Assessing the Effectiveness of Payment Arrangements for Watershed Ecosystem Services (PWES)

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1 Introduction

There is growing interest among both conservation and development organizations in the development of Payment Arrangements for Watershed Ecosystem Services (PWES), for which the primary motives have been to create a steady flow of funding needed to achieve conservation objectives, to contribute to poverty alleviation by creating economic incentives for conservation, and to reduce disparities in the costs and benefits of management actions needed to produce ecosystem services. Willingness to pay (WTP) for watershed ecosystem services has been driven by an increased perception of threats to their continued provision and a recognition of the limits of regulatory approaches and the absence of economic incentives. A recent review by IIED identified 287 initiatives of Payments for Ecosystem Services (PES), of which 61 are for watershed services, and most of which are in initial or planning phases (Landell-Mills and Porras 2002). Key services paid for have included: ensuring regular flows of water, protection of water quality, and control of sedimentation.

Although critical to the establishment of payment arrangements, assessment of various aspects of the effectiveness of actions taken to ensure their delivery have received much less attention in current initiatives, which have focused primarily on the identification of potential buyers and on systems for collection of payments (Pagiola, Landell-Mills et al. 2002). However, to the extent that ecosystem services are a common pool resource, the value placed on them by actual or potential beneficiaries depends not only on demand, but also on stakeholder confidence in the effectiveness of proposed management actions needed to ensure that the service is actually delivered and that they will have access to the stream of benefits. These in turn, depend on:

- the integrity of ecosystem functions or processes that support service provision,
- effectiveness of institutional arrangements needed to insure their provision, and on
- whether impacts or benefits are economically significant at the relevant scale,

all of which are often assumed rather than assessed.

Given the complexity and natural variability of inter-dependent and site-specific factors that ultimately determine outcomes, and the impossibility of obtaining complete information, these factors are inherently uncertain. Market mechanisms, on the other hand, tend to be more effective when uncertainty is low, because buyers like to know if they are getting what they pay for. A precise determination of costs and benefits and their distribution, for purposes of establishing market values, presumes the ability to link actions and outcomes, so as to be able to demonstrate this. Making uncertainty explicit may be a harder sell, but is critical to managing buyer expectations and maintaining their cooperation in the long term.

A key challenge then, is to develop a site-specific assessment process in support of PWES initiatives. The purposes of such an assessment should be to:

- identify, assess and prioritize watershed ecosystem services,
- support the development of equitable institutional arrangements that ensure access to benefits by those who pay the costs of providing services, and
- monitor implementation to determine if objectives are actually being achieved.

Absent an independent and transparent process of assessment, initiatives are often based on myths about land and water relationships that can lead to inappropriate actions and also to scapegoating of marginal groups in remote upper watershed areas. Myths about land and water relationships fall 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 the levels of evapotranspiration and infiltration, 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, that receives runoff at different rates from many different sources in the upper watershed.
- 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.

Equally misleading is the notion that science can provide certainty, though it can allow a better approximation as to the magnitude and direction of impacts, monitoring, and more informed decision-making. Because of complexity and uncertainty, to some extent myths are unavoidable but need to be continuously questioned as new knowledge becomes available, and replaced when they have outlived their usefulness or are not relevant to a particular context. A more constructive approach would be to use information provided by assessments to develop a range of plausible "scenarios" which, like myths, can be used to "package" extensive amounts of information about complex problems into narratives that are comprehensible to stakeholders and enable them to also have greater awareness of uncertainties – just as they do for weather predictions.

This paper presents an overview of a draft assessment guide being prepared to support the development of payment arrangements, and identifies the kinds of information needed to select an appropriate approach, with a special emphasis on the identification and measurement of ecosystem services. A second purpose of this presentation is to obtain feedback so as to insure that the final product is relevant to user needs.

