutional arrangements are not put in place, stakeholders will be disenchanted with the payments scheme. In turn, disenchantment is bound to lead to the scheme's collapse and the loss of environmental benefits. Effectiveness of management actions will therefore depend on:

- the integrity of ecosystem functions that support service provision,
- the economic significance of impacts or benefits at the relevant scale, and
- the effectiveness of institutional arrangements needed to insure their provision to those who are entitled to them.

Over the long term, a comprehensive, integrated and site-specific assessment of biophysical characteristics is critical as a basis for monitoring and evaluating the environmental services that payments actually elicit. However, given the large temporal and spatial scales of watershed and ecosystem processes, and lag times between multiple causes and effects, a comprehensive assessment is generally not available before a PWES initiative is undertaken. Such an assessment typically requires steady and adequate long term funding for research. The lack of research of this sort should not be seen as an impediment to the development of PWES initiatives. Rather, initiatives should be structured so as to obtain this information needed for modeling and monitoring over time, and be sufficiently flexible so as to be able to adjust programs in response to better information as well as to changing conditions.

Assessment is also a fundamental component of adaptive management – an approach which explicitly recognizes that knowledge is always incomplete and that regards management strategies as experiments based on a hypothesis regarding expected outcomes. Unexpected outcomes or surprises then become opportunities for learning (Holling 1978; Gunderson, Holling et al. 1995). Therefore, a key element of effective management strategies is to maintain the flexibility needed to adjust programs in response to new information as it becomes available, often through unexpected occurrences .

In the development phase, the key challenge is to gain the confidence and collaboration of stakeholders in spite of the uncertainty, by providing a base of relevant information – including uncertainties, that facilitates:

- awareness of ecosystem services,
- awareness of conflicting uses that threaten their provision,
- implications of their loss for livelihoods and quality of life,
- the development of effective management plans that also insure stakeholder access to benefits, and
- equitable payment arrangements as an incentive for implementation.

An underlying premise of this project is that a good assessment can increase the confidence and willingness to pay and participate of direct resource users as well as external donors which may be interested in indirect or non-use values. However, it is often overlooked because of institutional pressures to show immediate and tangible "results" as well as the political difficulties presented by uncertainty. In other cases, assessment may provide extensive information focused on narrow technical questions that is not decision-relevant as it fails to address the key concerns of stakeholders. Use of the information by stakeholders may be the ultimate test of the quality and effectiveness of the assessment itself.

Project objectives and overview:

The overall objective of this project is to provide a knowledge guide that identifies the kinds of information needed from a site-specific assessment conducted for a PWES initiative. Although it provides an overview of approaches for evaluating biophysical, and economic, as well as institutional and policy aspects of payment arrangements, all of which are critical aspects of effectiveness, particular emphasis is on the biophysical aspects of defining and measuring ecosystem services. This is critical for building stakeholders WTP'for watershed environmental services, including enhancement of their confidence that they are getting what they are paying for. This matter of confidence-building has received less attention in current initiatives, in which the main focus has been on identification of potential buyers who have the ability to pay as well as systems for collection of payments (Pagiola 2002). Biophysical characteristics also have implications for the transaction costs and therefore the feasibility of institutional arrangements.

This report is particularly intended for use by project managers who may have specialized knowledge in particular fields but who may need a broader framework for inquiry addressing the site-specific context of a PWES initiative. A closely related purpose of this report is to help project managers organize a broad range of information in a way that is relevant and useful for decision-making. The report should also enable organizations and government officials interested in developing such initiatives, or donors interested in supporting their development, to evaluate opportunities and have a better appreciation of what may be required in practice.

The project team consists of individuals who have knowledge of hydrology, water resources engineering, economic valuation, and of policies and institutions that are needed to support PWES. Given the complexity of the topic, and the time and resource constraints on the project, and the need for a broader base of experience with actual assessment of PWES initiatives, the guide does not pretend to be comprehensive. However, it can be considered successful if it contributes to an ongoing process of learning based on the experiences of practitioners. In addition to this report, the project includes the "FLOWS" listserve on which new information will be reported.

The report begins with an overview of existing PWES initiatives. Their structure is examined, as are the services being paid for. In addition, the information and/or myths on which some of these initiatives are based are examined. This discussion provides a basis for presenting some lessons learned from the current body of experience and for drawing some preliminary conclusions regarding their effectiveness, drawn heavily from existing reviews.

Special attention is then given to the definition and quantification of watershed services and the ecosystem functions that support them. These services are examined in relation to variables associated with the water balance, which provides a set of methods for estimating and quantifying changes in the total amount of water derived from precipitation and how it is distributed and stored in various components of the watershed (i.e., soils, vegetation, groundwater, runoff and streams). Estimation of the water balance is then presented as a basic framework for investigating ecosystem processes that underpin specific services and for estimating their magnitude and direction. This section therefore includes an overview of basic principles of hydrology, key indicators, and data needed to accomplish this task. It concludes with a discussion regarding the use of models for understanding watershed processes, and for anticipating expected or potential hydrological responses to proposed management strategies. It also presents factors to be considered in the selection of appropriate modeling approaches, as well as in the calibration and verification of results.

Information about watershed processes and the magnitude and direction of changes, comprises a basis for estimating their economic significance to various kinds of users, absent which they cannot properly be considered "services." In other words, watershed "services"refers only to the benefits that watersheds provide for humans, which includes various forms of direct and indirect economic benefits, provided also that users actually have access to them. The subsequent section discusses the use of this information to identify conflicting uses, to evaluate trade-offs and the distribution of costs and benefits associated with management strategies, and also for monitoring whether objectives are being achieved, so as to be able to appropriately modify actions or objectives in response to feedback.

It needs to be emphasized that the value of watershed ecosystem services depends on the development of effective management strategies, which is as much an issue of governance as it is of the biophysical aspects of service provision. This requires an understanding also of the extent to which services have characteristics of public goods and of common pool resources, both of which have implications for transaction costs. These characteristics refer to the degree to which access to particular services can be made exclusive to those who pay for them, and to which there is rivalry over access. In a situation of rivalry, demand for a service exceeds supply. Therefore, any use of the service will reduce the amount available to others (Ostrom, Gardner et al. 1994). This section provides an overview of institutional and policy arrangements needed to insure access to benefits by those who are entitled to them, with a particular emphasis on institutional challenges faced in the context of Latin America, where there are several existing and planned initiatives, and where it is expected this report may be of special interest.

The report concludes by identifying some "Rules of Thumb" for the development of effective management strategies that are appropriate to their context. Sources of further and more detailed information are provided throughout the report.

Overview of existing PWES initiatives

A recent IIED review of 287 initiatives to develop markets for ecosystem services identifies 61 cases of PWES, though many of these have only been proposed or are in initial phases and cannot be fully evaluated in terms of actual service delivery (Landell-Mills and Porras 2002). However, based on these and other reviews of literature and case studies (UN FAO 2002) (Johnson, White et al. 2001; Perrot-Maître and Davis 2001) (Tognetti 2001) (Aylward 2002) (Pagiola, Landell-Mills et al. 2002), much can be said about the motivations for the initiatives, how they are structured, and the basis for payments.

Motivation for PWES Initiatives

Some general motivations underlying the growing interest in developing PWES initiatives are:

- An increase in threats or in the perception of threats, leading to an increase in the awareness of beneficiaries of the services provided by watersheds, and in Willingness-To-Pay (WTP) for them;
- To create economic incentives for upstream land users to adhere to conservation practices needed to insure delivery of Watershed Ecosystem Services (WES);
- Inadequacy of regulations alone to insure delivery of WES;
- To develop more cost-effective approaches for achieving regulatory standards;

- To contribute to the reduction of poverty and urban/rural disparities, i.e, to address inequities in the distribution of costs and benefits of providing WES;
- To create a steady flow of funding for the management of upstream protected areas.

Voluntary participation of providers will generally depend on whether economic incentives offered are sufficient to offset opportunity costs.

Some specific factors that have led from inaction to action identified in case studies:

- Directly perceived threats to welfare – e.g.:
    - direct threats to aquifers as a result of imminent development seen in initiatives to protect the Edwards Aquifer in Texas – the sole water source to the cities of Austin and San Antonio (Trust for Public Land 2001), and protection of the New Jersey Pinelands that overlie a large aquifer under sandy soils (New Jersey Pinelands Commission; Collins and Russell 1988);
    - reduction of the water supply in South Africa associated with the spread of alien vegetation in a generally arid region (van Wilgen and Le Maitre 1998);
    - catastrophic events such as the Dust Bowl in the 1930s, which led to the formation of the Soil Conservation Service, a predecessor to the Conservation Reserve Program that provides transfer payments to farmers for conservation purposes, although water quality was not included as a criteria for payments until the '90s (USDA 2000);
- Change in perceptions well documented in public opinion surveys, influenced by scientific knowledge and public education (Macnaghten and Urry 1998), leading to changes in values that are expressed in various forms of WTP for a broader range of assets, including those with indirect or non-use values (e.g., wildlife habitat, water quality, landscape enjoyment and biodiversity). These may be paid for through individual donations to non-profit organizations that may then fund
- Regulatory action as an incentive to find more cost effective strategies, e.g.:
    - The cost of compliance with new regulations that would have required construction of a filtration plant, created an incentive for New York City to invest in upstream conservation as an alternative (Echavarria and Lochman 1999; Perrot-Maître and Davis 2001);
    - Ability to reduce the cost of required emission reductions through tradable permits between point and non-point sources, for which reduction costs are expected to be lower (Faeth 2000). For example, it has been estimated that a trading program under development for Long Island Sound will reduce costs of achieving nitrogen reduction goals by $200 million over a 15 year period (Environomics 1999);
- Special opportunities to act, e.g.,
    - Availability of a bargain - in the case of the Edwards Aquifer in Texas, the failure of the Savings &Loan Associations made large areas of land available at relatively low prices (Trust for Public Land 2001);

- Funding availability through advance sale of credits for wetland restoration combined with the availability of large suitable areas for doing so (Liebesman and Plott 1998; Ohio Wetlands Foundation 2001);
- Opportunity to further both environmental and short-term economic objectives - in South Africa, removal of alien vegetation to increase the water supply was made politically feasible by designing the program to also meet the key social objective of poverty alleviation, carried out through a program of training and employment of the poor (van Wilgen, Richardson et al. 2001);
- Sectoral policy reforms, e.g., to replace subsidies provided under the EU Common Agricultural Policy that are being phased out, that encourage overproduction and environmental degradation, with incentives for environmental stewardship (Hampicke and Roth 2000) (UK 2002);

How are PWES Initiatives organized?

Mechanisms used to provide financial incentives for the provision of ecosystem services are essentially an institutional arrangement among stakeholders, that determines how services are paid for – i.e., who is paid to take particular actions, who pays for it, and how it is implemented. Some may involve intermediary organizations that play various roles, such as product certification, formation of associations needed to reduce transactions costs and make it possible for negotiations to occur and for agreements to be made among numerous individual stakeholders. These often involve use of a combination of various kinds of individual instruments, key types of which have been:

- Voluntary Contractual Arrangements (VCA) - these typically involve the negotiation and agreement of a contract in which resource users, who benefits from watershed services, pay upstream landowners for adoption of management actions needed to insure their provision. Intermediary organizations such as User Associations may be involved as a way to reduce transaction costs when there are numerous stakeholders involved whose cooperation is necessary. An important consideration is the relative power of stakeholders at the bargaining table, what information is available to them, and whether any significantly affected stakeholders have been excluded. One example is an agreement between the La Esperanza Hydropower Company and the Monteverde Conservation League in Costa Rica, which owns the forested area upstream from the plant (Rojas and Aylward 2002).
- Transfer Payments (TP) – these are payments made directly to landowners to create an economic incentive to adopt specified management practices, in recognition of the value of ecosystem services, rather than as subsidies. These are generally in the form of contracts, but may also consist of compensation for the costs of mandatory actions. Funds may be derived from various sources, including user fees, taxes, and donations which are normally channeled through governments or NGOs, who play various roles, such as in the establishment of conservation priority areas where services will be paid for, as well as in

contracting with landowners. Prices may be fixed or established through bidding systems.

- Marketable Permits (MP) – once allowable levels of pollution and resource use have been established through policy, marketable permit systems provide a way for these to be allocated by the market rather than by government. They may be in the form of a cap-and-trade or credit programs. The first requires establishing an aggregate limit on resource use or pollutant emissions. The second instead offers credits for reducing emissions or resource use beyond legal requirements. Key issues include the initial method for allocating rights, and rules for transferability. For example, limits on transferability may be used to prevent concentration of rights in the hands of a few, or to maintain rights within a particular community, but may also reduce efficiency by reducing the pool of buyers and sellers. Methods for allocating rights include:
    - Random access (lotteries)
    - First-come, first served (also grandfathering or historic use)
    - Administrative rules based upon eligibility criteria
    - Auctions

  Grandfathering is often favored because it is less politically difficult than approaches that change existing rights, and protects existing investments. A second issue associated with allocation of rights is to structure them so as provide enough security to promote investment while insuring also that they can be changed in response to environmental variability and new information (Tietenberg 2002). They tend to be most effective for controlling point sources of pollution but several initiatives are underway in the US to allow trading between point and non-point sources for improving water quality (Environomics 1999). A program of trading in salinity credits is being implemented in the Macquarie catchment of New South Wales, to reduce water tables that bring salinity to the surface and threaten irrigation land, by providing credits to those who plant native vegetation upstream, that are purchased by downstream irrigators who benefit from reduced salinity (Perrot-Maître and Davis 2001).

- Tradeable Development Rights (TDR) involve the separation of development rights from other kinds of rights associated with a parcel of land, such as right of occupancy, rights to water, or rights to mine subsurface minerals or timber. These rights may then be sold separately from the land, and allow the purchaser to build in areas designated for development. Important considerations are legal, monitoring, and enforcement capacity. The more successful initiatives are generally part of a comprehensive regional plan that justifies the designation of conservation and development areas. Conservation easements are the principal example of this approach with respect to forestland and often are used as a way of keeping land in its natural state. This instrument is found in the Pinelands Development Credit Program in New Jersey (New Jersey Pinelands Commission; Collins and Russell 1988), in a Wetlands Mitigation Banking initiative of the Ohio Wetlands Foundation (Ohio Wetlands Foundation 2001) and elsewhere, to prevent development of farmland, and to reduce forest fragmentation.

- Certification and labeling – in which products certified to have been produced consistent with specified management practices may gain a market advantage among environmentally conscious consumers. It requires a trusted organization to serve as a certifying entity, and public education. The main example of the use of this approach explicitly for purposes of watershed protection is the "Salmon Safe"initiative in the Pacific Northwest .

Sources of funds:

- User fees – work best when it is possible to limit benefits to those who pay. The can however be differentiated by types and amounts of use as appropriate for meeting social objectives, such as alleviating poverty, and recognition of the "right to water" as a fundamental human right. Examples include fees added to existing charges for water delivery that are specifically designated for financing conservation activities, and licensing of activities that reduce streamflow.
- Taxes – these may be necessary when benefits cannot be limited to a specific group of beneficiaries as a way to overcome free-riding, or for policy reasons, it is considered fair that payments be made mandatory and responsibility be more widely shared; examples might include reduction of flood damages and protection of biodiversity and indirect uses associated with it. Given the large size of upper watershed areas relative to lower, these tend to be the main source of transfer payments to farmers for conservation practices. In Colombia, watershed management is funded through a 6% tax on the revenue of large hydroelectric plants, of which 3% is transferred to autonomous regional corporations who have authority for catchment management, and 3% to municipal governments, partly for purposes of basin protection and sanitation projects. In addition, 1% of funds invested by towns in water projects must be invested in watershed protection (Perrot-Maître and Davis 2001) (Becerra and Ponce De León 1999).
- Donations – may be more appropriate for more globalized benefits, such as protection of biodiversity, or to address root causes of problems that are beyond the control of local stakeholders, such as those associated with hydropower infrastructure and commercial logging concessions driven by national and international level interests. External sources of funding will be harder to sustain and may be more appropriate for sponsoring activities needed to develop an initiative, such as the conduct of independent assessments, rather than for operations and maintenance.

PWES are often also complementary to other objectives, (e.g., carbon storage, protection of landscape beauty and wildlife habitats) and may be part of a package of services that can attract additional funding sources.

In general, mechanisms for creating economic incentives consist of various kinds of arrangements between buyers, sellers, and often, intermediary organizations established to reduce transaction costs associated with agreements among numerous parties. These range from informal, community-based initiatives to more formal contracts among private parties, and various combinations of market-based, regulatory and policy incentives required at larger scales, when threats are beyond the control of affected communities. In general, benefits will be more tangible, and contractual arrangements more feasible, at smaller scales, and where property rights and stakeholders can be better defined. At larger scales, where it is harder to link causes and effects, and rights and responsibilities are harder to define because of public good or common pool characteristics of the resource, there will be a greater need for the involvement of government or other intermediaries to facilitate transactions among more numerous and diverse stakeholders and to establish priorities. Because of environmental heterogeneity, agreements at larger scales will also need to be more standardized and require more regulatory infrastructure but have the advantage of a larger pool of buyers and sellers (Rose 2002). Actions at either scale are likely to be more effective when they are associated with a comprehensive basin-wide management plan that can be used to justify specific actions, by demonstrating their relative value for achieving objectives, and that has been developed in collaboration with stakeholders.

What are the watershed services being paid for?

Services of interest and paid for in case studies consist of management practices needed to maintain watershed processes associated with:

- Total water yield
- Maintenance of dry season flows;
- Attenuation of peak runoff of storm flow;
- Protection of water quality – through reduction of inputs of nutrients and salinity levels, and allowing normal rates of sediment flow;
- To protect biodiversity
- To protect wildlife habitat.

A key question, seldom adequately answered, is whether or not the services are actually being provided, and what is required to insure future provision. This depends both on relevant ecosystem processes, on the extent to which their impacts and benefits have economic significance, and on whether and how these are linked to management practices.

Given the complexity and range of variation of multiple inter-dependent and site-specific causal factors that ultimately determine biophysical outcomes, including those associated with human decisions, and the multiple interests and rationalities of stakeholders, complete information can never be obtained and uncertainty is inherent, even when studies are comprehensive. For example, in the Rio Chiquito watershed of Lake Arenal in Costa Rica, where a comprehensive assessment was conducted, it was possible to establish very general land water relationships. However, unexplainable data inconsistencies remained, for which possible reasons could include quality of rainfall data given a less than ideal number of monitoring stations, reliance of models on annual average figures and values reported in the literature, simplifying assumptions used in models, and landscape heterogeneity (Aylward and Tognetti 2002).

Aggregate analysis can also obscure important processes as well as opportunities. Also in the Arenal study, the results of aggregate analysis suggested that, neither the market by itself, nor payments offered by the government for reforestation, provided an incentive to reforest steep slopes used for cattle ranching and agriculture in the Rio Chiquito catchment area. Ranching was found to produce higher net present values than reforestation and thus to be more economically efficient. This was because, in addition to generating greater returns for the landholders than could be obtained from incentives offered for reforestation, the increased water yield that resulted from deforestation outweighed the costs of sedimentation, because that yield was of direct benefit to a downstream hydroelectric facility. However, these costs and benefits are not all distributed equally. A subsequent companion study that examined the costs and benefits from the perspective of major stakeholders, and which made distinctions among various kinds of landholders, found that the higher returns per hectare depend in part on location in the catchment, that they accrue primarily to large landholders. However, incentives offered for conservation may still appear attractive to small landholders, who, in this case, had lower opportunity costs and, not coincidentally, often occupy the steeper slopes where conservation measures could be expected to provide disproportionately greater benefits (Aylward and Tognetti 2002).

This kind of variation in response is also illustrated in Central America, where Bravo-Ureta et al (2003) compared three IDB financed watershed management projects in Honduras (El Cajon), Guatemala (Chixoy) and El Salvador (PAES). One of the main objectives of the study was to identify the determinant factors leading to the adoption of new conservation technologies by the beneficiaries, all of which were low-income small farmers mostly occupying steep slopes. The project in El Salvador comprised three distinct sub-projects, as they were executed by different contractors using different approaches (PAES 1, PAES 2 and PAES 3). A number of beneficiaries were interviewed in the three areas: 210 in El Cajon, 647 in Chixoy and 530 in PAES (175 in PAES 1, 177 in PAES 2 and 178 in PAES 3).

In general, they concluded that the probability of adopting new technologies tended to be higher as the level of education of the farmers rose, they spent less time in out of farm work, proportionally they controlled more land, and had a tendency to intensive cultivation. They were also more aware of erosion as a problem and had a tendency to participate in community organizations. It is also interesting to note that the adoption rates were higher for the PAES projects in El Salvador. The other programs were based mainly on technical assistance mechanisms, while in El Salvador a more systematic market support and more elaborated incentive mechanisms were used. These mechanisms gave each farmer a subsidy for one or two years, according to the individual plan elaborated by the farmer with support from the contractor, and the farmer had to return a percentage (usually 80%) to his or her producer organization. The study concluded in principle, that the incentives were a key factor in the higher adoption rates for El Salvador, although this question was not explicitly included in the interviews. No conclusions could be obtained as to sustainability, as both the Chixoy and El Cajon projects were just completed and PAES was in the final stages of execution.

Land and water myths

In the absence of actual assessments of land and water relationships, PWES initiatives are often based on myths, or presumed relationships between land use changes and hydrological ones, which can be grouped into 3 general categories:

- Inappropriate generalizations from one site to another, and in particular, application of knowledge from temperate to tropical zones.
- Forests and water myths – e.g., that forests significantly reduce or prevent flooding and increase dry season flows. Whether or not this occurs depends on numerous site-specific factors that determine whether levels of reduced infiltration as a result of compacted soil is exceeded by levels of evapotranspiration and therefore, the quantity of water that is available to stream flow. For example, soil that has been compacted as a result of previous management activities, the presence of roads, and other construction associated with development, can disproportionately affect drainage patterns. Another example is that forests may significantly reduce flooding in the immediate vicinity but have an insignificant impact beyond a certain distance downstream, where runoff is received at different rates from many different sources in the upper watershed and is averaged out at this larger scale, to the point at which it is scarcely if at all detectable.
- Erosion myths – that land use practices in limited areas upstream can have a significant impact on downstream areas, particularly in arid areas with naturally high rates of erosion. For example, modifying land use practices in areas where erosion is naturally high will not prevent sedimentation of dams. Presence or absence of groundcover, roads, and construction activities may also affect levels of erosion and sedimentation, regardless of whether trees are planted.

Bruinjzeel (1990) comments that "adverse environmental conditions so often observed following deforestation in the humid tropics are not so much the result of deforestation per se but rather of poor land use practices after clearing the forest." The former usually has the equivalent of leaving the deforested area as fallow, whereas the latter results in rain fed agricultural production or animal husbandry, incorporating slash and burn techniques, and hastily developed roads.

Even when these links can be established, a review of the literature (Aylward 2002) raises questions about whether the magnitude of damages is economically significant when considering just the relationship between land use and hydrology - though they may be significant when considered as one of many goods and services provided by a basin, and of multiple management objectives. Another possibility is that impacts are significant at a smaller scale than that of the investigation, to a more limited group of stakeholders.

Although myths may be unavoidable given the difficulties of linking management activities to long-term and offsite impacts, the drawback is that allocation of resources is then guided by political expediency rather than by where they can be most effective. In Central America for example, watershed management concerns go back to the early part of the last century but did not get placed at the top of political agendas until they were seen as threats to higher priority interests downstream: the sedimentation of hydroelectric dams which could threaten urban energy supplies – examples of which are the El Cajon hydroelectric dam in Honduras, and the Lempa river in El Salvador; the operation of the Panama Canal, and reduction of vulnerability to disasters following Hurricane Mitch (Kaimowitz 2001). However, impacts on this infrastructure were subsequently found to be either negligible or difficult to quantify (Basterrechea, Dourojeanni et al. 1996) (Vaughan and Ardila 1993).

What was the basis for valuation and decision-making?

Faced with uncertainty about costs and benefits, establishing the actual value of protecting the services provided by watersheds becomes in part a value judgment and a matter of policy, as are most decisions of a public nature. Trade-offs tend to be evaluated and prices paid based on the opportunity costs of forgone land uses, costs of implementing management plans, costs of alternative courses of action (e.g., regulatory costs avoided), and reduction of threats and uncertainty associated with proposed changes in land use.

For example, in the New York City case, the value of taking action – to invest in upstream conservation measures and upgrading of infrastructure – is simply based on the cost of meeting the regulatory standard, rather than on whether the standard is economically appropriate. Valuation of the actual changes expected as a result of the standard itself would require estimating impacts of various combinations of the many discrete options associated with the program, and which would be "combinatorially daunting" even if the data existed (Simpson 2000). However, the regulatory standard is presumably based on scientific criteria that guide the decisions of the regulatory agency. In some cases, particularly when human health is involved, it is a matter of policy that standards be established without regard for cost, though the cost of compliance often triggers public debate and may lead to reconsideration of the standard. Decisions about trading programs are also based on the potential for avoiding regulatory costs.

Other justifications for cost decisions found in selected case studies include:

- Costs of implementing management plans – examples are Quito Ecuador (Echavarria 2002), in which water use fees for conservation are used to cover operation and maintenance costs for upstream protected areas, and the Cauca Valley in Colombia in which farmers, through their water user associations, finance the implementation of existing management plans (Echavarría 2002);
- Individual Willingness-To-Pay in the form of user fees and purchase of products certified as being produced consistent with specified conservation management practices (e.g., the Salmon Safe initiative);
- Political Willingness-To-Pay as indicated by national budget allocations of tax revenue (Günter, Schläpfer et al. 2000), examples of which are Direct Payments to farmers in the US and Europe;
- Landowners Willingness-To-Accept compensation or cost of supplying the service, sometimes determined through bidding processes, examples of which are payments to farmers in the US Conservation Reserve Program, and payments to owners of forest land in Costa Rica, through the FONAFIFO program (Chomitz, Brenes et al. 1998);

- Comprehensive management plans – it is easier to justify decisions regarding permitted uses of individual land parcels when these are consistent with an agreed upon comprehensive plan that provides a strong rationale, an without which they may be seen as arbitrary, an example of which is seen in the tradable development program for the New Jersey Pinelands (New Jersey Pinelands Commission).

What was needed to create an enabling environment?

As discussed in the introduction, the value of watershed ecosystem services depends on stakeholder confidence in access to benefits, without which they cannot properly be considered "services". Some studies, for example, have found a higher WTP, even for less tangible values such as protection of wetlands along an international bird migration route, under scenarios in which all of the relevant stakeholders participate, in this case, all countries along the migration route (Koundouri, P. et al. 2003). Another study reported that in Brazil, which adopted a nationwide river basin management policy, domestic water users were found to be willing to pay more for water when the revenue from water fees is invested in the basin where the funds are generated, and when users are able to participate in decisions as to how the revenue is spent (Porto, Porto et al. 1999). Other studies have found differences in WTP that depended on the protection mechanism suggested, and whether it was regarded as fair and effective (O'Connor 2000).

Access to benefits is primarily determined by various forms of property rights which define rights to particular streams of benefits as well as responsibilities for their provision. Thus they determine who has access to particular resources, and whether those who pay the costs of management practices have access to any of the benefits, and therefore have an incentive for conservation. In the absence of tenure security, land users may lack the authority to enter into binding agreements. In some cases, upstream land users may be blamed unfairly for impacts to which their contribution may be insignificant, and displaced without any form of compensation, as values rise.

Inherent in all forms of payment arrangements is the recognition or establishment of appropriate forms of property rights or tenure security without which they could not exist. Many approaches involve the definition of rights in innovative ways, such as the establishment of credits that can be traded, or the more specific definition of rights and responsibilities in contractual agreements for watershed management (Landell-Mills and Porras 2002).

Payment arrangements also generally rely on the existence or the development of other appropriate supporting institutions, which refer to relationships established among buyers, sellers, and intermediary organizations that serve to insure enforcement and reduce transaction costs. A second important institutional consideration is that, given the size of upper watershed areas and the need for coordinated action, formation of organizations such as farmer or landowners associations, watershed councils, and land trusts, who are able to develop priorities and plans of action on which stakeholders can agree to collaborate, generally increase the capacity not only for action but also to attract funding, because it increases the confidence of buyers that actions will be effectively implemented. Intermediary organizations may also play important roles in gaining recognition of property rights as well as in technical assistance and marketing.

Activities associated with the establishment of an enabling environment also include:

- Establishment of policy objectives
- Research and assessment to:
    - Define and quantify services
    - Identify effective management actions
    - Identify distribution of costs and benefits
    - Raise awareness and WTP of stakeholders
- Reduce barriers to participation
- Monitoring and enforcement

Activities necessary to establish an enabling environment are all sources of transaction costs, which can potentially be a constraint on the development of PWES schemes, particularly when there are numerous stakeholders. According to the IIED review, there is little actual research on transaction costs, and it is not always clear that PWES schemes are more cost-effective than other courses of action. However, in evaluating transaction costs, it should be kept in mind that over time, history suggests that they may be reduced through improvements in technology and institutional arrangements (North and Thomas 1973). Also, that institutional capacity building can have broader social benefits and are intertwined with the ongoing development of democratic institutions that may have even greater value than individual PWES initiatives, particularly when they are designed to begin to overcome barriers to participation of the poor…. (Landell-Mills and Porras 2002)

Effectiveness of PWES initiatives

Given that most PWES initiatives are relatively recent, it is difficult and perhaps premature for them to be fully evaluated to verify provision of services. However, there has also been little investigation with respect to other aspects of effectiveness and there is little or no data collection regarding actual transaction costs (Landell-Mills and Porras 2002). For example, no conclusions can be drawn regarding cost effectiveness compared with regulations, which has been among the key motives for their development. However, transaction costs associated with PWES initiatives may also have positive spinoff benefits that have not been taken into consideration (Landell-Mills and Porras 2002).Whether or not they contribute towards poverty reduction likely depends on whether this objective is effectively considered in the design of the program.

Experience to date suggests that the key challenges are to define the services, identify the actions needed to insure their delivery, and create the proper incentives to take them – all of which is complementary to identifying able buyers and increasing their WTP, which has received greater attention.

PWES should be regarded as a long-term process of institutional development – i.e., an issue of governance, in which assessment is critical.

Defining and Quantifying Watershed Services

Watershed services are products of ecosystem functions or processes that provide direct and indirect streams of benefits to humans, and may include:

Provision of freshwater for:

- consumptive uses (drinking, domestic, agricultural and industrial),
- non-consumptive uses (hydropower generation, cooling water and navigation),

Regulation:

- Flow regulation and filtration – i.e., maintain water quality, water storage in soils wetlands and floodplains which can buffer flood flows and drought, erosion/sedimentation control, control of the level of water tables that bring salinity to the surface, maintenance of wetlands, riparian habitats, fisheries, and other wildlife habitat for hunting and for migratory birds, rice cultivation areas, and fertilization of floodplains. Natural flow regimes are also important elements in the development of mangroves and in maintenance of estuarine and coastal zone processes, which are critical habitats for fisheries as well as for other marine life. Transport of normal sediment loads also protects coastal areas from erosion that occurs when sediment is retained behind dams and which can reduce storm damage.

Cultural services

- recreational and tourism uses
- existence values)

Supporting services:

- Insurance against uncertain effects of a change in conditions by maintaining natural flow and disturbance regimes, i.e., support for ecosystem resilience for which thresholds are generally uncertain.

Whether or not these services are provided therefore depends on the function of linked ecosystem processes, for which the water balance provides a framework for investigation. Potential impacts are categorized in Table 1, and ultimately have impacts on ecosystem and living aquatic resources.

1. Impacts of land use on the hydrological and sediment-related processes:

    a. Mean surface runoff
    b. Peak flow/floods
    c. Base flow/dry season flow
    d. Groundwater recharge
    e. Soil moisture recharge
    f. Erosion and sediment load

2. Impacts of land use on water quality

    g. Nutrients and organic matter
    h. Pathogens
    i. Pesticides and other persistent organic pollutants
    j. Salinity
    k. Heavy metals
    l. Changes in thermal regime

Table 1: Categories of land use impacts on aquatic ecosystems and processes

Source: (UN FAO 2002)

The Water Balance as a Framework to establish a PWES scheme

Having correct expectations of impacts of landscape management practices on the water balance is a point of departure forestablishing tradable environmental services. As a result, planners require knowledge on the various hydrologic components of the water balance and on the relevant linkages of these components to the landscape. Combined with an accounting of water needs and uses, it is possible to assess ecosystem functions that support the production of freshwater services, and quantify their flows in catchments. The value of the watershed service is dependent on both the supply and demand of the resource, which implies that it is also necessary to account for water use in a basin. Water use efficiency (Mei, Küffner et al. 1993) or catchment efficiency, (Molden 1997) may serve as an index of willingness to pay and also provide insight on effectiveness of management actions. This is because, as a general rule, when water use efficiencies are high, water is limiting and WTP for improvements are high. When water is scarce and yet efficiencies are low, it can also provide clues of resource mismanagement. In other words, understanding of the supply and demand of hydrologic services can help to determine stakeholder willingness to pay for services associated with its provision, and in the identification of priority areas for implementation of conservation practices.

However, given that watersheds are natural systems, which respond to variable climatic input and that are impacted by humans, they are inherently uncertain systems. Types of uncertainties in the natural system include variation in inter- and intra-annual climatic factors and in human land use practices, all of which affect relationships between land and water that are poorly understood to begin with. This poses a challenge to the establishment of markets for environmental services, which require adequate knowledge of links between land use management practices and hydrological outcomes (downstream) as well as on the distribution of costs and benefits among stakeholders. Markets require certain expectations regarding the delivery of services. Since the ability to collect hard data on variation over a short period is unlikely, observations must be integrated with current understanding of general hydrologic principles, paired catchments, and can even be supplemented with "soft" data based on local knowledge. Ongoing monitoring programs are often needed to improve system understanding over time, which may lead to the modification of watershed management strategies and perhaps also, the terms of payment schemes for environmental services.

Variability in data is key to understanding the underlying hydrological processes of a catchment (Yew, Dlamini et al. 1997). A dry year worth of data followed by a wet year is generally more insightful than monitoring consecutive years of similar climate regime. However, planning to monitor around such transitional regimes such as *El Niño Southern Oscillation (ENSO)* climatic cycle is problematic. For example El Niño events, as well as the converse La Niña events, had already devastating effects on the countries of Latin America and the Caribbean (LAC). The economic value of global losses due to floods, drought, forest fires, destruction of coral reefs, changes in the fishing industry, etc, associated with the El Niño event of 1982/83 was estimated to be 13 billion US dollars. The 1997-98 El Niño costs in LAC countries were even larger. Early warning could have reduced these losses considerably (WMO 2003).

That prompted seventeen countries in Latin America, to undertake a study on mechanisms to ameliorate the negative impacts of El Niño, with technical support from the World Meteorological Organization (WMO) and participation from the International Food Policy Research Institute (IFPRI), the International Research Institute for Climate Prediction (IRI), and the National Ocean and Atmosphere Administration (NOAA), through an IDB financed technical cooperation. The general objective of the study was to design and determine the feasibility of a project or projects aimed at establishing a regional system or systems that produce and utilize early warning of impending danger and related social and economic consequences, based on the actual predictions of ENSO, in order to ameliorate the socioeconomic impacts of the phenomenon.,. The implementation of this projects has not started yet and clearly, as there is no guarantee a good quality data set exists or can be achieved. Given that relations are known to exist between the *El Niño* global climate cycles and local climate cycles which can be determined from local knowledge,,conceptual understanding can be improved by integrating "hard data" with findings in catchments that have similar geo-morphologic and climatic characteristics, and expert knowledge in the form of "soft data".

"Soft data" are available from local knowledge and professional experience in a region, and has been shown to improve conceptual understanding of hydrology when "hard data" is limited (Seibert and McDonnell 2002) (Oba 2001) (Sinclair and Walker 1999). Archaeological evidence of hydraulic structures is often used to attest to the successful application of local knowledge. More recently, (Aboites 1998) describes interviews of the

Yaqui people in the design of one the largest irrigation schemes in Northern Mexico during the early 20th century, whereby magnitude and frequency of flood peak events were documented by engineers. However, "soft data" should be used with care. First, because people with local knowledge are rational beings that take into account their personal needs and the benefits of any existing rural outreach/development programs. For example, the possibility of a program to pay people to plant trees might likely invariantly lead to stakeholders noting the benefits of forests. In effect, several watersheds throughout Latin America have reforestation programs that promote reforestation for one reason or another. Second, because interviewers and interviewees may have their biases as a result of the "successful" promotion of myths from poorly thought-out outreach programs. However, valid or not, farmer understanding of relationships between land use practices and outcomes can help to understand the basis for their land use decisions. For example, in a case in Northern Thailand, forests above rice paddies are owned as a unit because they are regarded as the source of flows of cool water into rice paddies, which is important for sustaining crabs, frogs and fish that inhabit the paddies (van Noordwijk, Poulsen et al. in press).

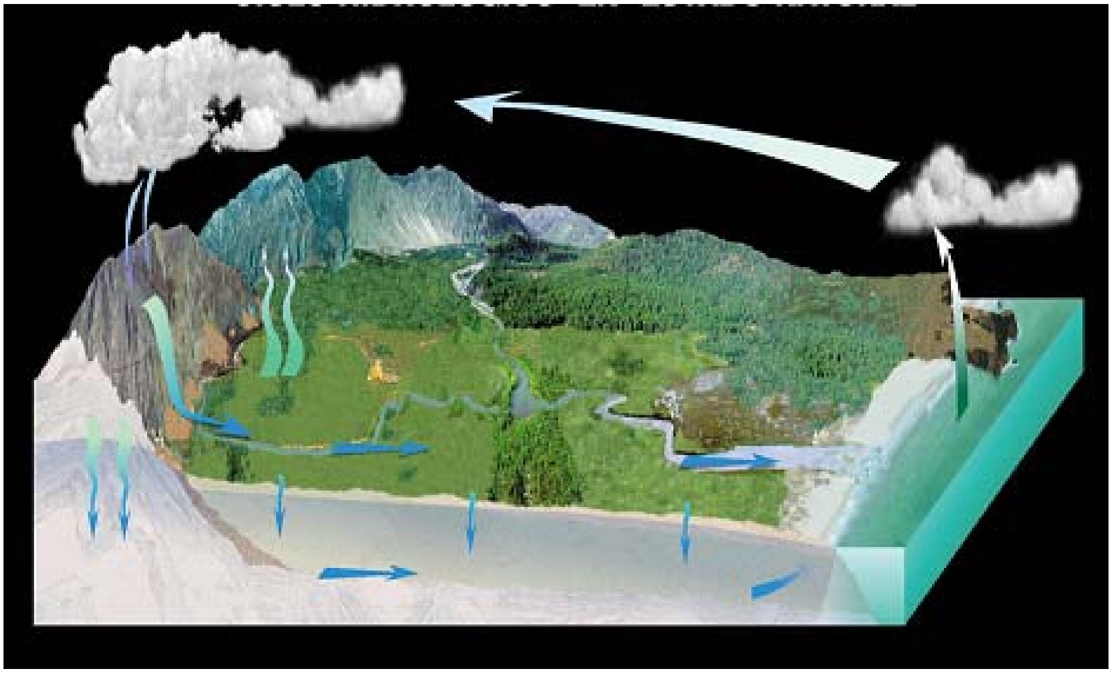
Therefore, "soft data" should, at a minimum, be critically reviewed by experts and verified through monitoring programs. In addition, non-biased and loosely structured interviews to obtain "soft data" should be developed by water resource or hydrology professionals that are familiar with the site of interest in conjunction with people experienced in developing such questionnaires or facilitating focus groups. A process of triangulation, which involves examination of consistencies and inconsistencies among multiple sources of quantitative and/or qualitative data, may also lead to new perspectives and overlooked questions for research (Creswell 1994).

The objective of this section is to provide an overview of procedures for water accounting and for obtaining a gross estimate of water balances as a basis for analysis of basin productivity (Molden 1997), and analysis of trade-offs associated with achieving various management objectives. In general, the water balance describes the allocation of water that results from the biophysical characteristics of a basin, which is altered through human uses. Water accounting classifies the uses and productivity of those components of the water balance allocated towards human sustenance. For purposes of establishing PWES, the challenge is generally to accomplish this accounting task with limited data and resources for detailed research, yet sufficient to provide justification for payments.

Water Accounting

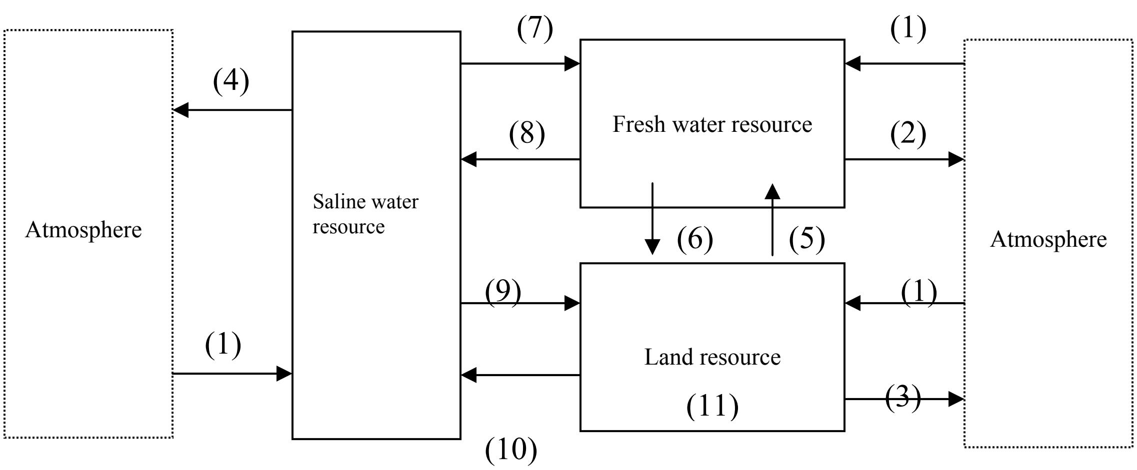
Water accounting is a procedure for analyzing the uses, depletion, and productivity of water in a water basin context (Molden 1997). It would be rare nowadays to find a watershed or river basin where human activity is not part of the hydrological cycle and where the "natural" water balance is not modified by human activity. Water accounting provides insight on the components of a basin's water balance directly impacted or altered by human uses, both in quantity as well as in quality.

Figure 1: The Natural Hydrological Cycle



SOURCE: Geological and Mining Institute of Spain, Ministry of Science and Technology

Figure 2: Quality Interchanges in a Natural Hydrological Cycle



SOURCE: Department of Civil Engineering, Division of Hydraulics and Sanitary Engineering, University of California, Berkeley

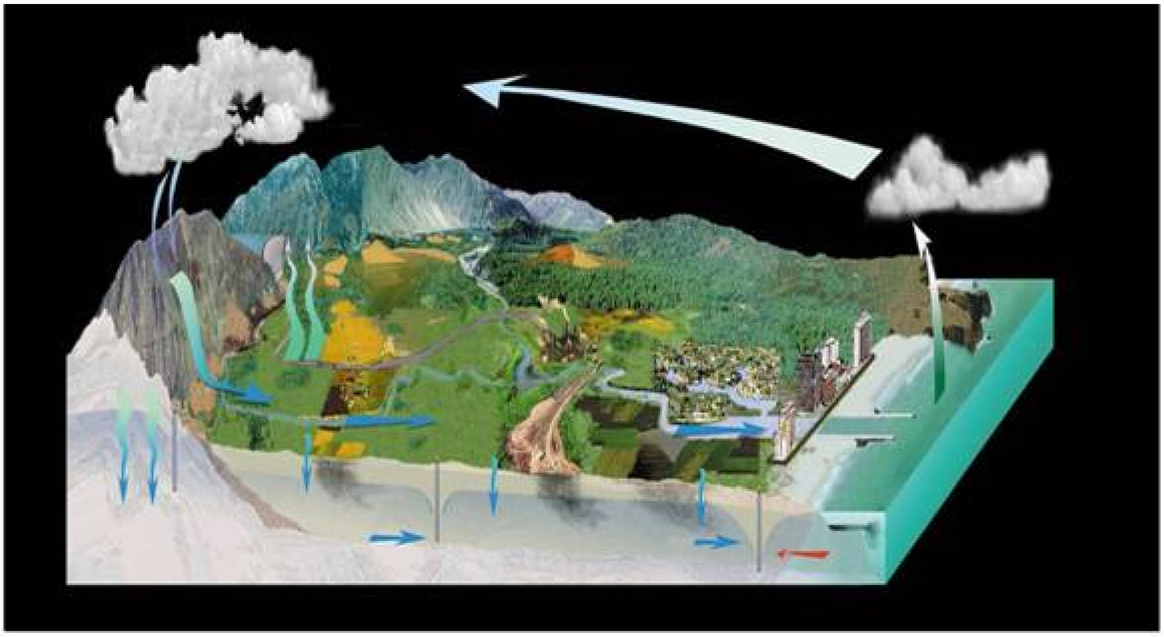
In a natural system, (as illustrated in Figure 1 and Figure 2), meteorological water goes from the atmosphere to the fresh and saline water resource and to the land resource carrying dissolved gases, dust particles, smoke particles, bacteria, salt nuclides and dissolved solids (1). Water returns to the atmosphere by evaporation from the fresh water resource carrying water vapor and salt nuclides (2), from the land resource by evapotranspiration carrying water vapor, vapors from vegetation, dust and organic particles (3), and by evaporation from the saline water resource carrying water vapor and salt nuclides (4). Silt, organic debris, soluble and particulate products of biodegradation of organic matter, silica, mineral residues of earth materials, bacteria, dissolved gases and soil materials are carried from the land resource to the surface water resource by surface runoff (5). Likewise, dissolved minerals from surface debris and primary rocks and dissolved gases are carried from the land resource to groundwater by infiltration (5). Flood waters and springs return silt and other materials and minerals from the surface and groundwater resource to the land resource (6). The interchange between tidal water and continental saline water and freshwater increases the salinity of the latter (7). At the same time, this interchange carries (1) and (5) plus organic debris from the freshwater resource to the saline water resource through river and groundwater discharge (8). Saline water intrusion increases salinity of the land resource (9) and beach erosion carries soil and vegetation from the land resource to the saline water resource (10). Finally, biochemically unstable matter from life processes of animals and from death of plants and animals are accumulated as solid residues on the land resource (11) and may again find their way to the water resources, both fresh and saline.