2 Information priorities

Assessment refers to information of relevance for evaluating trade-offs and making decisions. More information is not necessarily better, and may even make things worse by providing an illusion that the problem is understood and can be adequately responded to (White 1996). Often, endless data is gathered on narrow technical aspects of a problem, when what is needed is to be aware of aspects that are overlooked altogether, particularly sources of significant uncertainty. In this section, we provide a framework for assessment of the site-specific context by identifying various categories of information needs, and discuss their relevance for decision-making. Rules-of-thumb and more detail on methodological approaches associated with gathering the necessary data are provided in the full draft of the knowledge guide.[1]

2.1 Identifying and Measuring Watershed Ecosystem Services

Watershed services are a product of ecosystem processes or functions through which they are maintained. These processes may include elements of the landscape context, such as climate and upstream land uses, which may enhance or interfere with the natural flows of water and sediment. However, they cannot be considered "services" unless they also have some form of economic significance for identifiable stakeholders. Economic significance of ecosystem functions, and consequences of change, will depend also on their magnitude, the scale at which they are significant, and on downstream uses of water and land that are dependent on these natural flows and that are within the relevant scale (Aylward 2002). Therefore, they cannot be assessed purely from an abstract and biophysical point of view, without reference to the social context - through which they are given value (Geores 1996).

Although services need to be defined in a site specific context, they can be generally classified in two broad categories, of those which provide direct and indirect streams of benefits to humans. Specific kinds of services include:

- Provision of water for:
    - consumptive uses (drinking, domestic, agricultural and some industrial uses),
    - non-consumptive uses (hydropower generation, cooling water, and navigation),
- Flow regulation and filtration – i.e., maintain water quality, water storage 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, 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 coastal storm damage.
- Cultural services (recreation, tourism, existence values), and
- 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.

2.1.1 Water balance as a framework for investigation

Estimation of the water balance, which refers to the change in water storage within a watershed, combined with an accounting of water needs and uses, provides a good point of departure for assessment of the ecosystem functions associated with freshwater services, can provide clues to mismanagement and also reveal the extent to which water is a limiting factor during dry periods. This can, in turn, help in the estimation of demand and what stakeholders may be willing to pay for services associated with its provision, and in the identification of priority areas for implementation of conservation practices.

[1] Available soon, on request, from stognetti@mindspring.com.

Of greatest importance is to know the range of variation. Although the ability to collect hard data on variation over a short period is largely up to chance, it can be supplemented with "soft" data based on local knowledge that can be obtained from stakeholder interviews and knowledge of similar catchments. Through monitoring, it can also be improved over time.

The water balance is essentially what remains once streamflow, actual evapotranspiration, and loss to deep water aquifers are subtracted from overall precipitation, or:

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

Where, *ΔS* is the change in storage, *P* is net precipitation, *Q* is Streamflow, *AET* is Actual Evapotranspiration, and *G* is loss to deep-water aquifers not accounted for by streamflow. The primary data requirements for calculating a water balance are:

- Precipitation, which, in addition to rainfall, may also include interception and condensation of water by cloud forests. Given the spatial and temporal variation of rainfall, the quality of measurements of direct precipitation will depend on the placement of gauges as well as the period of time over which the data is collected. Interception by cloud forests is much harder to measure and has been based on approximations of the amount of net water gain that can be attributed to rainfall and cloud interception. However, these figures have not accounted for significant sources of variation such as location of the slope in relation to winds, which affects the amount of cloud moisture captured, and the intensity of storms – which affects the amount of precipitation intercepted and collected by the canopy. Precipitation gains also vary by season and are higher in the dry season (Bruijnzeel 2001).
- Actual Evapotranspiration (AET) – this also depends on numerous variables that include: precipitation, temperature, solar radiation, soil type, drainage, wind, canopy, understory interception, vegetation type and maturity, and land use change. It is also important to account for seasonal variation, and to delineate areas with significant sources of variability, for which estimates should be made separately. The latter is also central to the identification of effective management actions. Measures used to estimate AET are:
    - The total supply of energy and water, on a seasonal basis, which indicate the outer bounds;
    - Potential Evapotranspiration, which can be estimated from reference evapotranspiration rates, and adjusted for the effects of vegetation, using crop coefficients – rules of thumb values exist though actual data is still scarce for tropical forests;
    - Soil capacity to store water, and whether plants have access to it, is a key source of variation in AET. This depends on whether the soils are shallow or deep, and whether vegetation has deep or shallow roots. Available water capacity in the soil can be estimated based on the depth of the root zone and soil porosity for which average figures are provided in the literature, for types of species, biomes, and soil types (Canadell, Jackson et al. 1996; Neitsch, Arnold et al. 2001). A general rule of thumb is that land use change will have greater impact on the water balance where soil cover is significant in that deforestation tends to reduce the root zone as well as degrade the soil structure, reducing porosity.