Water in its natural state is seldom adequate for direct human uses. It is often distant to where it is going to be used, as the location of population centers, irrigation fields, industries, etc., respond to other socioeconomic variables. The quality of the resource often differs from that required for a particular use and usually, the natural hydrologic regime has large variations both in quantity as well as quality. Hydraulic works are then built to divert the water from a given river or lake and transport it to where it is going to be used, such as a city, irrigation field or industry. Sometimes the purpose of the diversion is to maintain a given elevation so that that the potential energy of the flow can be transformed into electricity. Then the water is treated to eliminate objectionable components and to reduce concentrations to those that can be tolerated for human consumption or recreational use. Usually, dams are built to form reservoirs where water can be stored in order to regulate the flow and use it in the naturally dry periods or to store flood peaks so they do not cause damages downstream. From a systems analysis point of view, this could be represented as shown in Figure 3: hydraulic works are the "operator" that transforms input location, quality and regime vectors into the desired output vectors at the desired location and with the desired quality and flow regime for specific uses.

A basin's water balance can be altered by hydraulic works that divert water for beneficial use or by artificial reservoirs (dams) that modify the timing or magnitude of releases. The use of dams increases the overall storage capacity of the watershed because often the "natural" storage potential is inadequate for a particular use of benefit to humans. Diversion of water can essentially re-allocate water released by the watershed or "natural" reservoir. Water diverted for irrigation use, may, in part, recharge the watershed through infiltration, or may be lost from the system through evapotranspiration. Irrigation impacts on the water balance can be significant during the dry season, when water for crops is needed the most. But not only the water balance is changed. The water quality of the watershed is modified as well.

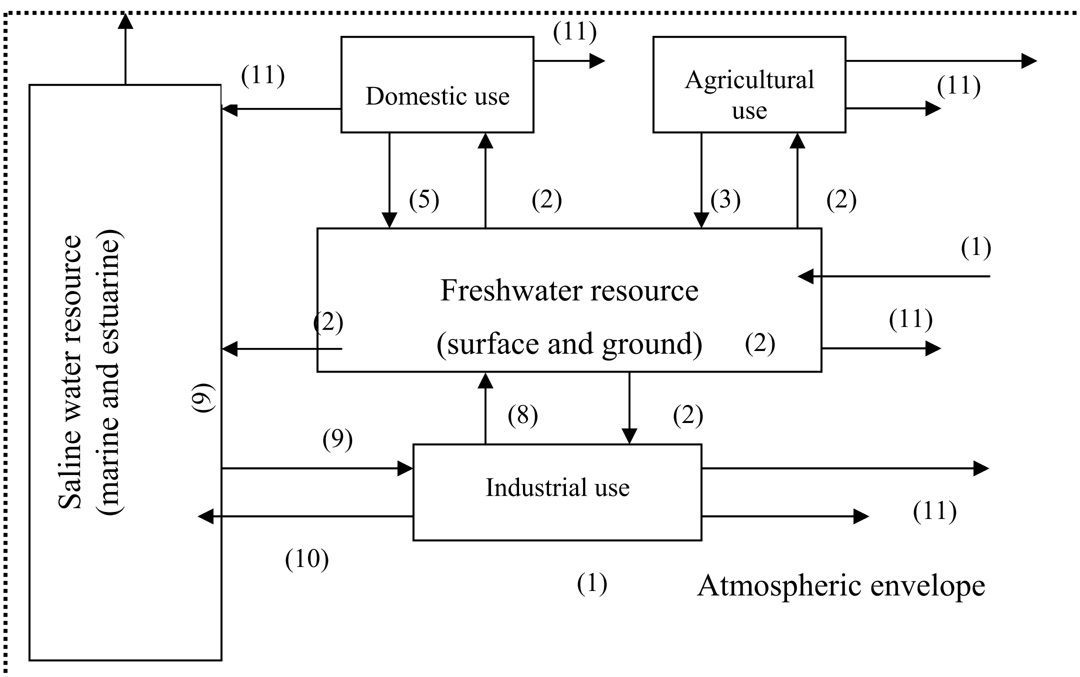
Figure 3 and Figure 4 as well as the discussion that follows pertain to simplified illustrative cases - the real conditions in any given river basin may be much more complex.

Figure 3: Modified Hydrological Cycle



SOURCE: Geological and Mining Institute of Spain, Ministry of Science and Technology

Figure 4: Quality Interchanges in a Modified Hydrological Cycle



SOURCE: Department of Civil Engineering, Division of Hydraulics and Sanitary Engineering, University of California, Berkeley

In a modified system, the quality of the meteorological water (1) is the same as in a natural system, unless severe air pollution modifies it[1]. The quality factors of the freshwater resource, both surface and groundwater (2), are governed by the quality of the inputs from the atmosphere (1), the return flows from agriculture (3), the wastewater from domestic use (5) and the return flows from industry (8) and groundwater, modified by natural systems applicable to rivers, lakes, etc., and artificial systems of return water treatment. This is the same quality that agricultural, domestic and industrial uses (treated as required by individual industries) as well as the saline water resource will receive, possibly after natural self-purification (2). The quality of irrigation return water (3) will be the same as (2) but with added salts, nutrients, pesticides, organic debris and increased salinity by consumptive losses. The quality of domestic return water (5) will be the same as (2), in some cases after treatment, plus degradable organic matter (human body wastes, ground garbage, grease, detergents, etc.), dissolved solids, bacteria, viruses and some industrial wastes (8). The quality of industrial return water (8) will be the same as (2) plus added organic matter, metal ions, chemical residues, etc., higher temperature and increased salt concentrations. The quality of industrial cooling water (9) will be the same as the saline water resource. The industrial cooling water return (10) will be the same as (9) but with added temperature and salt concentration. Consumed water from natural sources as well as from domestic, agricultural and industrial uses (11) will be mostly from evaporation and production and will have the effect of increasing the salt concentrations in the return flows and hence in the receiving waters. Assessment of direct human impact on water is achieved through baseline water accounting, which also provides insight on the real efficiency of water use in a basin, such as is described by (Seckler 1996).

Conflicts may arise because competing uses seek to use the same source of supply, use the same stored volume in a reservoir, or need to release water when other use needs to store it. Conflicts in quality are by no means less important. High irrigation efficiencies (Mei, Küffner et al. 1993) and orderly irrigation rotations (Norman, Walter et al. 2000) generally indicate that water is a scarce and valued commodity with a potential for high WTP for a PWES. High uses of water for irrigation often conflict with other uses of water during the dry season. In the tropical[2] highlands, interest in temperate horticulture crops that are irrigated during the dry season often competes with domestic water supply catchments. Similarly, an accounting for domestic water (in terms of water use efficiency) can enable planners to identify whether strained water supplies result from O&M, competing water uses or infrastructure, or whether the hydrologic health of a catchment is indeed the culprit.

It is crucial to distinguish physical from economic scarcity by accounting for human use of water during the dry season because often limitations are due to competing uses of water, which intensify as population grows and/or there are increases in sectoral uses, which are not necessarily linked to population. In effect, increases in competing water uses may be at fault for water scarcity during the dry season rather than the deforestation.

[1] Cases of acid rainfall are now not uncommon in heavily industrialized areas.

[2] The tropics are herein defined as systems with marked changes in precipitation with limited variability in day length and temperature, at latitudes between the tropics of Cancer and Capricorn. The systems may be arid or humid, at high or low elevations.

Under such circumstances, the problem lies in the institutional, socio-political or infrastructure arrangement, rather than in the supply of the resource. Detailed information on catchment efficiency and water accounting can be found in Mei et al (1993), and Molden (1997), respectively.

Water accounting is needed to construct water budgets (balances between supply and demand) in river basins. These water budgets are one of the principal tools for the water resources planer at the river basin level and thus, are a basic ingredient of integrated water resources management (IWRM). The basic problem to solve in IWRM is to allocate water among competing uses in such a way as to maximize the benefits for the entire system and not for any individual use. Traditionally, water allocation has been done in a discretionary manner, following water-use priorities contained in the water legislation, established arbitrarily according to the perceived importance of a given activity. This practice became inadequate as water demand arose and water scarcity started to appear in some critical areas. Thus, one of the most important Dublin principles is that water is also an economic good and therefore, allocation should be done based on economic instruments to guarantee that the volumes go to the use with the highest value. One of the tools that have begun to be implemented for that purpose is the water rights market, where water rights are traded between users according to their needs. This, however, has not been well received universally, because of strong cultural beliefs about the natural rights to water and the consequent aversion to treat is as a commodity. An alternative has been the allocation of water by consensus by a river basin council, commission or committee (García 1998). The main point is that the shift from development of new supplies to an emphasis on reallocation of existing ones raises issues of underlying rights and policies for prioritizing uses, that justify decisions about water allocation. More information about these issues can be found in the Golbal Water Partnership's Technical Advisory Committee (GWP-TAC) publications such as those by Rogers et al (1998), Solanes and Villarreal (1998) and the TAC (2000).

The Water Balance and the End Points of Interest in a PWES.

Meeting watershed management objectives essentially involves alterations to the water balance, either through direct changes in water allocation between the needs of ecosystems and various competing human uses, or indirectly, through land use changes that alter flows of water and sediment, as well as levels of Actual Evapotranspiration. The water balance provides a tool that can be used to assess the opportunity costs of preserving the natural system. The water balance, or budget of a watershed may be represented by:

$$\Delta S = P - Q - AET - G \quad (1)$$

Where, ΔS is the change in storage, P is precipitation, Q is Streamflow, AET is Actual Evapotranspiration, and G is loss to deep-water aquifer not accounted for by streamflow.

Measurement or approximation of these components, and an understanding of how they interact in a site-specific context provides a basis for defining and quantifying ecosystem services as well as for identifying and clarifying uncertainties. However, given typical data limitations, several simplifications and assumptions must be made. For example, data on Actual Evapotranspiration, storage (especially storage in the unsaturated soil), and losses to deep groundwater are rarely available, especially in regions with limited resources. Classic assumptions are that the change in storage over an average annual cycle is zero, and that losses to deep ground water are negligible. The largest feat becomes thus to estimate basin-wide AET such that seasonality is accounted for, which remains one of the most difficult questions in the science of hydrology (Dias and Kan 1999). This information is key to estimating opportunity costs of changes in the water balance that result from actual or potential changes in land use management associated with particular services, or end points of interest.

This section discusses links between expected changes in the water balance and common points of interest that are regarded as watershed services. . It includes discussion of qualitative aspects, state-of-the-art knowledge, data needs and methodology. End points of interest for a PWES scheme may be generalized as follows:

1) Total Flow Yield: *Key parameters that affect Total Flow Yield are Actual Evapotranspiration and Precipitation. Actual Evapotranspiration is linked to the type of landscape vegetation , and the seasonal fluctuations of the energy and water budget in the system. Precipitation on a basin depends on continental-scale factors but deposition volumes may be impacted at the local scale by interception or condensation of rain or cloud water by vegetation cover. . Although associations between presence of forests and total rainfall at the local scale cannot be entirely ruled out, even if and when these exist, they are likely to be exceeded by increased evaporation. It is also possible that the presence of forests can have impact on regional climatic patterns and therefore on total rainfall at continental scales (Calder 1999). At more local scales, the formation and height of dry season clouds, might have implications for moisture levels available as fog drip by cloud forests (Lawton, Nair et al. 2001) . These impacts however, will not be addressed in that they are considered either insignificant or insufficiently understood for purposes of PWES .*

2) Regulation of Streamflow: *Regulation of streamflow implies managing for dry season flow (baseflow component of streamflow) or Stormflow (runoff component of streamflow). Baseflow and runoff also have linkages to actual evapotranspiration and precipitation but the discussion focuses on particular management implications.*

    a. Dry Season Flow: *This discussion is centered around sustaining stream flows during the dry season, that is, prolonging the baseflow. It is a period when water is scarce and is often the limiting factor for production to small scale irrigators (who have limited hydraulic structures for water storage), and other users. A sustained dry season flow is also generally the key to managing the* *biological health of a stream. Baseflow, which comes from shallow ground water storage, depends on the geomorphology and landuse of the catchment. Therefore, landuse management in aquifer recharge areas is the principal variable, which can be targeted so as to improve the recharge of the shallow aquifer that contributes to streamflow, while minimizing losses to evapotranspiration.*

b. Stormflow: *Stormflow (or storm runoff) produces peaks in the stream that can result in flooding or erode stream banks, and is dependant on rainfall intensity and duration, landuse, and geomorphology of the catchment. The discussion will address ways to determine sources of runoff on the landscape and understand runoff processes, which can be used to target management actions.*

3) Water Quality: *The water quality constituent for stream impairment focused on in this text is sediment, and the discussion will be limited to its relation to runoff and to its source. However, some parallels may exist for other water quality constituents, the transport of which is generally linked with processes of runoff and erosion. Related issues such as, dilution of contaminant loadings (e.g. sustaining baseflow) to reduce adverse impacts on human health are specifically addressed as they are beyond the scope of this report.*

4) Protection of biodiversity and living aquatic resources: *Natural Flow Regimes are critical to the protection of biodiversity and living aquatic resources at the level of ecosystems, which are associated with landscape processes. Components of the flow regime are the natural patterns of variation in the quantity and timing of the flow of a river, including natural disturbances, which are associated with basin climate, geology, topography, soils and vegetation (Poff, Allan et al. 1997). For example, wetlands, riparian habitats, mangroves, and coastal zones, which also support many direct uses, may all rely on regular flood pulses and transport of normal sediment loads for their maintenance. The principals related to regulation of streamflow are largely applicable but focus on the entire catchment and larger scale influences may be necessary.*

5) Recharge of groundwater in Deep Aquifers: *Watershed management to recharge a deep aquifer is for the most part beyond the scope of this text. A few thoughts will be included, but in general, impacts to the shallow aquifer that contribute to baseflow, will be felt to some extent in the deep aquifer. Deep water wells are a source of significant concern (e.g. San Salvador water supply, or Mexico City subsidence) but are largely unregulated in most parts of the world.*

Total Flow

Precipitation, P

At a minimum, establishing links between hydrology and landscape requires information about the amount and seasonal distribution of rainwater that enters the catchment system, and the distribution of streamflow volume. A monitoring plan for rainfall is therefore essential. Fortunately, daily rainfall amounts may be the least expensive component of the waterbalance to monitor. Raingages placed at various points in a watershed can provide adequate information on daily precipitation, which can be reported by people trained to take daily readings. A cross-check of readings provided by a number of people can be used to control the quality of the monitoring program.

Much research on rainfall in the humid and arid tropics points to its spatial and temporal variability. It is generally agreed that temporal variability of rainfall has a greater impact on small catchments (Osborn and Lane 1969) and that spatial variability has greater impact at the larger scales (Michaud and Sorooshian 1992). Chomitz and Kumari (1998) remind us that spatial variability of rainfall at large scales must be considered in evaluating the relationships between large storm events and floods, which are often mitigated as the scale of influence grows. The quality of data on precipitation will therefore depend on the spatial distribution of a network of rainfall gauges as well as the time over which it is collected. The setting up of a rainfall gauge network system might consider the following items:

i) The World Meteorological Organization standard of rainfall gage distribution network is between 100-250 km^2 to 600-900 km^2 per gage, for mountainous and flat topography, respectively. However, these values should be used with caution because they do not consider effects of site-specific factors such as wind, elevation, and slope. Catchments with greater relief will likely have greater rainfall variability and may justify lower values. The number of gages will ultimately depend on available resources and accessibility to different parts of the catchment. In general, even in developed countries, better data are available for the more densely populated downstream areas than for the large and remote upper watershed areas.

ii) Planners should consider a minimum of two gauges, at the headwater and base.

iii) Remote sensing and local knowledge can be used to determine sections in a basin that have significant rainfall differences. Thematic Mapper (TM) satellites obtain spectral signatures from the earth's surface that can be used to obtain vegetation indices, such as the Normalized Difference Vegetation Index (NDVI) or Normalized Difference Wetness Index (NDWI) that provide insight on landscape moisture conditions.

In some humid forests, precipitation must be adjusted to include water that condenses on the vegetation from clouds or fog, and reductions due to canopy interception, which are not reflected in raingauge data. The issue of interception will be discussed further in the

ET section. Given that neither rainfall interception nor cloud impaction can be easily measured, differences between measured rainfall and throughflow/stemflow inputs to the ecosystem must be used to approximate these values (Bruijnzeel 2001). However, it is also extremely difficult to collect representative samples of throughflow (Shellekens, Bruijnzeel et al. 2000) given that it is unevenly concentrated . Bruijnzeel (2001) presents a critical look at findings of typical gains and losses with respect to rainfall that have been attributed to canopy interception and cloud condensation in forests, which are summarized in **Table 2**.

Table 2: Typical gains and losses with respect to rainfall that have been attributed to canopy interception and cloud condensation in forests

Humid Forest Type	Percent Changes in Rainfall due to interception and cloud condensation.
Tropical Lower Montane Forest not affected much by Clouds	-33 to -19% (Bruijnzeel, 2001).
Tropical Lower Montane Cloud Forest	-20% to +1% (Bruijnzeel, 2001).
Upper Montane and Low-Elevation Dwarf Cloud Forests.	-19%to +79%(Bruijnzeel, 2001)
Temperate Cloud Forests	+18% to +40% (Harr, 1982),
Less Dense Forests	Negligible No Change

The findings in **Table 2**, derived from pre-1993 studies, are fairly consistent with findings from eleven post-1993 studies (Bruijnzeel 2001). However, using the post-1993 studies, Bruijnzeel (2001) shows that care must be taken in how tropical forests are defined. He also identifies sources of variability that can be significant but that are generally not accounted for. Among these are that:

- Tropical forests on slopes on the leeward side of moist winds are less likely to capture cloud moisture and will be at the lower ranges of precipitation gain;

- Rainstorms of lower intensity will be at the lower range of gains in precipitation because the intercepting canopy behaves like a reservoir that has limited capacity to collect water and can tip if overfilled. Shellekans et al. (2000) (2000) show unusually high interception ratios (low gain in precipitation) around 50% due to low intensity rains in the maritime forest, El Yunque, in Puerto Rico.

Precipitation gains from cloud condensation (if clouds form during the dry season) can be an important contributor to yield during the dry seasons even when yearly totals suggest these gains are insignificant (Bruijnzeel 2001) (Keppeler 1998). This is also when water availability is more likely to be limiting and willingness to pay for it highest. Therefore, it is important to consider seasonal differences.

Table 3 summarizes results of a case study in the the Rio Chiquito watershed of Lake Arenal, which show the seasonal variability of gains and losses attributed to cloud condensation and interception. The case study shows that most proportional gains are obtained during the dry season when water is needed the most (Aylward and Echeverria 2001). These gains may be particularly important at smaller scales, e.g. for communities or for water supply to small municipalities or hydropower facilities at the fringe of a cloud forest. Larger gains due to fog drip are also found in the literature. Harr (1982) observed increases in water yield gains of up to 500mm resulting from fog drip in a temperate cloud forest.

Table 3: Capture of Horizontal Precipitation in the Cloud Forests of Rio Chiquito (Aylward and Echeverria 2000)

Month	Precipitation in the open (mm)	Precipitation under Forest Cover (mm)							
		Primary Forest Fragment		Secondary Regrowth Fragment		Low Primary Forest		High Primary Forest	
Dry: March 95	68	144	+53%	95	+28%	125	+46%	83	+18%
Wet: August 95	455	345	+32%	425	-7%	346	-32%	364	-25%
Wet: October 95	436	497	+12%	446	+2%	333	-31%	340	-28%
Dry: January 96	216	321	+33%	246	+12%	387	+44%	255	+15%
Yearly Totals	3,301	3,759		3,558		3,496		2,986	
Gain/(Loss) as opposed to pasture		458	+12%	257	+7%	194	+6%	(315)	-11%

Source: Adapted from (Fallas 1996)

In sum, spatial and temporal variability of rainfall, and interception and condensation of moisture by dense vegetation are sources of uncertainty that begin at the onset of determining the water balance. Therefore, rainfall monitoring and making assessments potential sources and magnitude for gain or loss of net precipitation are important tasks. Seasonal variation of moisture in particular, has implications for scarcity and for user willingness-to-pay.

Seasonal ActualEvapotranspiration, AET

Actual Evapotranspiration (AET) is a principal component of the water balance, usually consisting of over half the water that rains in a watershed per year. As a result, reasonable approximations of AET are important to estimating impacts of landuse change in a PWES, or evaluating opportunity costs of conserving a landscape. However, it is also the most difficult parameter to evaluate in the water balance (Kolka and Wolf 1998) because it is a function of numerous variables that include: precipitation, temperature, solar radiation, soil type, drainage, wind, canopy and understory interception, and vegetation type and maturity. Its magnitude and seasonal variability are not easily measured using standard techniques due to the size and complex surface dynamics of forested systems (Kolka and Wolf 1998).

AET is driven by the available supply of energy, such as from sunlight and wind, but is limited by levels of soil moisture and by the extent to which this can be accessed by plant roots. The supply of water is constrained by the capacity of the soil to store water and release it, and by access to it by vegetation roots. The implications of this are that accessibility to water in shallow soils is similar for plant species with deep and shallow roots, whereas deeper soils will exclusively supply plants with deeper roots. Impacts of land use on the water balance are therefore expected to be more significant in catchments with significant soil cover. In the tropics in general, water available for evapotranspiration is markedly seasonal because moisture in a basin is supplied by seasonal rainfall. This seasonal supply of water limits the amount of water that can be vaporized, despite the relatively continual source of energy in the tropics.

Estimates of AET are made by first approximating reference evapotranspiration (ETo) and potential evapotranspiration (PET), which describe the energy and resistance to vaporize water. Reference Evapotranspiration (ETo) is a measure of the energy available to vaporize water as a function of climate and vegetation type. It is indicated by the amount of water that transpires for alfalfa at a height of 30 to 50 cm that is never limited by water supply. To account for the effect of vegetation type on energy supply, estimated rates of reference evapotranspiration are modified so as to obtain Potential Evapotranspiration (PET). If data is available, ETo and PET are relatively easy to approximate and do not require highly skilled staff because algorithms and worktables are plentiful.

1.1.1.1 Reference (ETo) and Potential (PET) Evapotranspiration

The principal problem in estimating the energy available for evapotranspiration is that data are often limited or algorithms used are far too data intensive. Many ETo algorithms are excessively complex with limited returns on their accuracy, especially if there is no data from field studies or from detailed meteorological monitoring programs that can be used to corroborate them. It is generally sufficient to approximate ETo and PET using simple methods at the onset of a watershed program, which can be improved later once more information is gained from monitoring programs and on-site experience.

ETo can be approximated using regional charts, often available at government environmental agencies. These charts are often based on a combination of interpolating pan-evaporation data, and ETo algorithms. Seldom available are in-catchment measured ETo data by evaporation pans. A rule-of-thumb is that local pan-evaporation data should be used if it exists because it is the most reliable source of information, but is often not available. Regional charts can be used to approximate regional ETo or provide estimates of the variables needed in ETo algorithms. If regional charts do not exist or monitoring programs are shown to be of poor quality (i.e. no quality control), highly skilled scientists and engineers will be needed to sieve through available information, effectively use knowledge/studies from other similar or nearby regions, and provide estimates of uncertainty.

There exist a plethora of algorithms to estimate ETo, many of which are modifications to address transpiration uncertainties that are often site-specific. The classic algorithm is the Pennman-Monteith equation that is data intensive, requiring air temperature, windspeed, relative humidity, and solar radiation data, thus making it impractical in many developing regions. Data for the Pennman-Montieth is available in a global climate dataset provided by the International Water Management Institute (IWMI) but may be inaccurate or lack resolution needed by local planners. Much simpler and less data intensive algorithms exist but these may poorly represent important physical processes. Droogers and Allen (2002) propose an algorithm that uses data often found in developing regions, called the ‘modified Hargreaves’ equation, which produced superior results than the Pennman-Montieth when information is uncertain.

$$ETo = 0.0013 \times 0.408 \times RA \times (T_{avg} + 17) \times (TD - 0.0123P)^{0.76} \tag{2}$$

The ‘modified Hargreaves’ uses the average of the mean daily maximum and mean daily minimum temperatures (T_{avg} in ^{o}C), the difference between mean daily maximum and mean daily minimums (*TD*),, *RA* is extraterrestrial radiation (*RA* in MJm^{-2}d^{-1} and precipitation (*P* in mm per month), all of which can be relatively easily obtained. Temperature and precipitation data are often available from regional charts or direct measurement. Radiation data, on the other hand, is far more expensive to measure directly but can be reliably estimated from tables (i.e. Hargreaves, 1994) or equations (i.e. Allen et al., 1998).

Local Pan Evaporation data are extremely valuable because they provide true representations of evaporation losses from the interactions of wind, solar energy, and relative humidity. In addition, collecting pan-evaporation data with Class-A pans[3] is relatively inexpensive compared to a fully equipped meteorological station. Nonetheless, given a limited budget, resources are better spent funding rainfall and streamflow monitoring plans because, relatively speaking, ETo does not vary much within seasons or between years, and reasonable approximations are possible with the above mentioned algorithms.

To summarize, planners should consider the information that is readily available upon selecting appropriate algorithms. Subsequent collection of pan evaporation data, provided resources exist, can be used to corroborate estimates of ETo, modify assumptions, and eventually substitute algorithm approximations.

These estimates of ETo do not consider the effect of the plant interface on potential losses of water from the catchment. Estimates of potential evapotranspiration (PET) determines the amount of water that can transpire as a result of vegetation type given the energy conditions (ETo). An example of landscape impact on evapotranspiration losses is the case of the Mae Theng watershed in Thailand, illustrated by Chomitz and Kumari

[3] Class-A pans are 1.22 m in diameter and 24.4 cm high on a low platform. The pan evaporation from the Class-A Pans are multiplied by 0.7 to obtain the free water surface evaporation.

(1998), where the establishment of fast transpiring pine forests to replace logged traditional deciduous types resulted in water scarcity and thus a change in the traditional crop cultivated by farmers. The energy (ETo) and water supplied to the Mae Theng watershed did not change yet a change in tree species modified the water balance.

PET is approximated by multiplying ETo with a transpiration coefficient, kc, and assumes plants are not constrained by water availability.

$$PET = kc \times ETo \qquad (3)$$

The transpiration coefficient is based on plant maturity and plant type, as well as wind and humidity conditions, which affect plant uptake of water. Not surprisingly, these values can be obtained from many existing charts developed by irrigation research centers or organizations (e.g. see Allen et al. (1998) or website for FAO tables). As a result, kc coefficients are generally available for cultivated crops and are can be used to estimate actual evapotranspiration when plants are not limited by water[4].

Coefficient values for natural vegetation, such as forest strands, are usually extrapolated from tables, such as Table 4. However, they should be evaluated by experts for natural vegetation and unusual conditions. These values are likely to be lower for arid vegetation, which may tend to have low metabolism rates, and closer to unity for well-established grasses and crops. They may also become slightly lower when the canopy of a forest is wetted by intercepted rainfall because the high evaporation rates of the intercepted water (Schellekens et al., 1999) likely cools the canopy itself and reduces the humidity gradient between the leaf boundaries. Despite the scarcity of PET data for tropical forests, the values being continuously added to our knowledge base are consistent (Bruijnzeel, 2000). Rules-of-Thumb values for tropical and temperate forest PET are 3.5 mm/d, and 6 mm/d for fast growing tropical tree plantations (Roberts, 2001). 'kc' values for forest varies between 0.5 and 0.6 (Bruijnzeel, workshop presentation). In the humid tropics it is reasonable to assume that natural vegetation is always mature since the landscape is usually a mixture of species at ongoing life cycles. Plant maturity stage is more important in landscapes dominated by agriculture, particularly for irrigated crops where high demand for water to establish young crops may coincide with periods when water in the catchment is limiting and thus have an important impact when managing for low flow objectives.

Table 4. FAO Mean Crop Coefficients, K_c, for Subhumid Climates

K_c values for selected crop types at their final growth stage for non stressed, well-managed crops with minimum relative humidity ~ 45%, and wind speed (u_2) ~2 m/s) for use with the FAO Penman-Monteith ETo

Small Vegetables	0.95	Cereals	0.4
Roots and Tubers	0.95	Paddy Rice	0.9
Legumes (*Leguminosae*)	0.55	Forages	0.88

[4] Irrigators will provide adequate water to their crops whenever possible.

Oil Crops	0.35	Coffee – bare ground cover	0.95
Palm Trees	1	Rubber, and Conifer Trees	1

Table 4 shows that transpiration coefficients, k_c, does not vary significantly between certain plant species (e.g. small vegetables and conifer trees). In effect, the use of transpiration coefficients might be insufficient to explain the changes in evapotranspiration losses in the above Mae Theng watershed example from (Chomitz and Kumari 1998). Actual evapotranspiration is often more dependant on the existing moisture conditions, plant adaptation to the environment, and root depth of plants. Namely, in determining actual evapotranspiration, "*the interseasonal and interregional variation in ETo is much larger than the intercrop variation [of k_c]*" (Seckler 1996).

1.1.1.2 Estimating Actual Evapotranspiration (AET)

Table 5 shows the root depth of different plant species. Comparing Table 5 and Table 4 it can be noticed that differences between root depth of certain species are bigger and consistent when compared to transpiration coefficients of those same species. In particular, trees have deeper roots than shrubs, and to a greater extent herbaceous plants and as a result can access water at greater depths (Table 5). Notice the root depth differences between tropical evergreen and deciduous trees as a possible explanation to the example of the Mae Theng watershed. Transpiration coefficients, on the other hand, often have similar values for these plant species (e.g. Notice the k_c value for both small vegetables and pine trees in Table 4is close to unity) because they assume all plants have equal and ready access to water. Canadell et al. (Canadell, Jackson et al. 1996) review 290 observations in the literature and conclude that rooting depths are more consistent than that previously believed among similar biomes and plant species.

Table 5. Maximum root depths by species and biomes (Canadell et al., 1996).

Root Depth by Species	Root Depth by Biome
Trees 7.0 m	Cropland 2.1 m
Shrubs 5.1 m	Desert 9.5 m
Herbaceous Plants 2.6 m	Sclerophyllous Shrubland & Forest 5.2 m
	Tropical Deciduous Forest 3.7 m
	Tropical Evergreen Forest 7.3 m
	Grassland 2.6 m
	Tropical Grassland/Savanna 15 m
	Tundra 0.5 m

Adequate approximations of actual evapotranspiration (AET) require incorporating plant access to water and seasonal availability of moisture in the calculations. Although other factors might be involved in the process of AET, such as plant physiology (e.g. conifers can exhibit substantial stomatal control due to reduced aerodynamic resistance during dry periods[5]), they are beyond the scope of this text, which only presents guidelines for developing the key baseline estimates.

[5] From FAO website on mean crop PET coefficients.

Often AET is estimated while calibrating a hydrologic model to optimize observed and simulated streamflow given an expected precipitation forcing. A modeling approach requires a specialist skilled at interpreting both the modeling and hydrologic processes because a danger exists in poorly capturing conceptual processes in the "black box" between rainfall input and streamflow output. Obtaining true AET is extremely difficult to measure (particularly in forests) and research is often disarticulated due to disciplinary differences. For example, research that measures for sapflow to understand AET of specific sets of specimens neglects basinwide AET effects associated with the full range of species.

Fortunately, simple models exist with which to develop baseline estimates of AET that keep track of the soil water budget. One such popular water budgeting approach, presented by Thornthwaite and Mather (1957) (1957), requires PET (discussed in previous sections), precipitation, effective soil depth (root depth or depth to bedrock, depending which is lower), and soil porosity (available in soil tables or by soil scientist assessment). The Thornthwaite-Mather (T-M) budgeting approach treats the soil profile as a reservoir that has water entering the system as rainfall, and leaving as either AET or as excess surface or sub-surface water based on retention and storage rules (see Annex A). Modifications by Steenhuis and Van Der Molen (1986) [6] and Kolka and Wolf (Kolka and Wolf 1998), with the US Forestry service, have improved the method of estimating AET in climates with pronounced dry periods followed by wet ones (and vice-versa), such as in the tropics. In general, estimates of AET, using such methods, must account for energy (PET) and water budgets (precipitation) on a daily basis over several years and reported at larger time steps, such as average monthly or annual rates.

Actual evapotranspiration (AET) in the tropics is seasonal because it is related to the seasonality of rainfall, while in contrast, the amount of energy (PET) that drives this process is fairly constant. PET and ETo estimates are therefore often insufficient to develop water budgets. Annual estimates of AET, on the other hand, are necessary for linking landscapes and water budgets because they account for (i) moisture distributions throughout the year, (ii) the amount of water the soil can store and retain, and (iii) the access that plants have to it. Fortunately, there are fairly easy water budgeting approaches such as T-M method, that can be taught to planners and field staff by specialists, and that can provide baseline expectations of landuse management on annual water yield goals.

The T-M method is very simple and therefore, does not consider other impacts of landuse change on AET, such as soil storage capacities that may be lost by poor management. Such detailed analysis is often site specific requiring skilled personnel and specialists that are likely to extrapolate findings from the region. For example, it may be important to investigate the relevant impacts of landscape changes on soils given that plant yields are highly correlated to AET (Seckler 1996). Namely, opportunity costs may exist to appropriately manage a forest because a reforested landscape may produce lower quality trees (lower yields) if soil is irreversibly degraded, due to reduced soil-water retention and storage.

[6] The Thornthwaite-Mather water budgeting approach in Annex A is based on modifications by Steenhuis and Van Der Molen (1986)

Methods to directly estimate basin AET without the use of hydrologic models or soil-water budgeting accounting is very difficult. Proposals have been made to use remote sensing methods (e.g. Wiegand and Richardson, 1990) but these are usually only practical for irrigation systems or riparian zones, where the water supply is continuous, otherwise intensive and expensive field verification with consecutive 'fly over' studies throughout a year are necessary.

A sophisticated and relative simple procedure with which to estimate actual basin-wide AET from readily available data is presented by Dias and Kan (1999). To obtain basin-wide AET for two catchments in Brazil, they analyze streamflow recession to determine differences between soil water storage and precipitation. They assume loss to deep groundwater is negligible and the only data needed is daily streamflow and precipitation (see Annex B). The method requires good analytical skills and data that is easy to obtain but can also produces some errors that are expected from the methodology. The approach by Dias and Kan (1999) captures the seasonality of AET that should be used to tailor AET models and budgeting approaches, thereby securing consistency with observed transpiration losses by directly testing and verifying assumptions and methods. Most importantly, this can be done with easily obtained data and underlines the importance of monitoring streamflow and precipitation, and ensuring this data quality.

Changes in actual evapotranspiration impact the annual water yield of a catchment, which are linked to seasonal availability of moisture and vegetation types that can access this water. As a result, planners require knowledge of landuse in the catchment. At the onset of an assessment, this knowledge does not necessarily need to be highly detailed but does need to distinguish key species for their capability to obtain water for transpiration, i.e. rooting depth, PET (kc values for plant maturity stage[7]), and general soil characteristic[8] (capacity to store and retain water). The text has thus far highlighted the important differences between herbaceous plants and trees in their capacity to access water due to rooting depth. This is but one important criteria in landscape assessments for water balance estimates and other criteria might be considered should planners deem necessary. In general, the availability of streamflow and precipitation data complements landscape information and provides planners with baseline water yield expectations from watershed management plans.

Data on land cover and landuse.

A major obstacle to determining AET is the difficulty of obtaining site-specific land cover and land use data that reflects significant heterogeneities of the landscape. These tend to be areas that are small relative to their impacts, may reflect particular land use practices, and are often difficult to detect with most remote sensing technology (van Noordwijk, Poulsen et al. in press). As noted above, many of these features operate at the scale of individual hillslopes.. For example, narrow riparian areas can have effects on

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[7] It would be reasonable to assume that an undisturbed landscape in the humid tropics has mature vegetation.
[8] Soil data is usually available from soil maps but specialists should make on-site assessments if possible.

hydrology that are disproportionate to the area they occupy because they have easy access to the watertable resulting in AET close to PET.

Remote sensing technology that obtains spectral images of landscapes has the potential for supporting the management of river basins but still needs better implementation (Bastiaanssen 1998). The most readily available and useful tools for landscape assessments are topographic maps, soil maps, and aerial photography. In general, topographic maps provide hydrologists with convergence areas where intermittent streams or wetlands are likely to occur, they show perennial streams, provide indication of soil depth by means of slope steepness, and usually show approximate areas that were forested or under agriculture at the time of map production. These are some observations from topographic maps that can provide indicators of expected vegetation types. However, topographic maps are seldom updated and will have errors associated with existing landuse. Aerial photography is often obtained with more frequency, usually by the military or air force to ensure updated maps for strategic purposes. These aerial photographs can often be purchased (unless in a zone of conflict or national security) and viewed with stereoscopes to provide 3-D images that are the best source of information, updated with regularity. These 3-D images can easily distinguish between shrubs and forests unlike many forms of satellite images, and are particularly useful when they are taken in the infrared spectrum because heat distinguishes the most active vegetation. Their principal drawback is that landscape analysis using aerial photographs may be time-consuming for large basins. Clearly, 'ground truthing' or rapid assessment of remotely observed landscapes are an integral part of this process (see Freudenberger (1995), Oba (2001); Sinclair and Walker (1999)).

New active remote sensing technologies such as LIDAR (light detection and ranging) are beginning to be applied to the development of more detailed profiles of the structural characteristics of forests, and their operational use is rapidly becoming more feasible (Dubayah, personal communication). LIDAR is able to detect the vertical structure of forests by measuring the time it takes for a laser light beam to travel round-trip between the sensor and the target as it is reflected from the canopy and ground surfaces. Most work in this field has been based on data from sensors mounted on aircraft but data products with global coverage using satellites are expected from NASA's ESSP Vegetation Canopy LIDAR (VCL) mission at a date to be determined. The VCL mission is expected to provide global datasets of topography, canopy heights and also surfaces of canopy components, (i.e., foliage twigs and branches), which can be used in models to infer a number of other forest characteristics such as successional stage, species composition, biomass, and spatial patterns of both topography and canopy heights (Dubayah and Drake). LIDAR data has already demonstrated the ability to provide more precise estimates of carbon storage in the La Selva tropical forest (Drake, Dubayah et al. submitted manuscript). By allowing better delineation of forest patches with distinguishing characteristics, LIDAR is also expected to significantly reduce uncertainties in watershed process models, and ultimately, in land and water relationships.

In sum, a combination of various approaches and technologies are likely to provide the best characterization of land cover and landuse. These approaches may use state-of-the-art three-dimensional active remote sensing techniques (LIDAR), which may be supplemented by two-dimensional passive spectral imagery (LANDSAT) to more broadly detect special features identified in rapid rural appraisals of landscape management (Freudenberger 1995; Sinclair and Walker 1999; Oba 2001).

Landscape Impacts on Total Annual Water Yield.

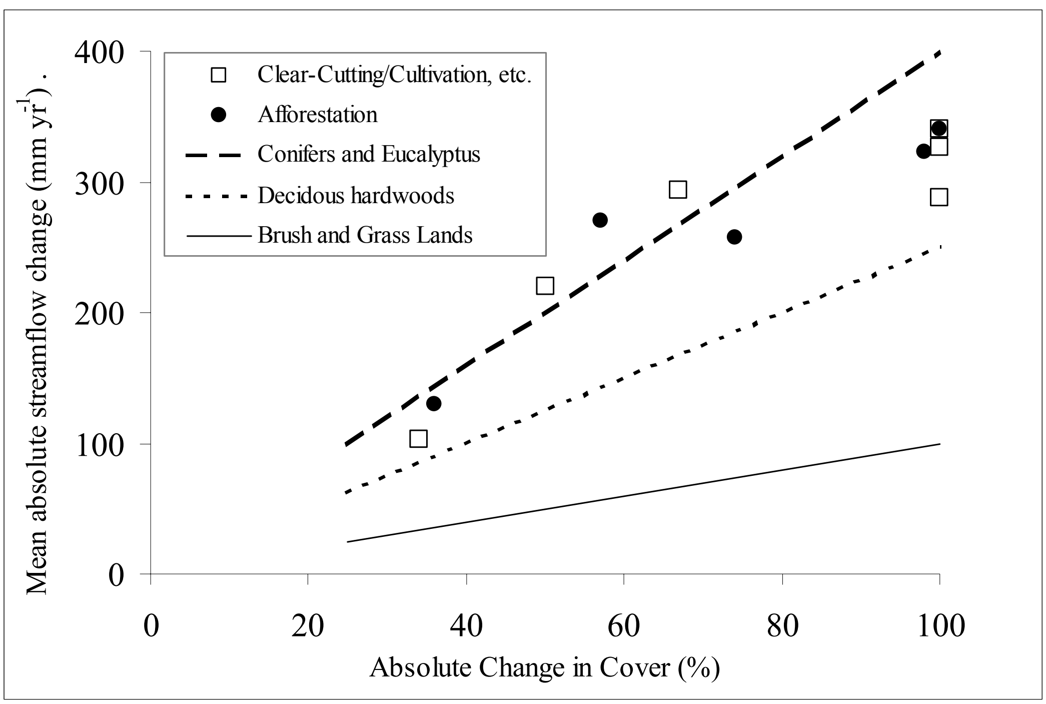
It is extremely difficult to make generalizations based on research about the impact of landuse change on total water yields for three principal reasons;

(i) It has been shown that soil depth, seasonal distribution of moisture, and plant access to water are all key factors in determining AET, meaning that literature observations are often site specific.

(ii) Too many studies in the tropics examine the impact of pine or eucalyptus forests, which may be of concern but may inaccurately depict other forest types or deep rooted vegetation.

(iii) Wide variations in experimental methods have often made pooling of results difficult (Oyebande 1998).

The classic literature review of forest clearing impacts on water yield is Hibbert's (1967) summary of 39 studies, where he essentially concludes that forests reduce water yield in an unpredictable way. Additional reviews of literature by Bosch and Hewlett (1982)[9], and Oyebande (1998)[10] indicate that, on the contrary, it is possible to find direct relations between water yield and forest cover, which may be used as general rules of thumb as long as task managers realize that no error limits on the statistics were attempted.



[9] Bosch and Hewlett (1982) study 94 forest clearing or planting experiments, including those by Hibbert (1967).

[10] Oyebande (1988) examines 23 studies in tropical and subtropical watersheds.

Figure 5. Average annual water yield changes following changes in vegetation cover (points are based on Oyebande, 1988; lines are based on Bosch and Hewlett, 1982).

Figure 5 shows expected changes in water yield due to afforestation or clearing[11] from a review by Oyebande (1998). In addition, it includes line plots that are trends identified in a review by Bosch and Hewlett (1982)[12], where the coniferous line plot is fairly consistent with Oyebande (1998). Oyebande (1998) argues that the plot seems to show a 5mm change in annual water yield for each percentage of change in forest cover. However, the plot seems to indicate a maximum threshold change of average water yield possible around 350mm. Oyebande (1998) and Hibbert (1967) suggest maximum water yield changes during the first year after deforestation, which is less useful for the long term planning of a PWES, can result in annual streamflow increases of up to 580[13] and 450 mm. Effects of landscape vegetation change should therefore be gauged when vegetation and soil has reached a stable equilibrium. For example, growth of scrub vegetation and sealing up of soil macropores can reduce yield gains in subsequent years after deforestation.More relevant to a watershed planner is that there is a maximum demand (PET) and supply (precipitation) for actual evapotranspiration, which is mediated by soil storage, and which will constrain loss of vaporized water from the system.

Preliminary expectations of the proportion of AET in the water balance can be made by adopting this supply-demand-storage hypothesis as discussed in Milly (1994). To do this, we may use Budyko's (1974) dryness index[14] that provides insight of a catchment's supply-demand character, assuming we can neglect the permeability effects of a soil on hydrology. Given that the dryness index, DI, is a function of PET and precipitation it is not surprising that catchments with similar DI have similar vegetation (and thus, perhaps, soil structure) that have evolved accordingly. Farmer et al. (Farmer, Sivapalan et al. 2003) points out that catchments with similar DI have similar flow duration curves (see Figure 6), which means they have characteristics of baseflow maintenance and stormflow frequency. Budyko (1974) finds relatively consistent relations between the ratio of actual evapotranspiration and precipitation, in studies of DI data from a large number of catchments, meaning that characteristic catchments (with similar DI) have similar proportions of AET losses.

Milly (1994) finds that when DI > 2 (an arid watershed) in catchments that have out-of-phase PET and P seasonal patterns, such as in the tropics AET often ranges between 70-90% of total precipitation. This expected percentage decreases in more humid tropical catchments that have a lower DI. Typically, when DI = 1, expected AET tends to range between 60-80% of precipitation, and when DI = 0.5, AET tends to range between 40-50% of precipitation (Milly 1994). Perhaps more importantly in terms of landscape

[11] Water yield increases from clearing are averages from about 3 years post treatment. Water yield declines from afforestation are averages made about 20 years post treatment, from planting of Eucalyptus and Pine trees.

[12] In sum; Conifers and Eucalyptus, 40mm change in water yield per 10% change in landscape cover; Deciduous hardwoods, 25mm per 10% change in cover area; Brush and grass lands, 10mm per 10% change.

[13] This value is probably too high because Oyebande (1988) extends a fitted curve beyond data points.

[14] The dryness index (DI) is defined as ratio between annual potential evapotranspiration and precipitation, i.e. DI = PET/P.

impact on the water balance, Milly (1994) analytically estimates that a reduction of about half of the plant available water storage capacity, resulting from degraded soils or shorter root depth, will increase water yield by approximately 10% of precipitation when DI > 0.8, and that such percentage gains become less significant as DI approaches zero. For example, using Milly's (1994) analysis, a change in forested cover at El Yunque may result in a smaller percent increase of water yield than a similar change of cover at Bullock creek (Figure 6). The key point then, is that water yields in drier catchments (DI > 0.8) are likely to be more sensitive to landscape or soil structure changes than in wetter catchments.