A monthly water budget of the soil, together with data on precipitation, net precipitation, and potential evapotranspiration, can be used to estimate seasonal AET, using Thornthwaite-Mather Soil Water Budget equations (Thornwaite and Mather 1957). Because of simplifying assumptions, this model may result in overestimates of AET but is useful for comparisons of management options. Basin-wide

AET can be estimated using stream flow and precipitation data to compare differences between storage and net precipitation.

A major obstacle to determining AET is the difficulty of obtaining site-specific land cover and land use data that reflects significant heterogeneities generally found in a landscape. Many of these features operate at the scale of individual hillslopes, and can be difficult to distinguish even with most remote sensing technology. For example, the characteristics of narrow riparian areas can have effects on hydrology that are disproportionate to the area they occupy. Some characteristics of forests, such as tree heights, can be difficult to measure even on-site where there is dense canopy.

However, 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. 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 distinctive characteristics it is also expected to significantly reduce uncertainties in watershed process models, and ultimately, in land and water relationships.

Many hydrology studies assume that net losses to the water balance from deep percolation are negligible. This is usually a reasonable assumption at the head of a watershed. Some indicators that can be used to easily verify this are:

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

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

However, it is extremely difficult to quantify the impacts of upstream landuse management on downstream groundwater recharge without field surveys and modeling studies. It is also difficult to monitor well water use, given the lack of groundwater regulations or enforcement in most parts of the world, and to develop deep-water budgets.

Streamflow data can be used to develop flow duration curves and determine the dryness index of a catchment (Farmer, Sivapalan et al. 2003). Changes in flow duration curves can provide clues to the consequences of changes in management practices. The more arid catchments generally have steeper flow duration curves and therefore, a more limited capacity to sustain dry season flows. However, exceptions can be found in particular watersheds, where water is stored in fractured rock, which limits access to it by vegetation, or where there are deep soils, and medium or shallow rooted vegetation, which minimizes loss to evapotranspiration – a situation common to the Andes.

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 or very deep permeable soils. Storage and release of water from wetlands also depends on site-specific processes (Bond, Jones et al. 2002).

2.1.2 Significance of impacts and benefits

The water balance and flow duration curves can then be used in process models to determine the significance of watershed processes for providing particular services. This requires:

- identification of the magnitude and direction of impacts,
- identification of the scale at which these changes can be detected, and
- a survey of downstream uses of land and water, stakeholder vulnerabilities, and potential conflicts.

Economic impacts will depend on downstream uses of land and water as well as the interests and vulnerabilities of stakeholders, among whom impacts are often not evenly distributed. Given that these impacts may be both positive and negative, depending on what is valued and measured, it is important to consider their full range, and also their relative magnitude or significance (Aylward 2002). Of particular relevance is to identify and account for competing water uses when it is most scarce, in the dry season. Also to be considered are vulnerability to floods, drought, and disruptions of the natural flow regime. Ultimately, the definition of watershed ecosystem services also provides a basis for identifying threats to their continued provision.

An analysis of the various kinds of assets that are used to sustain livelihoods provides a way to identify impacts that need to be considered in decision-making, from the perspectives of stakeholders (Ashley and Carney 1999). It can also show the role of ecosystem services in sustaining livelihood.

2.2 Identifying effective institutional arrangements

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).

This implies the need to develop effective institutional arrangements to control access, without which economic value cannot be captured, and which are therefore also a prerequisite to the development of payment arrangements. They are also a source of tremendous site-specific variation that needs to be considered in order to develop effective PWES initiatives. Of primary concern are 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. Institutional arrangements also refer to relationships established among buyers, sellers, and intermediary organizations so as to reduce transaction costs.

Private property rights are only one of a number of different kinds of arrangements for controlling access to resources – property may also be publicly owned or held in trust by the government, or be held in common by a community – publicly or privately, and may include informal rights based on customary practices and social norms. 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). 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.

For example, security of land tenure may provide a greater incentive for switching from crops to agroforestry, because the benefits do not materialize for several years. Rights to water based on historic use or "prior appropriation", which usually require also that the water be used in ways that are socially beneficial, 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. 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, but may limit the ability to transfer the water and to develop water markets. The ability to develop water markets could provide incentives for greater efficiency and provide a source of revenue for the development of upper basin areas. However, riparian rights make it possible for local communities to control access and exercise customary rights, which provide an incentive for conservation because access in the future is ensured. In an open access situation, the incentive is simply to consume resources before someone else does.