Regulation of Streamflow

Changes in total annual water yield is insufficient to characterize changes in the timing of and maintenance of streamflow. Often watershed managers consider the socio-economic impacts of changes in streamflow during the dry season because this is when water is limiting in agriculture and people develop local production strategies to manage risk. Conversely, watershed managers may be interested in streamflow during the wet periods because this is when costs of flooding and erosion are greatest. Estimates of streamflow based on water balance equations are very unreliable and are seldom used for practical purposes, because of the uncertainties in the estimation of AET and other variables,. Measured data is always preferred. When it data is not available, regional formulas that relate precipitation and basin area to streamflow, or even conceptual rainfall-runoff models are better substitutes.

Land management can alter the streamflow regime if basin-wide AET is significantly modified by changes in vegetation type that establish new rooting depths, or alteration of soil structure that affects its permeability and storage capacity. Valuation of PWES services associated with streamflow is uncertain because landuse-hydrology linkages are often site specific, and require detailed studies and modeling. However, an attempt will be made based on general expectations that can be made for characteristic catchments.

Dry Season Flow

Dry season flow or baseflow results from the release of water to streams from shallow aquifers and occurs throughout the season, whenever there is streamflow. This section addresses baseflow during the dry season, when it is often the dominant component of streamflow. Principal characteristics of the watershed that affect this flow are:

i) Permeability of the landscape, which determines the ability of the contributing aquifer to recharge during a rain storm. Key factors that determine permeability are vegetative ground cover, soil depth and soil texture. This is because roots, micro-biota, and organic matter facilitate infiltration of water from rainfall, deeper soils take longer to saturate, and the texture may increase or restrict infiltration, depending on the clay content. In general, the presence of more vegetation with deep roots increases infiltration of water into the ground during a rain storm, which is then stored in the shallow or stream contributing aquifer.

ii) The hydraulic properties of the contributing aquifer largely control the release of water from storage as baseflow. In this text, we limit discussion to the storage properties of the aquifers, in which incorporate soil depth and porosity are compounding effects. Other parameters such as hydraulic conductivity and slope also impact the timing and magnitude of baseflow. Water stored in the shallow aquifer that is slowly released to streamflow may reside within pores of a soil or in fractures of bedrock. For the most part, watershed management does not impact the hydraulic properties of these shallow aquifers, although severe soil loss can have adverse impacts on storage capacity.

iii) Plant root access to the stream-contributing aquifer depletes stored water that would otherwise be released to the stream. As in item (i), watershed management activities can modify baseflow. However, in this case, an increase in vegetation, particularly deep rooted, usually has the opposite effect, of decreasing baseflow. Water stored for baseflow is largely depleted by plants that can access the saturated zone or water-table. Vegetation near riparian zones can readily access this water, whereas saturated zones in deep soils or within fractures of bedrock are harder to access. Therefore, knowledge of the dominant aquifer source of baseflow (item iii) is necessary to predict magnitude of impacts of landscape plants on baseflow.

Landscape impacts on baseflow are very site-specific (Calder 1998) and it is difficult to provide general rules of thumb. On one hand, vegetation improves the permeability of a watershed thereby increasing baseflow. On the other hand, vegetation can access water and reduce baseflow. In addition, the objectives of maintaining baseflow generally pertain to securing or reducing stress during the dry season or droughts when water is limiting and thus the value of water high. Many irrigators who lack the resources to construct significant hydraulic retention structures rely on diversion of baseflow for their livelihood during the dry season. Sustaining baseflow during the dry season also has consequences for biodiversity, in that small variations of baseflow during the dry season may have impacts disproportionate to its volume. For these types of management questions, planners require reasonable expectations of baseflow despite the difficulty in incorporating all the interdependent variables.

Conceptual hydrologic modeling will usually be required to quantify impacts of landscape management on baseflow, and requires substantial expertise. Modeling must adequately capture the conceptual processes of baseflow in order to have identifiy effective management interventions. At the very minimum, models should be calibrated to correctly represent low flows but this does not guarantee that the underlying processes are correctly represented. As is illustrated in Figure 6 this requires correctly predicting flows that rarely exceed the mean flow (parts of the flow curve below y-axis = 1). The section on conceptual process models discusses state-of-the-art approaches to correctly representing the underlying hydrology.

A useful indicator of characteristic streamflow for a catchment is the index of dryness, DI, defined as the ratio of potential evapotranspiration to precipitation (Farmer, Sivapalan et al. 2003). Catchments with similar dryness index, DI, have often been found to have flow regimes with similar characteristic (see Figure 6). Catchments with lower DI values, i.e. more humid catchments, often have more prolonged baseflow because they have well formed and recharged shallow aquifers. Flow regimes with steeper flow duration curves have limited potential for baseflow and/or have significant numbers of water users, diverting and depleting water (e.g., irrigators). It is the catchments with potential prolonged low flow regimes that may present opportunities to significantly impact drought flow expectations through watershed management activities. As stated previously, quantifying the magnitude and direction of impacts on baseflow are more complex and site specific, often requiring hydrology models to represent the conceptual hydrologic processes.

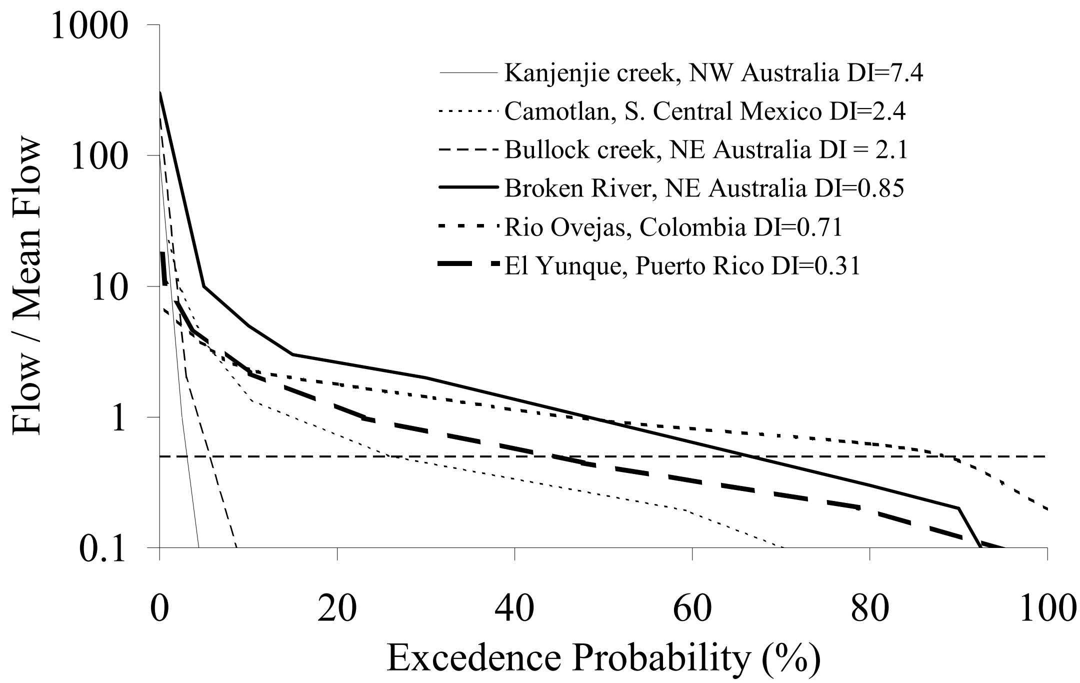


Figure 6 Flow duration curves for several tropical watersheds with different index of dryness indices, DI. Dividing expected flows by the mean flow normalizes the curves. The dashed horizontal line denotes half the average flow. Sources: Australian catchments (Farmer et al., 2003); Camotlan (Comission Nacional de Agua, Mexico); Rio Ovejas (Luijten, Jones et al. 2000); El Yunque (United States Geological Service).

Changes in water yields associated with management actions can modify the flow duration curve, and provide a basis for approximating their costs and benefits . However, the distribution of changes in water yield is not evenly distributed throughout the flow regime. Investigations at Cathedral Peak watersheds show these seasonal variations in water yield due pine afforestation that reduced dry season flow by 15 mm although annual streamflow was reduced by 440 mm (Bosch 1979). Availability and proper interpretation of daily streamflow data, to verify assumptions and modeling approaches, is essential in planning for drought flows. For example, Figure 6 shows that the arid catchment at Camotlan, which has a DI value of 2.4 (similar to Bullock Creek) has a prolonged baseflow regime that makes it more akin to humid basins. Camotlan lies in the mountainous Southern Sierra Madre and baseflow is established by rainfall recharge, during the wet season that percolates into the fractured matrix of the rocky hillside. Despite the lack of soil cover on the hillsides of the Camotlan watershed, water is stored in the rock where vegetation has limited access, thereby minimizing evaporative losses. Therefore, land management practices that encourage infiltration in the hillsides effectively recharge baseflow. Not surprisingly, traditional practices include detention basins on the hillsides and the construction of filtration galleries that assist in collecting percolation water. In effect, regional non-profit organizations are highly successful in improving the production of springs by constructing small retention reservoirs upstream, which recharge the fractured bedrock aquifer.

In general, flow duration curves with prolonged low flow regimes indicate the presence of shallow aquifers that contribute to streamflow, which can be impacted by landscape management practices. The duration curve representing Rio Ovejas has well sustained baseflow and low high flows (Figure 6). The soils in the watershed are very permeable[15], retain water very well, and are generally deep enough not to restrain root growth (Luijten, Jones et al. 2000). However, the landscape is typical of well-populated Andean watersheds; dominated by pasture, bush scrub, and crops, which have medium rooting depths, which likely have limited access to the water table Basinwide aforestation might reduce drought flow if deeper tree roots obtain considerable access to the water table.

Expected impacts of management on annual yield of baseflow can be assessed by estimating the change in area under the flow duration curve[16]. The use of a flow duration curve incorporates the uncertainty of streamflow in a robust way that can be used to reasonably quantity expected flows for any given time period.. To illustrate, Figure 7 presents plots of two modeled scenarios in a hypothetical watershed, where the impact from 100% deforestation is assessed. In this hypothetical example, benefits of deforestation are high because soils do not degrade, water quality is maintained, and expected water yields from the deforested watershed are larger than the forested one (the total area under each of the curves). However, in this hypothetical watershed, costs may result from an expected increase in high flows (the area between curves in the 0 and 20% range), and costs may result from a reduction in expected flows at the midrange (area between the 20 and 70% range). In addition, benefits from deforestation, in this hypothetical example, may occur at the low flow range where drought flow is increased. Even though the magnitude of changes is very small at the low flow regimes, these flows

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[15] The high permeability of the landscape increases the recharge of the shallow aquifer at the expense of water flowing quickly into the stream. Notice the relatively low normalized flows at the lower exceedance rates.

[16] $$E(Q) = L \int_{\text{lower limit Pb}}^{\text{upper limit Pb}} Q(Pb)\partial Pb$$

E(Q) is the expected water yield during a span of L days. Q (Pb) is the flow duration curve function (Figure 6), and Pb is the probability ($0 < Pb \leq 1$).

may be associated with high willingness to pay by irrigators who do not have the means to construct retention structures.

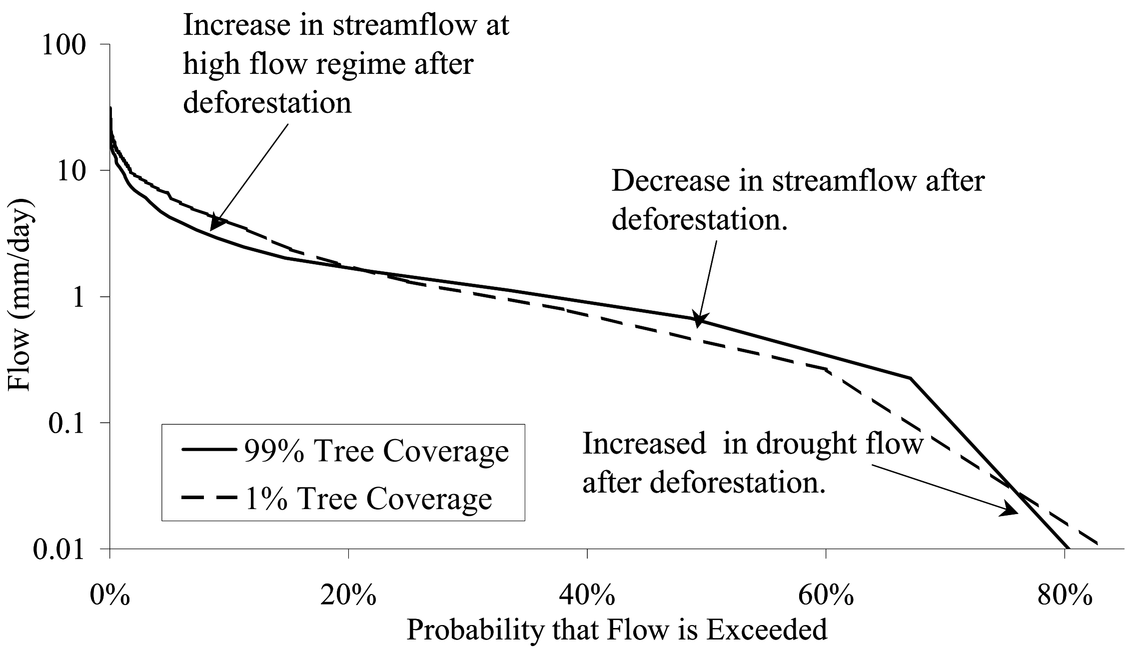

Figure 7. Two simulated flow duration curves of a hypothetical watershed, simulated with SWAT model,[17] using meteorological data from Tegucigalpa, Honduras, and assuming soils no more than 1m deep, and that there are no losses to deep groundwater. The thick plot represents the catchment at 99% forest cover. The thin dashed line represents streamflow prior to deforestation.

As always, gaps in data pose a problem for watershed planers, particularly in developing regions of the world, and creative expertise is often required. Poorly derived flow duration curves, i.e. from small incomplete data sets, can be verified or perhaps extrapolated from field observation and rapid ground assessments. For example, volumes of water flowing at bankfull in alluvial streams, necessary to maintain stream bank (Savenije In press), are estimated to occur between 1 to 3 years (Millet 2003). This type of knowledge can be used to verify the flows for corresponding frequencies in the duration curve. Similarly, expected duration of drought flows may be obtained from local knowledge by means of focus groups, thus providing "fuzzy" ranges of minimum flows and duration of these into the dry season. These activities requires fairly skilled assessments with expertise on hydrology and un-biased interview methodologies.

Changes in the water balance during the dry season are particularly difficult to gauge because, as stated by Calder (1998), "*Different, site-specific, often competing processes may be operating and the direction, let alone the magnitude of the impact, may be difficult to predict for a particular site*". A key process during dry periods is the interaction and timing of precipitation and PET (Milly 1994). Limited cases have been

[17] SWAT model; Soil Water Assessment Tool (Neitsch et al., 2002) is a popular hydrology model developed by USDA-ARS, Texas.

found in which deforestation decreases dry season flow (Whitehead and Robinson 1993). The effect of vegetation on dry season flow has been found to diminish (and becomes limited to about 0.1 to 0.3 % of the catchment) as the stream dries up (Bond, Jones et al. 2002). This is probably because, as baseflow dries up, it finds flow paths beyond the reach of most roots and therefore beyond the reach of ET. Ice (2003) finds that water yield increases due to deforestation are lowest during dry periods. Therefore it is often necessary to rely on conceptual process modeling as the best tool to predict the magnitude and direction of impacts from landscape management projects.

To sum up a misunderstanding of dry season flow: It is not a myth that forests improve the permeability of a soil horizon increasing the amount of water that can be stored. The myth is that gains from additional infiltrating water are available to streamflow. More likely than not, the forest itself will transpire much of it unless a flow path can be established beyond the reach of roots, such as in fractured bedrock rock or very deep permeable soils.

Stormflow

Stormflow, also known as quickflow or surface runoff, is one of the most important processes in catchment hydrology because it is associated with erosion, floods, and contaminant transport. Stormflow results largely from rainfall rates that are in excess of landscape infiltration, shallow subsurface flow paths, or from saturated areas in the landscape (see (Naef, Scherrer et al. 2002)). Nonetheless, many processes are still not well understood at larger basins scales (Uhlenbrook, McDonnell et al. 2003). In October 2000, 80 scientists from around the world gathered in Freiburg, Germany, to address modeling of dominant runoff generation processes at the meso- and macro- level to address the needs of future water resources management (Uhlenbrook, McDonnell et al. 2003). Although it is regarded as a major challenge, much has been learned and is being applied in watershed management for stormflow.

The need to predict flood-flow volumes, for hydraulic engineering design, in ungaged watersheds has resulted in a good understanding of landscape-stormflow linkages, even if methods are developed from empirical relationships rather than hydrological theory. One such method used in the US is the SCS-Curve Number method that has been successfully used since its development in 1940 by the USDA Soil Conservation Service. Tables are easily obtained that can be used to characterize the potential for storm runoff of a landscape using factors such slope, vegetation, soil, and management practice. This empirical method has been shown to adequately represent both the conceptual hydrology of infiltration excess runoff and, more recently, saturation excess runoff (Steenhuis et al., 1995). However, several hydrologic studies show little linkage between landuse and stormflow (Calder 1998)).

Storm runoff volume on the landscape is managed by modifying infiltration rates and/or surface roughness - the first enhances subsurface flow and the latter decreases the velocity of surface flow. Given that reductions in storm runoff volume often result a reduction in peak flow in similar proportions, these practices are often targeted to manage peak flows. Table 6 shows management practices that target infiltration or surface roughness aspects of the landscape to reduce the volume of storm runoff.

Table 6. Principal effects of land use and treatment measures on direct runoff (source: USDA: National Engineering Hand Book, 2000).

Management Practice	Reduction in direct runoff volume because of:	
	Increasing Infiltration Rates	Increasing Surface Storage
Plant or Root Density	Yes	
Mulching or Plant Litter	Yes	
Contouring		Yes
Contour furrowing		Yes
Level terracing		Yes
Graded terracing		Yes

Storm runoff peaks are also mitigated by runoff management practices because of a time lag between formation of runoff in the landscape and subsequent entry into the stream channel. Table 8 shows the management practices listed in Table 7 insofar as they increase the lag time of storm runoff into the stream. Landscape management does appear to reduce the volume of runoff into a stream, at least at the scale of management. However, conditions in particular tend to reduce the impact of landscape management on storm runoff volumes:

i) When the landscape is moist from heavy rains the permeability and surface storage capacity of a landscape decreases considerably, muting the capacity to mitigate extreme flood events through landscape management practices. Severe rain storm events usually occur during the rainy season when the soil is already wet. For example, Hurricane Mitch struck Central America towards the end of the rainy season, over a 6 day period, which resulted in extremely wet soil conditions. The amount and intensity of rainfall delivered by Hurricane Mitch was comparable to many of the severest hurricanes and tropical storms in the Atlantic basin (Hellin et al., 1999), though not the highest on record. It is unlikely that landscape management of storm runoff could have significantly mitigated the damage caused.

ii) Scale is an important consideration in determining the downstream impacts of landscape management of storm runoff for flood control. In larger basins the peaks of storm runoff often dissipate due to the larger area lag effects, the instream channel dispersion and transmission losses of the flow pulse, and the spatial extent of rainfall (it does not rain on all catchments in a basin at the same time). A widespread rainfall event may indeed result in a flood at a watershed scale but these are usually mitigated with emptied reservoirs managed for flood control. The question of the magnitude of scaling of landscape impacts to runoff volume is still not well understood. Stomph et al. (2002) largely explain the phenomena of runoff reduction with increasing slope length and validates a process based model in West Africa. More useful at this point though, are rules of thumb that might be taken from Kiersch (2000). Kiersch (2000)observes from case studies that land use induced change on the hydrologic regime is perceptible up to 3 orders of magnitude larger than the scale of management.

Table 8. Relative effects of land use and treatment measures on types of lag (source: USDA: National Engineering Hand Book, 2000). Small watersheds refer to those less than 800 Ha.

Management Practice	Effect on subsurface flow		Effect of increasing surface flow length or decreasing velocity	
	Small watersheds	Large watersheds	Small watersheds	Large watersheds
Plant or Root Density	Can be large	Can be large	Not usually considered	
Mulching or Plant Litter	Can be large	Can be large	Not usually considered	
Contouring	Can be large	Often negligible	Can be large	Negligible
Contour furrowing	Can be large	Can be large	Not usually considered	
Level terracing	Can be large	Can be large	Not usually considered	
Graded terracing	Often negligible	Often negligible	Can be large	Negligible

Landscape management options to mitigate the damage of flood events may be limited at a basin scale. There is also limited potential for mitigating extreme events, such as Hurricane Mitch, at any scale. Nonetheless, reductions of storm runoff volume, from chronic less extreme events, can be tradable as services at the watershed scale. Storm runoff is also the source of most constituents that degrade water quality, and increases the capacity of a stream to transport sediment – which is related by a power function to streamflow (i.e. a unit increase in streamflow results in a larger increase of its sediment carrying capacity).

Identifying the processes and sources of storm runoff provides a framework with which to define watershed services needed to assure water quality and reduce sedimentation. The impacts of landscape management on improved water quality and reduced sediment are felt at larger scales (an order of magnitude of 4 for nutrients) than the hydrologic changes (Kiersch 2000). In the case of sediments, these often become deposited on the stream channel but become re-suspended. Identified sources of storm runoff can be identified and reduced with a land use management practice that will either increase infiltration or surface storage, or reduce soil moisture by increasing deep rooted vegetation.

Infiltration excess runoff occurs in areas with poor permeability, such as roads, pavement, plots with poor agricultural practices, clayey soils, etc. It is in these areas that most potential exists in reducing storm runoff by means of changes in land use and management, such as those listed in **Table 6**. This is only feasible on sites where infiltration can be enhanced or surface storage increased (Naef, Scherrer et al. 2002). The sources of infiltration excess runoff are identified using soil maps or landuse maps because they are largely dependant on the surface landscape. Land use management to reduce infiltration excess runoff is only effective for intense rainfall events resulting from convective storms, in contrast to long lasting, low intensity advective storms. Since convective storms are also very localized, landuse management to reduce infiltration excess runoff is usually negligible at the large basin scale.

The sources of saturation excess runoff, on the other hand, are dependant on the morphology of a catchment and as a result have patterns that can be identified in the watershed. The topographic index[18] is used to identify the areas most likely to generate saturation excess runoff. In a GIS map the watershed can be represented by grid cells, each with a topographic index value estimated using morphological characteristics at that point. Cells with the highest topographic index are the areas that might be prioritized for management. Management actions on the highest topographic index zones will result in higher returns, with diminishing returns on management investments at the lower topographic index zones. Areas of high topographic index value might be designated "Hydrologic Sensitive Areas' (Walter, Walter et al. 2000) where plans to protect water quality might be put into effect. Figure 8 illustrates an example of the use of topographic index to identify areas to saturation excess runoff. The darker red areas are zones very likely to have saturation excess runoff, and are found along riparian zones. Indeed, riparian zones should be protected as a rule of thumb.