It is also important to identify informal use rights or norms, by considering "all of the strategies used by individuals to claim and obtain water" (Meinzen-Dick and Bruns 2000). Special attention needs to be given to water uses and rights associated with gender and with particular subgroups of the population. In what is referred to as a process of "legal pluralism" (Meinzen-Dick and Pradhan 2002), different claims often overlap and conflict in what is typically an ongoing process of conflict resolution and institutional development.

In that they are intended to provide security, property rights do not change easily or quickly, absent political momentum generated by events such as the end of the cold war or the fall of apartheid. However, they have never been static and tend to change as greater values are placed on particular resources and as technological improvements bring down the transaction costs of controlling access to them (North 1990). 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. Similarly, as a consequence of the growth of urban areas, rivers became more highly valued for sewage disposal than for supporting commercial fisheries, and land uses became restricted by zoning so as to protect existing values. Such changes may also be associated with the recognition of new kinds of problems, such as those associated with environmental degradation. Just as changes in rights are implicit in the development of physical infrastructure such as irrigation systems and dams, ecosystem management also implies the negotiation of new rights and responsibilities in which landowners are obligated to protect the ecosystem, and in which the use of land, water and other natural resources are limited to those uses that do not impair its function (Sax 1993). 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, and that may also be made feasible through technological improvements, such as in mapping and communication.

Water institutions that have been in place throughout Latin America in recent times have two fundamental characteristics. The first of these is extensive governmental claims on natural resources. In Ecuador, for example, the *Ley de Aguas* of 1972 effectively extinguished all private water rights. Traditional claims were recognized, but only as concessions from the state that were subject to revocation. The second fundamental characteristic of water institutions is heavy subsidies. What Ecuadorian farmers pay for irrigation water, for instance, is a tiny fraction of the cost of delivering this resource to their fields. Subsidized irrigators grow accustomed, in effect, to not paying any of the capital costs of the projects they benefit from, and do not take full financial responsibility for operations and maintenance. Under this regime, it is a tall order to convince them to pay for the conservation of upper watersheds, which are supposed to be the ultimate source of irrigation water.

However, the situation also means that municipalities are unable to recover their costs, and therefore with a lower capacity to provide services, which places the poor and underserved populations at an even greater disadvantage. A case in point is that of Quito Ecuador, where, as recently as the late 1980s, the municipal system was highly subsidized. Payments from customers covered no more than half of O&M costs and amortization expenditures. Financially strapped, the municipal company was only able to provide connections to about 60 percent of the population. The 40 percent of the population without service, primarily in peripheral slums, relied on water delivered by tanker trucks, and the price paid for this water was approximately ten times what better-off household with a piped connection were charged. During the next ten years, subsidies were drastically reduced, which provided the municipal company with the means to extend service to poor neighborhoods. As of 1998 (the year before a severe macroeconomic crisis began in Ecuador), revenues from customers were less than 10 percent below costs and nearly 90 percent of the metropolitan population had a piped connection. Not coincidentally, a small fund to help finance watershed management had also come into being.

Recent initiatives by the World Bank and other agencies to reform irrigation, potable water, and related sectors, so as to recover costs, seek to devolve the responsibility for managing irrigation systems and other water infrastructure to associations of local water users. These associations are expected to raise prices enough to cover operation and maintenance costs at least. The pay-off for association members, at least in theory, would be in improved reliability of service made possible by higher prices. Among the barriers to implementing reforms is that there is little ability to pay, and little public confidence that these would result in improved service reliability. This suggests the need for a long-term strategy, with a focus on what is needed to actually improve service, and on building public confidence.

Payments for watershed 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, payments to stakeholders could be seen as violating the principal of polluter pays, unless accompanied by sanctions on pollution (UN FAO 2002). However, for purposes of maintaining ecosystem services, payments are intended simply to provide an incentive for landowners to provide valued services in addition to agricultural products. Regardless of how resources are owned, the key question is whether or not they achieve the objective of improving the provision of watershed services, which is unlikely to happen unless they are regarded as fair. Ultimately, a key aspect of assessment is to effectively involve stakeholders themselves in identifying options that are feasible and fair in a given context. Key questions that have been used to facilitate discussion (Attwater 1997) are:

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?"

This kind of analysis can inform the selection of appropriate economic instruments and negotiation of equitable arrangements.

In theory, if all costs could be accounted for, property rights could be created and contracts negotiated among all gainers and losers so as to maximize everyone's benefit from the resources, assuming that transaction costs are negligible (Coase 1960). In practice there are significant transaction costs associated with control over access to common property resources, although these may be reduced over time, as a result of technological improvements (North 1990). In the case of watersheds, advances in mapping and remote sensing, as well as in communications may make monitoring and enforcement more feasible, and also make it possible to better inform processes of stakeholder negotiations. Given the need for collaboration over large areas in upper watersheds, another critical aspect of reducing transaction costs is to develop agreement among diverse stakeholders on an effective management plan, and establishment of organizational entities through which it can be implemented, which also provides something tangible that can be paid for, which may attract greater funding, whatever the source.

Use of economic instruments to provide delivery of watershed services essentially consist of the negotiation of various kinds of arrangements among buyers and sellers, which may take various forms depending on the nature of the service, and the socio-economic and institutional context. These range form informal, community-based initiatives, to more formal contracts between individual parties, and to complex arrangements among multiple parties through intermediary organizations, in which the government may play different kinds of roles. Specific instruments include user fees, direct payments, marketable permit systems, voluntary contractual arrangements, tradeable development rights, and certification and labeling. Individual initiatives may consist of a mix of market-based, regulatory and policy incentives that become necessary at larger scales, when threats are beyond the response capacity of individual communities. In general, benefits will be more tangible, and contractual arrangements more feasible, at smaller scales, where links between causes and effects can be more readily established, where property rights and stakeholders can be better defined, and agreements can be tailored to local conditions. At larger scales, where it is harder to link causes and effects, and rights and responsibilities are harder to define because of common property characteristics of the resource, there will be greater need for government involvement (Rose 2002).

3 Valuation of watershed services

The definition of resources or services is not static but rather, an ongoing contested process in which there are usually conflicting claims among multiples uses, interests and objectives. Values placed on services by stakeholders, whether the exchange is monetary or in some other form, are contingent upon how services are defined. This implies the need to consider trade-offs among multiple uses, interests and objectives, and to inform a process of conflict resolution and negotiation among stakeholders regarding equitable PWES arrangements. This should also include consideration of the values people place on places and ways of life for which they may be willing to make trade-offs that are not necessarily monetary.

A key question that has implications for selecting an appropriate arrangement is the extent to which tangible aspects of ecosystem services, such as provision of water for direct use, can justify the added costs of conservation actions when compared to the opportunity costs of forgone land uses. For example, a study in the Arenal watershed in Costa Rica (Aylward and Echeverria 2001) found that even in the absence of perverse subsidies, 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 was offered for reforestation. Further, the expected decline in water yield associated with reforestation was the dominant factor in the economic analysis because annual water yield was of direct benefit to a downstream hydroelectric facility.

However, these costs and benefits were not all distributed equally – a subsequent companion study that examined costs and benefits from the perspective of major stakeholders, and which made distinctions among various kinds of landholders, found that the higher return per hectare depend in part on location in the catchment, that they accrue primarily to large landholders, and that incentives that were being offered for conservation may still appear attractive to small landholders (Aylward and Fernández González 1998).

A key problem then is to define relative values of various watershed areas so that they can be prioritized for purposes of decisions about funding allocations. Construction of a multi-criteria framework can be used to structure available information, and make all of the considered factors, and decision criteria transparent to stakeholders, thereby allowing them to participate more effectively in negotiation regarding the development of equitable arrangements. It can also be used to prioritize further information needs.

4 Conclusion

Effective assessment of watershed services can increase user WTP and also the confidence of external donors. Ineffective management actions are often overlooked simply for lack of any incentive to conduct integrated and comprehensive assessments, and because consequences tend to fall disproportionately on marginalized stakeholders who have little if any voice in decision-making. When assessments are conducted, the problem is often defined narrowly, leaving large blind spots. Often, new and unanticipated kinds of problems only come to light as a result of independent assessments by NGO and academic researchers, who gather information and also disseminate it to those most affected. Cooperation in an assessment process can also be a starting point for PWES initiatives because information provides the basis for common understanding of problems that is a prerequisite to any form of collaboration.

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