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[18] $$\lambda = \ln\left(\frac{a}{\tan(\beta) D K_s}\right)$$

where a is the area contributing to a point in the watershed, β is the slope, D is the depth of soil, and K_s is the permeability of the soil (from Walter et al., 2000).

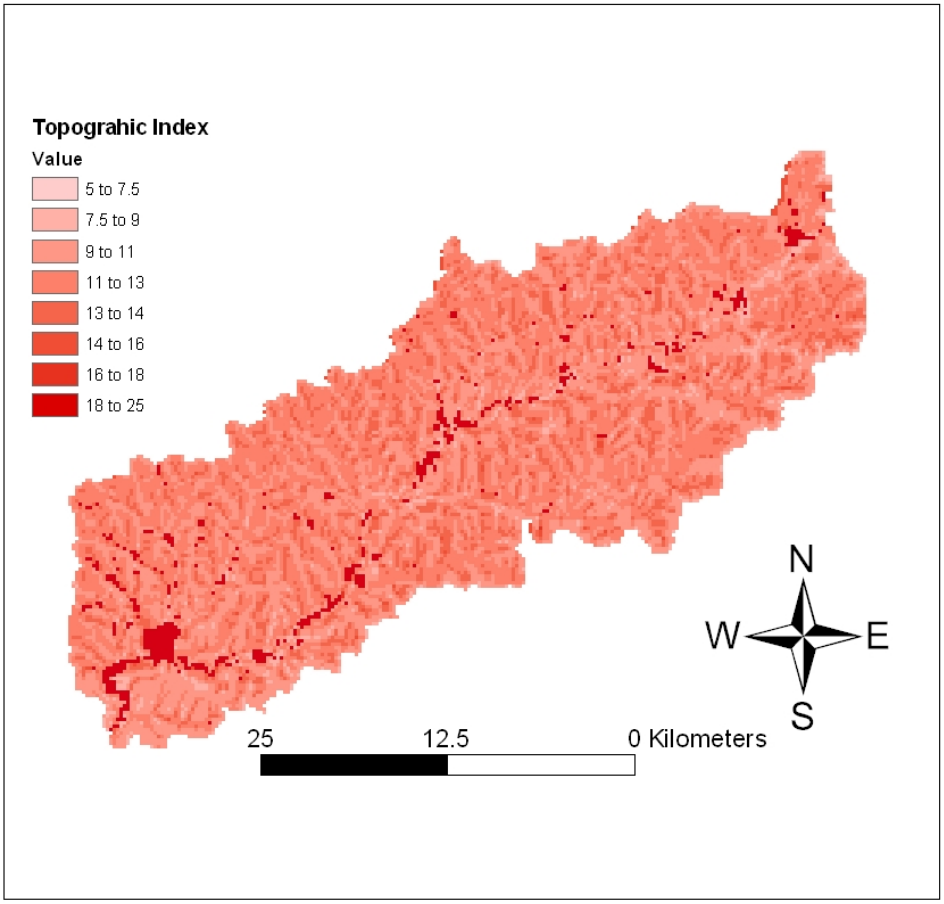

Figure 8. Topographic Index values for the Cannonsville Watershed in the Catskills, NY.

Water Quality

The discussion herein concentrates on sediment as a principal source of water quality impairment because it is greatly related to the water balance (i.e. stormflow), and because other polluting constituents from surface sources often have similar dynamic as sediment transport. Identification of sources of saturation excess runoff (i.e. using the topographic index) or infiltration excess runoff (from low permeable surfaces) is key to management of surface pollutants that may enter water bodies.

The main obstacle is a lack of information on erosion rates and sources of sediments in the tropics. The Universal Soil Loss Equation is often inappropriately used and sediment delivery ratios are unknown (Nagle et al., 1999). Sediment delivery ratios are related to the topography and vegetation of a catchment. In general, watersheds with dense vegetation, wetlands, riparian vegetation or with gentle relief topography have lower sediment delivery ratios.

Sediment budgets are key to targeting the most effective areas of intervention (Nagle, Fahey et al. 1999). Determining the sources of soil loss is difficult. Experimental field studies are difficult to perform and are often the source of error in measurements (i.e.

trenches and erosion pins disturb the soil). Measurements of Cesium from Radioactive fallout deposited in soil before the world ban on "open air" nuclear testing, is often considered the most reliable evaluation method. However, Cesium measurement in soil is usually expensive (particularly in developing regions of the world), requires highly specialized personnel, and is biased towards the finer sediment particles. Additional difficulties arise from the question of scale. For example, a soil conservation study may not adequately represent sediment loadings to a water body if much of it becomes deposited on the landscape itself. The following are key findings that may assist in evaluating or determining ballpark estimates of sediment yields;

a. Natural erosion rates can be extremely high in many geologically recent humid tropical watersheds. Landslides (mass wasting) are important point sources of sediment that are hard to measure or prevent (Nagle, Fahey et al. 1999) estimated 81% of total natural erosion came from landslides in a small forested Puerto Rican watershed).
b. Much sediment transported by streams is stored in the river bedload (34%-92%) itself. Therefore, current landscape management practices will have little immediate impact on decreasing potential sources of sediment (Nagle, Fahey et al. 1999). The bedload sediment originates from the landscape itself or from eroding river banks.
c. The increase of needed roads in the rapidly developing tropical hillsides is often responsible for collecting runoff and intercepting subsurface lateral flow, which concentrates to create mega gullies that often become landmarks with names (e.g. Cárcavas in El Salvador).
d. Much eroded material (~90%) is stored in the basin landscape itself (on the gentler slopes) as potential sources of sediment for the next big storm (Nagle, Fahey et al. 1999). .

Most importantly, the processes of water quality impairments are very complex and difficult to model. In addition, monitoring for water quality is difficult and expensive. Several constituents require refrigeration and laboratory equipment lacking in many developing regions. As a result, it is difficult to validate models used to provide expectations of water quality due to land management. Often the most important landscape management options to improve water quality are to limit polluting activities on areas of high topographic index (Figure 8). Namely, for any given storm return period there is an associated area of saturated excess runoff contribution, which can be equated to a catchment contributing area by summing areas of highest topographic index first.

Aquifer Recharge

It has been assumed thus far that net losses to the water balance from deep percolation are negligible. This may be a reasonable assumption at the head of a watershed but as the scale of management is increased, deep ground water losses or gains become more significant. One problem is to identify sources of externalities that have resulted from surface water use impacts or well pumping elsewhere. Mei et al. (1993) show that by including extractions of water from groundwater, inefficient uses of surface water might actually translate to high efficiencies at the basin level. Irrigators in the Bajío region of the Lerma Valley in Guanajuato, Mexico are well aware of the rise in water tables that follows crop irrigation. This water is reused by groundwater pumps towards the end of the dry season when water becomes extremely scarce. However, such linkages are usually managed at relatively small scales where the cause and effect of groundwater recharge is evident.

Improving the infiltration capacity of a basin improves the rate at which deep groundwater recharge occurs. However, it is extremely difficult to quantify the impacts of landuse management on groundwater recharge without field surveys and studies, or modeling by a geo-hydrologist or hydrologist. In addition, it is difficult to account for human use of deep ground water given the lack of groundwater regulations, and difficulty in monitoring or enforcement in most parts of the world.

Protection of Biodiversity

Natural flow regimes, which are critical to the maintenance of biodiversity at the landscape ecosystem level, support a number of direct and indirect values. These include maintenance of critical habitats for both freshwater and marine fisheries and wildlife that may be important for both subsistence and commercial purposes, shoreline stabilization which can reduce coastal storm damage, recreational and aesthetic values important for ecotourism, and biodiversity per se, which is linked to direct uses and may justify provision of funding from external sources based on global benefits that cannot be captured locally. . Components of the flow regime are the natural patterns of variation in the quantity and timing of the flow of a river, including natural disturbances associated with these flow patterns, and the interaction between basin climate, geology, topography, soils and vegetation, of which they are a product (Poff, Allan et al. 1997). For example, maintenance of wetlands, riparian habitats, mangroves, and coastal zones, which support many direct uses, may all rely on flood pulses and transport of sediment loads that follow regular patterns of variation .

It is difficult to ascertain the natural state of a stream prior to intensified human use of a watershed, a condition that may also be unattainable as a result of changes associated with human uses (Korte 1993). On the other hand, a fundamental characteristic of streams as well as entire fluvial geomorphic systems is that they change, not only in response to human uses, but also in response to both year to year fluctuations as well as to longer term patterns of variation (Graf 2001). Changes in natural flow regimes can impact the biology and geomorphology of a water body, leading indirectly to ecological adjustments (Whiting 2002). Resilience, which refers to thresholds within which these changes can be adjusted to, is therefore a key consideration but these thresholds tend to be highly uncertain. This implies the need for an integrated and adaptive approach to management, and to consider what is a feasible, desired and probable endpoint, given geographical and historical conditions and economic trade-offs, as well as to be explicit regarding risk and uncertainty.

Management of natural flow regimes would include activities discussed above aimed at sustaining baseflow, maintaining total yields to flush fine particles and maintain healthy in-stream sediment budgets, and maintaining water quality, but may also require a broader range of objectives to be considered in planning for river basin and infrastructure development patterns. Clearly, there are streams that are naturally dynamic resulting in natural losses (or gains) of the sediment bedload budget. A review by Whiting (2002) identifies the following factors to be considered in determining the quantity of water necessary for maintenance of ecosystem functions for individual basins:

- Flows needed to maintain sediment sizes and mobility or the substrate – alterations in substrate can affect the composition of biological communities who may also be smothered by finer sediment particles, as well as require sufficient water exchange for purposes of oxygenation and flushing of waste and finer particles;
- Flows needed to maintain the channel, for channel conveyance of water and to maintain upstream downstream connectivity– channels reflect the level of streamflow and tend to diminish in size when flows are reduced;
- Flows for habitat maintenance may include those for maintenance of channel features and bed structure and other special features such as sandbars and pools, maintenance and nourishment of the floodplain and riparian vegetation, and other features;
- Flow regimes also sustain the “hyporheic” zone, where surface and groundwater interact, which store and slowly release peakflows thereby sustaining baseflows – they also play important roles in water quality through the process of denitrification, which relies on maintaining the wetness regime, and through the release of cooler water important to lowering summer water temperatures.

In addition, the goal of maintaining suitability for flow dependent recreational uses, and for those that are enhanced by flow for its aesthetic values will also require consideration of appropriate flow levels.

Determining actual levels of flows to achieve these objectives is not an exact science and needs to be determined in the context of individual basins. Some general rules of thumb gleaned from this review are to establish flows as a percentage of streamflow and allow periodic highflows, at least at the bankfull level, which can help to maintain natural patterns of variation, flush fine particles, and maintain channel structure and continuity. Periodic floods may also be necessary to nourish and maintain wetlands and riparian areas. According to Whiting (2002), managing for a natural sediment budget in a stream, which supports stream integrity and natural habitats, requires the maintenance of natural bankfull flow levels (storm runoff levels with recurrence of 1.5 to 2 years) or at least 60-70% of these levels. The latter are levels estimated to effectively discharge fine sediment and to move heavier bedload. Periodic scouring by floods also helps to maintain channels by removing encroaching vegetation. Maintenance of upstream-downstream channel continuity will usually depend on both magnitude and timing of flow and also its velocity, quality and temperature and delivery of normal rates of sediment (Whiting 2002).

Key parameters that can be easily measured, used to characterize basin conditions and processes, and, over time, to provide indicators of change, outlined by Graf (2001) are:

- Width of the channel – this will reflect and respond to changes in discharge patterns – the bankfull discharge level is key to channel maintenance and stability;

- water discharge, - this is the most routinely measured and obtainable parameter, as it is an indicator of total water yield and tends to be used as a basis for water allocation;
- sinuosity – this refers to the ratio of the along-channel distance to the shortest straight-line distance (which ranges between 1.05 and >2), which reflects gradient, flow velocity, capacity for sediment transport, stream power, shear stress, and ultimately, channel stability
- changes in pattern – this will tend to reflect widespread adjustments in response to climatic and/or human influences

Other parameters, such as depth, flow velocity, and levels of sediment transport are difficult to measure directly because of high variability. However, changes in the size of sediment particles can generally indicate changes in material inputs and also in the flow regime (Graf 2001).

An important distinction is between headwater/hillslope and downstream fluvial-landscape interactions. In the former, landscape-riparian interactions are tightly coupled and are dominated by stochastic disturbances (e.g., landslides, debris flows, floods and droughts), and there is greater variability in discharge, creating a great diversity of biophysical conditions important in maintaining diversity of habitats (Gomi, Sidle et al. 2002). A key source of variation is in the distribution of the variation of soil moisture and saturation between the ridge that divides the drainage, and the narrow riparian areas alongside the stream, where important biogeochemical transformations occur that can impact water quality. Other key elements of heterogeneity are types of vegetation, which determines the exchange of water between soils, canopy and atmosphere, and the degree of soil permeability, which is altered through land use practices that range from deforestation and farming to the creation of impervious surfaces (Band, Brun et al. 2000).]

Hillslope processes can also lead to rapid changes in response to wet periods, and can create hazards. For example, roads can intercept flows of groundwater diverting it more rapidly to streams, contributing to higher peak flows or initiation of debris flows, or may act as dams that interrupt flows of debris (Nakamura, Swanson et al. 2000). Although unpaved roads occupy small areas of a basin in comparison with agricultural land areas, a recent study in Northern Thailand found their contribution to basin-wide runoff and stream sediment, to be of approximately the same magnitude (Ziegler, Giambelluca et al. In press). In other words, they play a key role in modifying flows and should be regarded as filters. Although it is beyond the scope of this report to fully discuss the hydrological and management implications of upland development patterns and how they might be addressed, it is important to identify such processes to the extent possible and consider them in planning for basin development activities, and to monitor potentially significant cumulative effects on the downstream fluvial processes and on the entire flow regime. Further discussion of these implications can be found in (Nakamura, Swanson et al. 2000), (Siegel 1996) and (Whiting 2002). In contrast, downstream fluvial processes are more continuous and are dominated by flood pulses and movement of the bedload.

Use of process models

Models are important tools for simulating hydrological and water quality processes that can be used to approximate the impacts of alternate forms of watershed management. More specifically, they can be used to identify expected hydrologic responses to proposed watershed management activities, so that they can then be quantified in economic terms. Uncertainty associated with weather variability may be simulated and represented by flow duration curves. However, models must be calibrated, validated and verified to represent the key hydrologic processes of interest. Unfortunately, critical review of this crucial step is often extremely poor.

In October 2000 at a conference in Frieburg, Germany, more than 80 researchers gathered to discuss state-of-the-art understanding of "Runoff Generation and Implications for River Basin Modeling". It was generally agreed that a proper representation of internal hydrological processes cannot be guaranteed by matching measured and simulated runoff data in modeling applications (Uhlenbrook, McDonnell et al. 2003). This is not surprising, given that statistical ratings of simulations are often biased towards matching extreme peak events because the favorite performance statistics are r^2 values that demonstrate whether there are correlations between the behavior of different variables. Problems also arise due to over-parameterized models that have developed to become 'black boxes'.

Model developers generally have a clear understanding of the key processes, data availability, and assumptions in their region of interest, which often is not the case when these models become popular and used in a different region. Popular models are often complex because they incorporate several modules and processes that are needed by many new users but these do not always take the time to follow all the subroutines, processes, and implications. For example, until recently, a popular model called SWAT produced runoff from snow covered surfaces that was akin to runoff from a concrete pavement. This was not a problem in Temple, Texas where it was developed and it rarely snows, yet it is often used in the North Eastern US. In effect, Yew et al. (1997) compared 6 models with varying numbers of parameters concluding that complexity was not necessarily a guarantee that key hydrologic processes will be adequately represented.

As a result some guidelines are suggested for modeling where data is limited and it is crucial to correctly interpret the underlying hydrologic processes.

1.1.1.3 Choice of model

Conceptual process models are a good bet as a first step in assessing and identifying the ranges of the key hydrologic processes and impacts of watershed management. These models are simple and lumped but offer the potential of hydrologic process understanding of key zones or reservoirs of catchment response (Seibert and McDonnell 2002). Seibert and McDonnell (2002) attributes a tendency away from fully-distributed, physically-based models to concerns with overparameterisation, parameter uncertainty, and model output uncertainty. In effect, a physically-based module (Green and Ampt infiltration module) for runoff estimation in SWAT was developed due to user demand but it is rare to find a modeler using the module in an application (Jeff Arnold one of SWAT developers, personal communiqué).

An advantage of using conceptual process models is that "soft data" from expert knowledge can be incorporated into the model structure whereby fuzzy ranges of expected values can be used to represent uncertain processes (Seibert and McDonnell 2002). Local knowledge (e.g. flood frequencies, time to peaks, duration of baseflow during the dry season, etc.) may be obtained from water resources practitioners in the region of interest, or local inhabitants. The disadvantage of using conceptual process models is that despite their simplicity (or perhaps as a result of it), they require expertise to use and to adapt as site-specific processes become understood.

"Generalized" and "operational" models (Wurbs 1995) are important to consider because they may be easier to use. Generalized models are those that have been developed to be used in systems of various configurations and locations, instead of having been developed for a particular problem and a specific site. Operational models are those, which are reasonably well documented and proven and can be used by many professionals and not only by those who developed the model. However, professionals should be aware of the "black box" syndrome and critically ensure that dominant hydrological processes are being simulated because these are often site specific. In addition, these models tend to require more parameter inputs, which must be assumed in data limited regions of the world, particularly if extensive field studies are not feasible.

1.1.1.4 Calibration

The following should be rules to live by given that, despite a decade of research into automated global search algorithms, manual calibration has not been replaced.

i) Multiple-criteria should be used in calibration to capture the key hydrological processes. That is, simulated data should represent as many possible known output variables (e.g. streamflow and groundwater levels). "Soft data" from expert knowledge might be incorporated as added criteria in the calibration process. In addition, single variables can be used to represent different processes (e.g. streamflow can be used as the first criteria, and changes in the rate of streamflow as a second criteria).

ii) Process and range of variability is more important than r^2, or finding a correlation between variables. In other words, it is far more important to correctly represent a key or dominant hydrological process than to have high performance statistics in the simulation. Borrowing a passage from Seibert and McDonnell (2002); *a better process representation of catchment hydrology […] should be "less right, for the right reason" than "right for the wrong reason"*.

iii) Keep in mind the objectives. Namely, if baseflow is the service of interest in a PWES scheme, the model should adequately represent the conceptual underlying process and might be calibrated using dry season flows.

iv) Consult the experts and make sure simulations are reasonable. The experts are regional/local water resources practitioners and local water users with direct and extended experience with the resource. Workshops and focus groups are insightful ways of obtaining local knowledge and "soft data".

1.1.1.5 The downward approach to modeling

A general rule is to use downward approach to modeling (large to small temporal scale) and to increase complexity of model as needed to explain phenomena (Farmer, Sivapalan et al. 2003). This is perhaps most applicable in the development of a site-specific conceptual process models but is also relevant for calibration, verification, and validation of any model.

i) Match inter-annual yields of a basin: That is, predicted annual water yields of a basin must be suitable for any given return period.
ii) Match intra-annual (monthly) yields: That is, predicted average monthly total streamflow is a suitable represention of seasonal variability.
iii) Match flow duration curves: That is, predict the probable distribution of streamflow in the catchment throughout the year. This enables an interpretation of expected flow given uncertainty, and ensures the hydraulic characteristic of the catchment is represented.
iv) Match the Time series: That is, the model must generate a suitable time series of predicted streamflow that matches observed streamflow. Generally, daily data fits poorly due to time lag issues, but these fits are improved by using larger time steps akin to weekly or monthly total streamflows. This is often the only step used for calibration/validation in many modeling approaches.

Ultimately, models must provide adequate expectations of landuse impacts on hydrology and water quality, which imply issues of scale[19]. In general, flows in large basins may seem unaffected by changes in land-use/land-cover that affects a small portion of the basin because of the larger persistence (storage) effects in such basins.. Likewise, it has been recognized for a long time that watershed management activities seem to have little effect in reducing sedimentation of downstream hydraulic structures (such as reservoirs) in large basins (Mahmood 1987) (Vaughan and Ardila 1993) (Basterrechea, Dourojeanni et al. 1996). Therefore, the effects of changes in land-use/land-cover must be examined on the basis of flow measurements of small rivers whose basins are subject to the changes in land-use/land-cover (WMO 1987).

Links between watershed processes and economic significance of impacts

As discussed in the introduction, ecosystem processes cannot be considered "services" unless they also have economic significance, directly or indirectly, which also implies actual access to benefits. For purposes of determining the economic significance of

[19] Kiersch, B (2000) suggests hydrology, including sediment loads, impacts are negligible above basins 3 orders of magnitude larger than the scale of management.

offsite impacts and benefits of good management practices, watershed process models should be able to provide an approximation of:

- the direction and magnitude of changes in parameters of interest , and
- the spatial and temporal scales at which these can be detected.

In other words, process models should provide a working hypothesis as to whether a particular change in land use is expected to increase or decrease runoff and sedimentation, by how much, and over what distance from the site, both in space and in time. These model results can then be used to establish rough equivalencies between the extent of changes in land use and changes in service provision, which provides a basis for amounts to be exchanged in PWES. Process models should also provide the basis for identifying a set of indicators of change in key processes, and a baseline for a consistent set of measurements of these over time, which can be used for purposes of verification and adjustment.

The direction of impacts can be expressed in terms of relationships between land use and hydrological outputs in relation to endpoints of interest. Key outputs, as discussed above are: sediment yield, annual water yield, peakflow, dry season baseflow, recharge of groundwater in deep aquifers, and maintenance of biodiversity. A review of the literature (Aylward 2002) generally confirms that land use change, especially the loss of forest cover, results in:

- Increases in sediment yield as well as the flow of chemicals and nutrients;
- Increases in water yield and peak flows;
- *Either increases or decreases* in dry season baseflow and also in groundwater recharge, depending on the outcome of interactions among site specific processes that determine the net effect of changes in evapotranspiration and infiltration.

Land use changes also contribute to overall changes in the natural flow regime (Graf 2001).

Whether or not any of these changes are of economic significance will depend on their links to welfare, and the values placed on them, as discussed in the next section. These will in turn depend on opportunity costs under existing land uses as well as stakeholder perception of threats and vulnerability to them, as well as assurance of access. For example, floods are extremes in what is largely a natural process of variation in flow which can benefit human welfare in numerous ways. However, flooding can also be aggravated through land use practices, and it can create costs when there is development in floodplains. In a study in the Arenal watershed in Costa Rica (Aylward and Echeverria 2001) ranching was found to produce higher net present values than was offered for reforestation. Further, the expected decline in water yield associated with reforestation was the dominant factor in the economic analysis because the higher annual water yield was of direct benefit to a downstream hydroelectric facility. Given that impacts of land-use change may be both positive and negative, depending on what is valued and measured and on inherent trade-offs, it is important to consider their full range, as well as their relative magnitude or significance (Aylward 2002).

As a general rule, land use impacts on flows of water and sediment are best examined and addressed at the level of individual hill slopes and patches, which are the source of significant landscape and land use heterogeneity that affects flow routes.In contrast, water quality and water diversions can be detected and impacts felt at basin scales, at which they are more appropriately addressed so as to permit consideration of trade-offs among all affected stakeholders. Water diversions and infrastructure such as dams and reservoirs also enable agricultural and urban development, that lead to a greater magnitude of land use changes. Basin scales also permit consideration of cumulative impacts of numerous and otherwise insignificant small- scale changes, and the influence of larger scale climatic factors on total water quantity and on extreme events that are a dominant factor in watershed processes.

An estimate of equivalencies between the degree of changes in watershed processes and service provision provides a rationale for levels of payments to be made in exchange for maintenance of particular types of land cover or for specific land management practices. For example, to be able to trade permits between point and non-point sources of nutrients, for purposes of reducing input of damaging levels of nutrients to streams,it is necessary to first identify the amount of forested area and/or specific conservation land-use practices needed to offset emissions from a particular point source. Contractual arrangements in which payments are made for specific management practices require an approximation of the relationship between these practices and nutrient inputs to water bodies. However, given the impossibility of obtaining complete information and unequivocally establishing links between multiple causes and effects typically found in a watershed, it may be necessary to base equivalencies on the identification of the relative values of various land areas and management actions for achieving desired outcomes. For example, steeper slopes would be of higher priority for practices that reduce erosion and sedimentation, regardless of the precise amount by which these practices reduce erosion. In the Arenal case study, fragmented primary cloud forest areas were found to have the greatest value for increasing dry season flows, compared with unfragmented primary cloud forests, secondary regrowth forests of both types, and open pasture areas (Aylward and Echeverria 2001).

Establishment of priorities in terms of relative values will require consideration of multiple management objectives, ecosystem functions that support them , a way to prioritize actions in terms of their relative contributions to achieving various objectivesand conflicts among them. – Payments may be one way of resolving the latter. Geographic Information Systems (GIS) can provide a useful tool for organizing and presenting this information in a way that makes all of the considered factors, options, and trade-offs transparent to stakeholders, thereby allowing them to participate more effectively in negotiation regarding the development of equitable arrangements.

Basic steps in organizing information are:

- Classification of individual land tracts into units that reflect similar biophysical characteristics and processes of interest, by types of ownership, and by land use;
- Identification of feasible land use and management options and opportunity costs, including business as usual, for different types of holdings;

- Identification of decision criteria that reflect existing policies and concerns expressed by stakeholders, e.g., equitable distribution of costs, benefits and risks;

- Ranking of options by each criterion, noting the degree of uncertainty and best judgment – rank may be indicated with actual monetary figures where these exist, other numerical values appropriate to the criteria, or qualitatively (e.g., high, moderate, low…).
- Identification of overlaps among areas of high priority under different criteria, e.g., small holders who have low opportunity costs, located on steep slopes in fragmented cloud forest areas;
- Identification of conflicts, e.g., high priority conservation areas in which opportunity costs are also high – these may require more stakeholder negotiation and special consideration of what is required to resolve particular conflicts;

This framework is only intended to provide a way to examine a complex problem and to identify trade-offs among multiple and conflicting objectives. Decisions will ultimately depend on what is required to justify expenditures in the particular case (Laurans 2001). For example, justification of a public expenditure may require that the amount saved (e.g., estimate of avoided flood damages as a result of wetland conservation) exceed the expenditure, for which technical criteria and rough estimates may be sufficient regardless of whether there are other environmental benefits. This latter expenditure includes all costs – including information – gathering, monitoring, etc. – of administering PWES. Greater precision may be required when different options have benefits that are close to costs.

For purposes of developing a PWES, what costs and benefits are formally considered in economic analysis and in political decisions will generally depend on recognized rights to ecosystem services and responsibilities for providing them. For example, prices paid to landowners for specific land use and management practices, will depend on what they are legally required to do, regardless of whether they are compensated. Payments to polluters as a way to avoid pollution are likely to be rejected as unfair and also unnecessary when it is against the law to begin with to emit particular pollutants, providing these can be enforced. Rights to specific watershed services may remain to be defined in the context of stakeholder negotiations over payment arrangements. Stakeholders will also generally want assurance that actions are effective and that they will have access to future benefits. Making uncertainties explicit is critical to the management of expectations.

Given high opportunity costs that may need to be compensated it will generally be less costly to maintain existing services, than to restore those that have been degraded. In addition to having a higher opportunity cost, restoration usually takes a long time, in which success may be highly uncertain. In another example, the La Esperanza hydroelectric facility in Costa Rica deemed it worthwhile to pay for protection of existing upland forest area, to avoid the uncertainty that would accompany land use change, regardless of actual impacts. Unlike the facility in the Arenal case, the La Esperanza facility has higher dependence on dry season rather than on total flows, in that it has less reservoir capacity, and also less dead storage capacity in the reservoir for sediment (Rojas and Aylward 2002). In the Arenal case, sediment trapped in the dead storage area of the reservoir was also of some economic benefit, in that this made more water available for hydroelectric production (Aylward and Echeverria 2001).

In addition to existing land uses, stakeholder vulnerabilities will also depend on threats to the continued provision of services and on how impacts and benefits are distributed among them. An analysis of the various kinds of assets that are used to sustain livelihoods, and also their links to the specific services, provides a way to identify impacts that need to be considered in decision-making from the perspective of stakeholders (Ashley and Carney 1999). Some key considerations in determining vulnerability are land and water use by various types of users and economic sectors, and the interaction of the hydrological cycle with water resource systems. In this kind of analysis, it is important to identify how land uses and water needs may be differentiated among subgroups of the population and by gender, as well as to recognize traditional rights or social norms regarding access to it. Dry seasons, when water is scarce, and periods of extreme events, i.e., floods and drought, will generally be more revealing of vulnerabilities and conflicts than "normal" periods.

Vulnerability to impacts of land use change on WES depends on the socio-economic and demographic characteristics of the basin, as well as on changes in the biophysical environment, and can help to determine economic significance of those impacts. For example, paving can reduce groundwater recharge and lead to increased runoff and localized flooding, as well as reduce dry season flows. However, vulnerability will also depend on development in flood-prone areas and the degree of dependence on groundwater over the relevant time period. It will also depend on the various kinds of social and economic disparities that exist, such as in various kinds of rights to land and water - that also create disparities in the ability to cope with and respond to changes in the water supply., It is important not to base vulnerability analysis on labeling and stereotypes of social groups, as it depends largely on development and social organization that is site specific, as well as on perceptions that people have of the options available to them (Handmer 2002) . An inquiry into stakeholders' own perceptions of their interests, problems and opportunities is important for understanding the local context, what resources are available to stakeholders, and values of ecosystem services that might otherwise be overlooked. Some key questions used to facilitate stakeholder discussions for an upland catchment forum in Thailand, drawn from a "soft systems" approach, (Attwater 1997) (Checkland and Scholes 1990; Checkland 1995) were:

1. "What management [actions are] needed and who would be responsible?"
2. "What inputs such as labor, information, funds are needed, and from whom?"
3. "What outputs would these [activities] generate, and for whom?"

An important distinction is between vulnerabilities at local and small, sub-basin scales from those at a large scale, which are sometimes in conflict. Some specific sources of vulnerability can be grouped into those associated with livelihoods and those associated with infrastructure:

- Livelihood vulnerabilities:
    - Malnutrition, famine and economic hardship as a result of insufficient water for agriculture.

    - Illness as a result of the lack of safe drinking water
    - Loss of habitat that sustains wildlife, fisheries, rice cultivation areas, and also loss of floodplain fertilization, all of which may be important in sustaining rural economies,
    - As services become scarce, development options are narrowed and the poor may be further excluded because they have less bargaining and political power;
    - Coastal erosion and floodplain development increase vulnerability to extreme events;

- Infrastructure vulnerabilities:
    - Water Resources Infrastructure for urban provision and sanitation – the purpose of water resources infrastructure is to reduce vulnerability to variations in the flow regime and enable agricultural and urban development, e.g., reservoirs provide buffers against low flow periods. However, this resulting increase in the concentration of population that is then dependent on infrastructure, is more vulnerable in the event of their failure, as a result of catastrophic events such as the failure of a dam, higher than anticipated sedimentation rates that reduce the life of a reservoir, or larger than anticipated changes in the flow regime as a result of climatic changes. Important considerations are: reservoir capacity, population served, water uses provided for, ability to pay for watershed protection, risk of exclusion of those who are unable to pay, whether or not there are other options as demand increases, e.g., for additional reservoirs.
    - Hydropower and irrigation reservoirs – vulnerability depends on reservoir capacity, which determines the extent to which they are dependent on dry season flows; and also the capacity for storage of increased sediment loads – as discussed in the section on myths, if reservoirs are built in areas of naturally high sedimentation, changing land uses won't help.
    - Irrigation – vulnerability to lower dry season flows unless supplied by a reservoir.
    - Navigation – vulnerability to siltation of channels and low flows.

The cost of a PWES scheme may appear prohibitive in light of the kinds of information potentially required. However, the main objective in the vulnerability analysis outlined above is to inform the identification of options and may not require exhaustive information for all of the mentioned categories. Similarly, the need for precise and detailed information on impacts of land use change may only be necessary for cases in which there are conflicts. In most instances, rough estimates of magnitude will be sufficient.

Value of watershed ecosystem services

The main focus of this report is on the scientific aspects of defining ecosystem services. Therefore, this section on valuation is limited to a discussion of the special implications of biophysical aspects of WES for determining values and how these are linked.

Economic value is typically associated with demand for a good or service, as indicated by Willingness-To-Pay for it. As a rule, WES have characteristics of public goods, i.e., that it is difficult and/or expensive to limit benefits to those who pay for them.At the same time, there is often rivalry in consumption of these services, which is the defining characteristic of common pool resources. In addition, WTP depends on stakeholder confidence in the effectiveness of proposed management actions needed to insure that the service is delivered and that they will have access to future benefits. This confidence depends not only on underlying ecosystem processes, but also on the effectiveness of institutional arrangements needed to insure continued provision. In other words, value depends as much on effective governance and institutional development as on determining supply and demand.

A key question for selecting appropriate arrangements is the extent to which Willingness To Pay (WTP) for the tangible aspects of WES is sufficient to justify the added cost of conservation actions when they are compared with opportunity costs of foregone land uses. These refer to services that have more direct use values, that are more likely to motivate local action and that can be somehow captured in market transactions. Examples would include water for direct consumption of water, flows needed to support ecotourism and recreational uses, differences in property values that can be attributed to aesthetics or, insurance against potential damages. In the absence of market values or monetary ability to pay, WTP can also be expressed through other kinds of trade-offs that stakeholders are willing to make to protect values that are threatened (e.g., providing labor, and participation in various forms of collective action), that are associated with places and with ways of life that depend on WES. In general, it is not until services become scarce or threatened and difficult trade-offs are faced that their value is even considered (O'Connor 2000). Less tangible values, such as those associated with non-use values of biodiversity, tend to rely on policy measures and on external funding sources, e.g., NGOs, governments and multilateral donors. Watershed functions may also benefit from complementary values, such as maintenance of existing forested areas for purposes of carbon storage.

Whether WTP is sufficient to create an incentive for land users to adopt proposed management actions will depend on what land users are Willing to Accept (WTA), which depends on returns to existing land uses and opportunity costs of those forgone. It will also depend on whether land users have rights that give them authority for land use decisions and enable them to accept payments.

Whether or not costs are justified will also depend on the significance of the intervention. Given the large size of upper watershed areas, protection of watershed services generally implies the need for intervention on a large scale, which requires collective action. However, even if all farmers in an upper basin were to participate in adoption of practices designed to reduce erosion, this may still not have a significant impact downstream if it occurs in an arid region where erosion is naturally high. Data from process models should permit the development of more site-specific rules of thumb to guide decisions regarding the minimum areas over which interventions need to occur if meaningful results are to be obtained.

Valuation of WES ultimately implies the consideration of trade-offs among multiple uses, interests and objectives, so as to inform a process of conflict resolution and negotiation among stakeholders regarding equitable PWES arrangements. It should also provide stakeholders with an opportunity to reconsider their values and priorities in light of new information and to reconcile conflicting objectives.

Institutional considerations

As discussed above, WTP for WES is also linked, inextricably, to the effectiveness of institutional arrangements needed to insure access to benefits by those who pay the costs of their provision, absent which value is no more than hypothetical, as it cannot be captured. Consequently, there would be no incentive for provision.

Payment arrangements need to be considered in the context of a global trend of institutional changes in water resource management, brought about by a general increase in water scarcity and diminished provision of watershed ecosystem services. Among the key element of these changes are efforts to improve recovery of costs, both for operations and maintenance of facilities – which would increase the capacity of governments to deliver basic water supplies and sanitation, and to cover the cost of conservation management and research activities – which would protect the provisioning capacity of ecosystems (Saleth and Dinar 1999). Recovering the costs of conservation will be particularly challenging where users are accustomed to high water subsidies and pay only a fraction of the costs of operations and maintenance. Therefore, it may require a long term strategy, absent broader macroeconomic reforms that tend to be associated with crises or sweeping political changes, such as the end of apartheid in South Africa.

If payment arrangements are intended to support provision of ecosystem services, a second contextual issue will be the need to account for and cap water uses, and establish a percentage of flow that is to be allocated to the maintenance of ecosystems and other designated priorities such as meeting human needs. Institutional approaches used for this include a system of registering water uses and licensing of streamflow reduction activities being implemented in South Africa (DWAF 1999), and caps on total water consumption for human uses as is being done in the Murray-Darling basin in Australia (Pigram 2000).

A key institutional arrangement with implications for cost recovery is that of property rights, which play a key role in economic incentives because they control access to benefits and also define responsibilities for actions needed to insure their provision. For example, absent clear land title, upper watershed land users will lack the authority to enter into contractual agreements and therefore be unable to benefit from payments. They may also risk eviction as values are placed on services to which they lack recognized rights (Landell-Mills and Porras 2002). Property rights may take different forms, ranging from informal rights or norms recognized by users, to various forms of formally recognized public and private ownership by individuals, groups or government entities. Failure to control access is often mistakenly referred to as a "common property" situation but is actually an “open access” situation in which no property rights are in effect (Ostrom, Gardner et al. 1994).

Appropriateness of property regimes depends on whether their inherent incentives are consistent with social objectives, as well as on the biophysical characteristics of the resource. For example, rights to water based on historic use or “prior appropriation”, which usually require that the water be used in ways that are considered socially beneficial, were consistent with the objective of promoting development in the western United States during the 1800s. However, it creates a disincentive for reducing consumption as this would lead to a reduction of the amount of water a user may claim in the future, and is inconsistent with uses associated with emerging social objectives of conservation, such as instream flow, that are not legally defined as “beneficial” (Wilkinson 1992). Rights to water based on possession of adjacent land or “riparian rights,” allows reasonable use that does not interfere with the reasonable use by others, and may allow communities to control access and exercise customary rights. The latter however may limit the ability to transfer water and thus to develop water markets, which can provide incentives for more efficient allocation among various uses (Meinzen-Dick and Bruns 2000). Riparian rights also exclude those who do not own land. In an open access situation, the incentive is simply to consume resources before someone else does.

Special biophysical characteristics of watersheds that have implications for the nature of property rights to watershed services are the separation of costs and benefits between upstream and downstream, and the vast size and remoteness of upper watershed areas. This results in time lags between causes and effects of watershed degradation that make it difficult to link specific management actions to outcomes. Disproportionate shares of erosion often come from areas such as forest margins, roads, footpaths, steep hillsides, gullies at the base of escarpments, and river banks. These tend to be de facto open access because there is little incentive to invest in their improvement or in controlling access (Swallow, Garrity et al. 2001). This implies the need for stakeholder collaboration to reduce transaction costs. and the formation of local watershed organizations

In addition, if significant stakeholders are disadvantaged and regard existing rights as inequitable, there will be little incentive to cooperate in their enforcement. Some studies have found differences in WTP that depended on the protection mechanism suggested and on the distribution of property rights. In some cases, stakeholders are unwilling to pay not because they are unaware of ecosystem values, but simply because they do not feel that it should be their responsibility to do so (O'Connor 2000). Direct payments for environmental services raise fundamental questions of who *should* pay and how much, and the extent to which providing these services should simply be regarded as an obligation inherent in the responsibility not to harm others. In some cases, transfer payments to upstream areas could be seen as violating the principle of “polluter pays”, unless accompanied by sanctions on pollution (UN FAO 2002). However, given the situation of the rural poor, often found in upper watershed areas, and low prices paid for agricultural commodities, direct payments for providing services of maintaining the landscape and water quality may also be regarded simply as recognition of the value of environmental services.

Key questions for assessment are to determine the incentives inherent in existing and proposed property regimes and their implications for the delivery of watershed services, to identify stakeholders who are advantaged or disadvantaged by them, and whether they are regarded as equitable. Appropriate or not, property rights do not change easily or quickly, absent political momentum generated by events such as the end of the cold war, as their purpose is to provide some security without which there is little incentive for investment. Thus, they cannot be arbitrarily changed. However, they do tend to change over time to reflect changes in social values, as new problems emerge, and as technological improvements bring down the transaction costs of controlling access to particular resources – and are not always compensated. For example, development of hydropower at the beginning of the industrial era led to a change in rights to the natural flow of water because it was considered to be of greater value to society, and continues to lead to widespread displacement of communities. Similarly, as a consequence of the growth of urban areas, rivers became more highly valued for sewage disposal than for fisheries and recreational values.

Just as changes in rights have been implicit in the development of physical infrastructure (hydropower, dams, irrigation and navigation), the rise of values placed on freshwater services implies the negotiation and definition of new rights and responsibilities in which uses of land and water are limited to those that do not impair ecosystem functions that support valued services (Sax 1993). Typically, different claims and sources of authority will tend to overlap and conflict in a process referred to as "legal pluralism" (Meinzen-Dick and Pradhan 2002). Therefore, changes in property rights tend to come about through a contested process that can lead to institutional development, and which is itself an important response to the diminished provision of freshwater services.

Any initiative to protect downstream water supplies or biodiversity either by providing compensation to upstream landowners for altering land use practices, or by attempting to hold them responsible for damages, in effect involves negotiating new and appropriate forms of property rights, that resolve conflicts between these objectives and existing practices. The definition of new forms of property rights may also be made economically feasible through scientific and technological improvements (North 1990), in this case, in mapping and communication, which can reduce the transaction costs associated with assessment, negotiation of agreements, and their enforcement among numerous stakeholders.

Property rights are also a critical consideration when an inherent goal of a PWES initiative is poverty alleviation. In cases in which upstream land users lack some form of tenure security, payment arrangements to owners may lead to displacement of users to ever more marginal land areas, as a result of increased values. In some cases, rather than offer payment arrangements, upstream land users lacking any rights have simply been scapegoated for problems to which they may have only marginally if at all contributed, by downstream stakeholders with greater political power. Similarly, water use fees may exclude the poor downstream. This issue is beginning to be addressed through recognition of water as a fundamental human right (reference UN). South Africa has addressed this in their new water law that designates a certain level of water for ecosystem and subsistence purposes before allocation for other uses for which water use licensing is required.

Another key aspect of institutional arrangements is in forms of governance and decision-making processes. Development of water resource infrastructure such as dams and hydropower has also been associated with and supported highly centralized authority for water resource management and is largely driven by geopolitical considerations in which local stakeholders have little in any voice. Because of environmental heterogeneity, highly centralized authorities tend to have a limited capacity to respond to livelihood concerns and to support the provision of watershed services at local levels.

The site-specific characteristics and variability of freshwater ecosystems and other natural resources implies the need for detailed local knowledge, discretionary powers and also greater representation and accountability. These can further an adaptive approach to management because they increase the capacity to respond to factors such as variations in rainfall and crises associated with extreme events, and to mediate conflicts. Transfer of rights can also provide a stream of revenue to local governments that can be used to build and sustain capacity for resource management. Provision of watershed and other ecosystem services may therefore also be inextricably linked with efforts to and to achieve democratic forms of decentralization, or to "pry open… local democratic space" (Kaimowitz and Ribot 2002).

Institutional arrangements also refer to relationships established among buyers, sellers, and intermediary organizations often created to reduce transaction costs. As discussed above, these are costs associated with negotiations, that tend to be significant in a watershed context in which there are numerous buyers and sellers, and the need for monitoring and enforcement extensive and often isolated areas. Recognition or establishment of property rights also has significant costs. Finding ways to reduce these costs is often key to the practicality of developing effective PWES arrangements, in which these kinds of costs can easily exceed gains.

However, impracticality of PWES today, in a particular setting, does not mean that this approach to watershed conservation will remain so forever. In their study of long-term institutional trends, North and Thomas (1973) contend that property rights, of the sort needed for efficient development of natural resources, have spread over time for two reasons. First, resources have grown scarcer, due to demographic and economic expansion. Second and in response to mounting resource scarcity, the technology and institutional arrangements for specifying and enforcing property rights have improved, thereby bringing down the costs of specification and enforcement. Applying this view of institutional change, one anticipates that, as the value of WES rises and as better ways are found to institutionalize watershed payments, PWES will become more widespread.

It should be kept in mind that this is no different from costs that have been and continue to be incurred in the development and maintenance of institutions that support existing markets, that are generally not paid for in the prices of private goods and services, all of which can have spin-off benefits. Potential spin-off benefits associated with markets for watershed services include: clarification of property rights, stakeholder cooperation in other areas important to livelihood as a result of strengthened institutions, technological transfer and skill development, development of market infrastructure, contributions towards the protection of other ecosystem services not traded in markets, improved scientific understanding and environmental education (Landell-Mills and Porras 2002).

Payment arrangements for freshwater and other ecosystem services is therefore a long term process of institutional development that needs to be considered in the context of broader issues of democratic governance. For markets to work, democratic institutions and equity are essential because there needs to be trust that people will obey rules and abide by agreements made, which may not occur unless arrangements are regarded as fair (Lipton 1985). This is a continuing challenge in developed and developing countries alike.

Some Rules of Thumb (RoT) for developing effective PWES Initiatives

This guide emphasizes the need for site-specific assessment as a basis for choosing effective management actions that are effective as well as locally appropriate. Although standard recipes are to be avoided, it is possible to provide some general rules of thumb. These rules address the options that might be considered under different kinds of conditions, as well as expected consequences of specific land use changes. In other words, given a particular combination of biogeophysical conditions, economic uses of the environment, and existing infrastructure, what hydrological services will be limiting factors and what are the approaches that are available to internalize externalities? After the question is addressed, it is critical to compare expectations with actual conditions and determine whether more detailed assessment is in order, to understand differences. Several more technical and detailed RoT are identified throughout the text, in sections to which they pertain. This concluding section presents selected RoT as general guidelines, with key elements of the assessment framework.

General Guidelines

- Don't confuse trees with the forests, soil, and land use practices that follow the clearing of trees. As noted by Bruinjzeel (1990) "adverse environmental conditions so often observed following deforestation in the humid tropics are not so much the result of deforestation per se but rather of poor land use practices after clearing the forest." Therefore, care should be taken when extrapolating from the wealth of studies on hydrological impacts of logging in temperate watersheds to the encroachment of tropical forests by the agricultural frontier. The former usually has the equivalent of leaving the deforested area as fallow, whereas the latter results in rain fed agricultural production or animal husbandry, incorporating slash and burn techniques, and hastily developed roads.
- In the absence of hard data, rapid assessments, soft knowledge, and rough calculations can be used initially and improved over time.
- Given inherent uncertainties, monitoring is an essential component of PWES initiatives. The purpose of this monitoring is to collect data needed to determine if these initiatives are effective.
- When payments are linked to compliance, monitoring should be relatively straightforward, because participants will have a vested interest in compliance (Pagiola and Platais, 2002).

- Manage stakeholder expectations by being explicit about uncertainty, and seeking agreement on indicators of effectiveness and standards for the quality of information.
- Account for human uses of water so that the role of these uses in the hydrological cycle can be distinguished from natural variation and impacts of land use practices. This accounting should make it possible to distinguish natural variation and impacts of land use practices from economic scarcity, whish. has implications for the kinds of measures taken.. Economic scarcity implies the need for changes in allocation of water among human uses. Natural variation may imply the need for changes in allocation between ecosystems and human uses through caps on all uses combined. Impacts of land use practices may imply the need to change management practices.
- Estimate the components of the water balance and how specific variations affect valued services. This assessment serves as a point of departure for identifying total flows available for allocation, and areas where watershed management practices can be most effective. These components are: Storage (S), Precipitation (P), Streamflow (Q), Actual Evapotranspiration (AET), and loss to deep groundwater aquifers (G). Table 9 summarizes data needs and special considerations associated with estimating components of the water balance. For more information about indicators see FAO report on "Monitoring and Evaluation of watershed management project achievements" by Hernández and Vélez (1995)

Table 9 Data Needs for Estimating Components of the Water Balance

Component	Data Needs / Tools	Comments
Precipitation *(key component that is highly variable but relatively easy to measure)* *-daily rainfall per unit*	Use Thiesen Polygons to extrapolate data from network of nearby catchments/basins.	Raingage network used should surround catchment of interest.
	Install at a minimum 1 gage. Ideally, one per 100-250 km^2 for mountanous regions, and one per 600-900 km^2 for flat regions.	This is World Meteorological Organization standard. However, most important to consider are the resources to monitor and maintain network, and local conditions.
Irrigated crop transpiration *-Hectares irrigated* *-Crop water demand*	Ground survey, irrigation schedules and rural assessment.	Crop types and rotation is often hard to ascertain remotely. Often biggest differences in water loss from actual evapotranspiration depends on the season a crop is grown rather than crop type (Seckler, 1996)
	Remote sensing use of LANDSAT-TM or NOAA-AVHRR imagery during the dry season.	(a.) Normalized Difference Vegetation Index (NDVI) is an indicator that uses near infrared and visible light spectrum to estimate vegetation signatures remotely. (b.) In arid to semi-arid regions bare soil reflectance tends to skew estimates resulting in that other spectral indicators be used such as Soil Adjusted Vegetation Index, SAVI (Huete, 1988). Many other algorithms exist and can be used.

	Periodic measurement of canal flow using flow impellor, weir or stage measurements.	Provides estimates of irrigation efficiencies, which cannot be determined by remote sensing. However, tightly rotated irrigation schedules with few irrigators breaking rules can denote higher irrigation efficiencies.
Municipal & industrial uses *-expected water use throughout year*	Census records, permits, direct measurement, interviews. (water quality factors discussed below).	(a.) Water use per capita will depend largely on the water supply arrangement, i.e. is it gauged or not? (b.) Estimate amounts of water that are recaptured into the system with adequate water quality.
Characterization of landscape. *-Percents of generalized landuse, i.e. % forested area, % shrub, % water bodies*	Remote sensing use of LANDSAT-TM and NOAA-AVHRR with "ground verification". However, LIDAR is remote technology for determining Forest strands and density because it provides indication of height.	(a.) Care should be taken for using correct spectral indices suited for arid and wet conditions (see crop transpiration comments). (b.) combination of imagery from wet and dry periods can provide insight on wetlands and riparian vegetation, which might be sources of high transpiration during the dry season.
	Ground survey, Rapid rural appraisals, local knowledge on current and past landscape.	For example, remote imagery may have difficulty discerning between shade coffee and forest, or tall grasses and wheat.
Evaporation from free water surface *-water loss*	Class-A Pan (1.22m in diameter, 24.4 cm in height) Evaporation water losses multiplied by 0.7	If all stream inflows, outflows, and estimated ET can be accounted for, significant differences in the budget may be due to ground water contributions.
Basin-wide actual evapotranspiration *-water loss*	Difficult to measure directly. Use indirect methods described in main text.	
Streamflow *-average water depth*	Streamflow gauge (streamflow gages or impellors are expensive) or weir measurements.	At a minimum flow should be measured daily during the rainy season. Weirs are usually not suitable for large rivers unless an over spilling dam already exists. During the dry season flow change is more gradual and measurements may be taken with less frequency unless it rains.
	Stage measurement by visual readings from graduated indicators or automatically by pressure transducer (this latter may be expensive and tends to need continual technical maintenance).	Stage measurements (height of water level) should be done initially to correlate with flow and derive stage-flow curves. Once suitable stage-flow curves it becomes easier to monitor streamflow by stage observations.

	Peak flow measurement by: -pressure transducer (expensive) -transparent plastic tubing with Styrofoam ball. -observations of leaf litter on stream bank.	Measurements of the highest water level during a storm may be important for storm runoff management and erosion.
	Bankfull cross-sectional area	In alluvial valleys, streamflow at bankfull is often well correlated with a 1-3 year return period storm.
	Hydraulic relationships	Manning's relationship can be used to estimate streamflow using water level depth when direct measurements are non-existant.
Water Quality / Stream degradation	Storm Runoff Sources *-Areas contributing to storm runoff* *- Ground surveys to characterize gullies, landslides, fields needing continual drainage, wet-spots near streams where agriculture/livestock or industry exist.* *-Shallow wells to monitor water table level around stream especially in riparian areas that might be a source of contaminated storm runoff.*	(a.) For example tracking source of gullies (roads vs. clear cut area). (b.) Shallow wells are usually not feasible unless a committed research institution exists.
	Biological survey of indicator species. Health care records.	Algae blooms or lack of sensitive species that existed prior indicate low water quality. Although not an objective measure it can help target more objective studies.
	Water quality sampling of constituents or sediment. *e.g. concentrations of sediment, dissolved oxygen, E.Coli, etc..*	Requires water quality laboratory or field probes. These are expensive but most objective ways of getting data. If storm runoff is source of water impairment constituent samples should be taken at least during several storm events a year, particularly during he rising limb of hydrograph. If source of contaminant is baseflow samples may be taken a couple of times a month.
	Stream sediment budget accounting for cumulative watershed effects	(a.) One less objective approach "*involves sampling a longitudinal reach of stream channel several hundred feet long, using a zig-zag pebble count procedure that crosses all habitat features within a stream channel*" (Bevenger and King, 1995). (b.) a more objective approach for determining sources of sediment is determining cesium-137 from fallout in particles but it is extremely expensive. (c.) Bathymetry is used to determine sedimentation of reservoirs.

Groundwater	Monitoring of deep aquifer and shallow aquifer wells	These are usually expensive and unpractical propositions unless several wells already exist. The existence of pumping wells (create draw-downs in water table) and poor groundwater use regulation will further limit this approach. Usually the most appropriate way is to approximate deep groundwater gains or losses to the system by making best possible water balance and accounting.
	Shallow aquifer contribution to stream by indirect methods	(a.) Baseflow separation techniques. (b.) analysis to determine hydraulic characteristics of streamflow recession (e.g. Brutsaert and Lopez, 1998)

- The range of variation is more important than average values, as extreme events and conditions have the greatest influence on watershed processes.
- Even if total precipitation gains in forested areas are insignificant compared with pasture, gains during the dry season may be significant, particularly in cloud forests and at the community scale. The magnitude of dry-season gains can have major implications for Willingness-To-Pay.
- It is insufficient to model streamflow without ensuring that the hydrological processes are being adequately represented.
- Flow probability duration curves (Fig. 1) from adequately calibrated models can be used to provide expectations of watershed characteristic flow and impact/opportunity cost of alternate landscape.
- Departures from curves of expected duration of flow can provide indicators of particular processes or of management impacts.
- Whether or not increases in the infiltration of water in forested area also increases streamflow will depend on whether or not it is available for ET, i.e., is within the reach of roots of trees and other vegetation.
- AET is a principal component of the water balance, and a key source of uncertainty because it is a function of numerous variables including climatic factors, vegetation, and land use - a major obstacle to the reliable estimation of AET is the difficulty of obtaining site-specific land cover and land use data that reflect significant heterogeneities generally found in a landscape.
- Impacts on AET from changes in forestry management (i.e. deforestation or aforestation) are likely to be greater than changes in fog drip or interception (Keppeler, 1998).
- Minimal areas of change in land cover are necessary before changes in streamflowcan be detected by monitoring. In general:

- Changes in streamflow are not detected in <u>temperate</u> forest when there is less than 20% change in landcover (Review of 94 temperate catchment studies by Bosch and Hewlett, 1982).
  - Changes in streamflow are not detected in <u>tropical</u> forest when there is less than 15% change in landcover, and average yields increase 50mm for every 10% reduction in forest cover (Review of 23 tropical catchment studies by Oyebande, 1988).
- There are general 'rules-of-thumb' expectations as to the extent of change in streamflow as a result of change in landcover, that vary by species (studies in Temperate humid catchments reviewed by Bosch and Hewlett (1982);
  - Conifers and Eucalyptus result on average in an increase annual yield of 40mm per 10% change in cover.
  - Decidous hardwood result on average in an increase in annual yield of 25mm per 10% in change in cover.
  - Brush and Grass result on average in an increase in annual yield of 10mm per 10% change in cover.
- Changes in baseflow are more site-specific (Calder, 1988) but clues about sustained flow capacity can be obtained by comparing normalized flow duration curves (**Figure 6**) of catchments with similar indeces of dryness. In general, landuse management changes are likely to have greater impacts on baseflow if soils are deep (i.e. deep roots can have a competitive advantage over shallow root systems). If shallow fractured bedrock exists, thereby limiting access to deep roots, increases in soil permeability are likely to improve dry season flow.
- Agroforestry can increase the available water capacities of the soil and permeability when low density tree networks can be established because the root network can protect soil, while lower densities avoid excessive losses by ET.
- Slope and topography of the landscape provide zones that dissipate (and allow sediment to settle) or accentuate runoff momentum. Management of the landscape should be consistent with catchment's goals, i.e. pasture cattle on steep hillsides is likely to be a bad idea because cattle create an intricate networks of trails but may have much less impact on a gentler slope. Similarly, cattle pastures in a zone that has a high topographic index (perhaps the gentler slope from the above example) will more likely threaten water quality.
- Soils in montane cloud forests are often extremely porous and permeable, and easily eroded when deep-rooted vegetation is gone.
- Soil and water conservation impacts on the water balance are expected to be greater in catchments with significant soil depth.
- Plot soil conservation techniques can be used to reduce soil loss and promote infiltration. A review of 20 studies (USAID, 1998) shows that median reductions in soil loss are 64% for grass barriers/ditches up to 80% for terraces, and increases in infiltration are around 50% for these techniques. However, these findings are not easily translated at the basin level given that deposition of soil often occurs before reaching a waterway, and increases in infiltration often just enhances AET.

Relationships between ecosystem services and components of the water balance

Total Flow Yield

Estimates of total flow are based on measurements of precipitation and Actual Evapotranspiration. Estimates of precipitation can be improved by identifying significant sources of variability that have implications for the amount of water that is intercepted by vegetation, or condensed from clouds (e.g., the position of slopes in relation to dominant winds).

Seasonal distribution of precipitation is important. Even if the precipitation gains in forested areas are insignificant compared with cleared areas, changes during the dry season may be significant, particularly at the community scale, and these changes have implications for Willingness-To-Pay.

Dry Season or base flow

This requires knowledge of catchment geomorphology and land use, in addition to AET and PPT.

Impacts of land use on the water balance are expected to be more significant where there is deep soil cover.

Stormflow

This requires data on intensity and duration of rainfall, in addition to AET, PPT, catchment geomorphology and land use.

Water Quality

This requires data on runoff and erosion of sediment, which will also affect the transport of other surface pollutants.

Biodiversity

- This requires data on the entire flow regime - which includes general basin characteristics and disturbance patterns, in addition to the components of the water balance - and how it is related to specific management objectives.
- A key characteristic of river basins is change and thresholds of resilience are inherently uncertain – so an adaptive approach is necessary.
- Identify key processes and characteristics of both headland and downstream areas, including expected ranges of variation in key parameters of the natural flow regime. Also, trade-offs among objectives in a basin-wide planning and developmentmust be considered, with attention opaid to uncertainties surrounding these trade-offs.

Optimum levels of flow will depend on the objectives to be achieved and trade-offs that can be accepted.

- In addition to components of the water balance (above), key easy to measure parameters for characterizing a basin that are easy to measure and can be used as indicators of change are: channel width, water discharge, sinuosity, and pattern.
- Establishing flows as a percentage of streamflow and allowing periodic highflows, at least at the bankfull level, can help to maintain natural patterns of variation, flush fine particles, and maintain channel structure and continuity. Periodic floods may also be necessary to nourish and maintain wetlands and riparian areas.

Evaluating trade-offs and institutional considerations

- Prevent perverse payment incentives, such as cutting down trees to receive payments to reforest afterwards (Pagiola and Platais, 2002), by documenting existing resources prior to reaching a PWES agreement.
- Economic significance depends on the direction and magnitude of changes in parameters of interest, and the spatial and temporal scales at which they can be detected.
- Given all of the uncertainties of environmental information and willingness-to-pay measures, it may be necessary to conduct sensitivity analysis – i.e., to carry out economic analysis under various assumptions and scenarios.
- Ecosystem processes cannot be considered services unless they have some form of economic significance – or, produce a stream of benefits to which stakeholders are also assured of access.
- Using results of the water balance estimate, identify whether changes in key processes are expected to increase or decrease impacts or benefits, by how much, and at what distance from the site. The economic significance of these benefits may determine potential WTP to reduce threats to their provision and insure future access.
- Land use change away from forest cover generally results in:
    - Increases in sediment yield as well as the flow of chemicals and nutrients;
    - Increases in water yield and peak flows;
    - *Either increases or decreases* in dry season baseflow and also in groundwater recharge, depending on the outcome of interactions among site specific processes that determine the net effect of changes in evapotranspiration and infiltration
- Land use impacts on flows of water and sediment are best examined and addressed at the level of individual hillslopes and patches.
- Water quality, water diversions, as well as cumulative impacts and climatic changes can be better detected and impacts felt at basin scales, at which they are more

appropriately addressed so as to permit consideration of trade-offs among all affected stakeholders.

- Given inherent uncertainty and the impossibility of obtaining complete information, identify relative contributions of specific management practices in particular areas to various objectives.
- Present information in a way that provides transparency regarding factors, options and trade-offs considered in decision making.
- Basic steps in organizing information include:
    - Classification of individual land tracts into units that reflect similar biophysical characteristics and processes of interest, by types of ownership, and by land use;
    - Identification of feasible land use and management options and opportunity costs, including business as usual, for different types of holdings;
    - Identification of decision criteria that reflect existing policies and concerns expressed by stakeholders, e.g., equitable distribution of costs, benefits and risks;
    - Rank options by each criterion, noting the degree of uncertainty and best judgment – rank may be indicated with actual monetary figures where these exist, other numerical values appropriate to the criteria, or qualitatively (e.g., high, moderate, low…).
    - Identification of overlaps among areas of high priority under different criteria, e.g., small holders who have low opportunity costs, located on steep slopes in fragmented cloud forest areas;
    - Identification of conflicts, e.g., high priority conservation areas in which opportunity costs are also high – these may require more stakeholder negotiation and special consideration of what is required to resolve particular conflicts;

- Identify existing rights and responsibilities, that underlie the distribution of costs and benefits.
- Maintenance is less costly than restoration.Also because of higher opportunity costs for restoration and uncertainty regarding success – payments may also be based on the desire to avoid uncertainty of change.
- A rapid participatory approach to valuation may be sufficient, when it is clear and uncontested that a particular option has higher value. When this is not the case, and when there are conflicting values, a participatory approach may provide guidance as to where further detail and precision are necessary. Such an approach can also improve understanding of links between natural resources and community livelihoods.

- Vulnerability and response capacity are site-specific and need to be understood in context, from the perspective of stakeholders.
- Distinguish community-scale livelihood vulnerabilities from basin scale infrastructure vulnerabilities.
- Payments for watershed services alone may not be sufficient to cover all costs. Where possible, watershed services should be considered as part of a package of ecosystem services.
- Ability of land users to accept compensation will depend on tenure or property rights. Recognition of formal and informal rights may also be critical also if objectives include poverty alleviation.
- Are costs covered for operations and maintenance? If not, it may be difficult to recover costs for conservation unless initiatives are linked to broader structural changes aimed at more equitable distribution of water that lowers costs for the poor.
- Identify incentives associated with existing property rights and determine their implications for delivery of watershed services and whether they are consistent with social objectives. Inconsistencies may point to the need for changes in one or the other, which may also require resolution of conflicts.
- New values imply redefinition of rights and responsibilities – a contested process that involves resolution of conflict.
- Value depends on the effectiveness of institutions as well as on supply and demand. Identify arrangements that can most effectively reduce transactions costs.
- Effective democratic decentralization can increase capacity to respond to site-specific characteristics and variability and provide a stream of revenue to support local governance.

ANNEX A: Thornthwaite-Mather Soil Water Budget

Notation: AWC = *Available Water Capacity* *[depth]*

SW = Available Soil Water (i.e., above wilting pt.) [depth]

APWL = *Accumulated Potential Water Loss [depth]*

ΔP = Net Precipitation; $P - PET$ [depth]

P = Precipitation [depth]

PET = Potential Evapotranspiration [depth]

AET = Actual Evapotranspiration [depth]

The available water capacity is calculated by multiplying the depth of root zone by soil porosity (found in soil textbooks), which are variables of the Thornthwaite-Mather water budgeting procedure, that are approximated from literature or observation. Land degradation impacts soil porosity whereas a change in landuse modifies the depth of the root zone. Clearly, the maximum root zone depths are limited by available nutrients and soil depth.

Situation I. The Soil is Drying

We know the soil is drying because $\Delta P < 0$

$$APWL_t = APWL_{t-1} - \Delta P$$

$$SW_t = AWC \exp\left(\frac{-APWL_t}{AWC}\right)$$

$$AET_t = P_t + SW_{t-1} - SW_t$$

Situation II. The Soil is Wetting

We know the soil is wetting because $\Delta P > 0$

$$SW_t = SW_{t-1} + \Delta P$$

$$APWL_t = -AWC \ln\left(\frac{SW_t}{AWC}\right)$$

$$AET_t = PET_t$$

Situation III. The Soil is Wetting above Field Capacity or more

We know the soil is drying because $\Delta P > 0$ and $SW_{t-1} + \Delta P > AWC$

$$SW_t = AWC$$

$$APWL_t = 0$$

$$AET_t = PET_t$$

ANNEX B: <u>Basin-wide estimates of seasonal Actual Evapotransipration based on existing stream flow and precipitation records</u> (Dias and Kan, 1999).

1) Negligible loss to deep groundwater is a reasonable assumption. Clues:

(i) Stream is influent (flows increase, after considering irrigation extractions, as you go downstream).

(ii) Water levels in possible wells surrounding the area are not significantly below stream level.

2) Obtain the linear recession coefficient, T, for the basin by finding the intercept, 1/T, of the upper envelope of data (Figure 9). Q is stream flow data for stream levels that have been falling for at least 2-3 days.

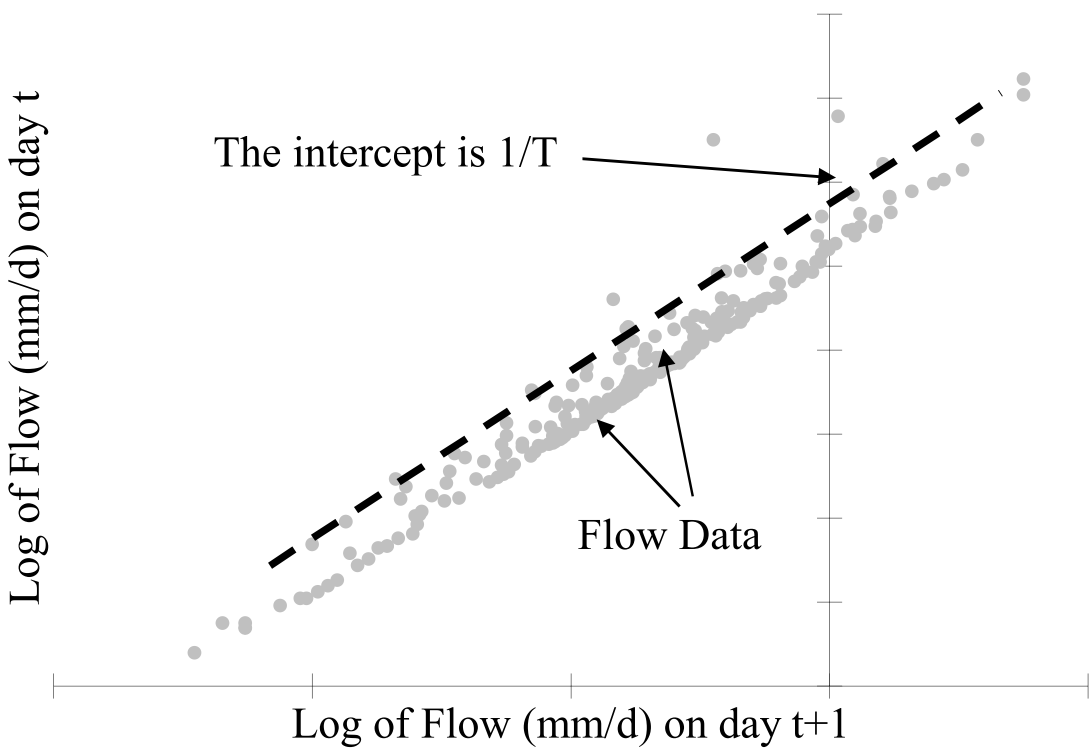

Figure 9. Plot to determine T.

3) Determine an effective storage, *S*, of the basin using;

$$S = T \times Q \tag{A.2.1}$$

Values for S are estimated where the streamflow levels have been falling for at least 15 days (threshold value recommended by (Dias and Kan, 1999)).

4) Determine AET using the following equation;

$$\sum_{\Delta t} AET = \sum_{\Delta t} P - \sum_{\Delta t} Q - \frac{S_f - S_i}{\Delta t} \tag{A.2.2}$$

where Δt is the time span between estimates of S.

5) Weighted averages must be done on estimated AET totals (summed through irregular Δt's) to obtain monthly AET values.

6) Landuse effects on AET can be evaluated by using this approach in nested catchments that have good historical streamflow and precipitation records. In addition, participatory appraisals about landuse change can also be used to gauge change on AET impacts (provided good historical hydrology records exist).

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