Markets for Watershed Services

Sylvia S. Tognetti
Consultant, Environmental Science and Policy

Bruce Aylward
Deschutes Resources Conservancy

Guillermo F. Mendoza
New York Department of Environmental Protection

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ABSTRACT

Increasing degradation of watersheds has led to increased recognition of the services they provide, in various forms of support for livelihoods and general well-being, as well as to a greater willingness-to-pay for them and to cooperate in initiatives to protect them. This is reflected in numerous initiatives, in which market-based instruments and other supporting institutional arrangements are used as a way create incentives and to recover the costs of watershed protection, as well as to allocate water more efficiently among various uses. Many of these payment initiatives have focused narrowly, on the role of forests in the hydrological regime, as a way to justify funding for their conservation, but should be developed in the context of basin-wide management objectives, which provides a framework for considering the full range of interests that share a common river basin, in the context of specific ecosystem functions that support them, and for identifying and quantifying trade-offs associated with various management options. In addition to forests, this would include consideration of the relative values of all types of landcover and land uses, such as wetlands, riparian areas, steep slopes, roads, and management practices. It also requires an accounting for the role of human consumption in the modification of the hydrological cycle, so that these changes can be distinguished from natural variation, and so as to be able to distinguish biophysical from economic causes of scarcity. Economic justification for an initiative may also require that initiatives aimed at protection of freshwater supplies be part of a package of approaches designed to capture the value of multiple ecosystem services found in landscapes, and to resolve conflict among various

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uses. Given the heterogeneity of landscapes, and the site-specific nature of ecosystem services, a key challenge is to develop the capacity for place-based approaches to monitoring and assessment. This article provides an overview of the types of economic instruments and institutional arrangements used to capture the value of watershed services, the assumptions on which they are based, institutional challenges faced in implementing them, and the kinds of scientific information needed to identify economic trade-offs, inform stakeholder negotiations, and support decision-making.

MAIN TEXT

Introduction

The general decline in the capacity of watershed ecosystems to provide essential goods and services has led to an increased recognition of the ways that they support human well-being. These include direct and indirect economic benefits, ranging from provision of freshwater for various consumptive and non-consumptive uses, to those associated with regulation of the flow of water and sediment, and support for ways of life that have cultural value and that involve land use practices consistent with continued provision of services. However, unless those asked to pay the costs of practices necessary to insure continued provision of these services are also assured of benefits, there is little incentive for them to do so.

Regulation of land use practices alone is often ineffective because they tend to place a disproportionate share of the burden on upstream land users without giving them a corresponding access to benefits. For example, it is common for states to claim ownership of forested areas, and to protect watersheds through policies that exclude local populations from access to resources on which they have traditionally relied, which may lead them to occupy more marginal land areas (Tomich et al., 2004). Those who do practice farming in upland areas may barely be able to recover their costs of production even in the absence of conservation measures.

This situation has led to an increased interest in the use of market-based mechanisms, as a way for upstream land users to recover the costs of protecting watershed services. A recent review by IIED identified 287 initiatives of payments for ecosystem services of forests, of which 61 were specifically for those associated with watersheds. The main concerns addressed in these initiatives have been maintenance of dry season flows, protection of water quality, and control of sedimentation (Landell-Mills and Porras, 2002).

In theory, market-based approaches can lead to more efficient allocation of resources and to more cost effective solutions to the problem of watershed degradation. In practice, there are a number of scientific and institutional challenges encountered in their implementation, and little evaluation of the transaction costs of addressing these challenges. However, in evaluating these costs, it should be kept in mind that the development of institutional capacity needed to effectively respond to watershed degradation can also have other benefits. For example, it can lead to social cooperation in other matters, development of skills, opportunities for clarification of land titles, and increase of scientific understanding, and environmental education (Landell-Mills and Porras, 2002). It should also be kept in mind that regulatory and market-based approaches are not mutually exclusive, and often complement one another. Given also that ecosystem services tend to be very site-specific, there may be no clearcut answer regarding *the* most efficient approach.

To the extent that there is rivalry over access to a limited supply of watershed services, a central challenge is to develop institutions or enforceable rules through which access can be limited to those who are entitled to them, and which also define responsibilities for actions needed to insure they are provided (Ostrom, 1990). To the extent that watershed goods and services have characteristics of public goods, which makes it difficult or expensive to limit access, the willingness of potential beneficiaries to pay for them, depends not only on demand, but also on whether they have 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. In other words, the value of watershed services will depend on:

- the integrity of ecosystem functions or processes that support service provision,
- the scale at which impacts or benefits have economic significance, and on
- the effectiveness of institutional arrangements needed to insure provision and access,

all of which are often assumed rather than assessed.

A fundamental paradox is that, given the complexity, natural variability and stochastic nature of multiple inter-dependent and site-specific factors that ultimately determine outcomes, and the spatial and temporal separation of causes and effects between upstream and downstream, and between the present and the future, complete information is unobtainable and uncertainty is inherent. 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. It is also a critical consideration in negotiating an equitable distribution of costs and benefits.

Absent an independent and transparent process of assessment, initiatives have often been based on myths about land and water relationships that can lead to inappropriate or partial solutions and also to placing a disproportionate share of the blame on marginal groups in remote upper watershed areas. Equally misleading are notions that science can provide certainty and that markets can solve all problems. However, science can allow a better approximation as to the magnitude and direction of impacts, monitoring, and more informed decision-making, while market-based instruments are critical to finding ways to cover the costs of providing valued services. Because of uncertainty, which is inherent in complex problems, to some extent myths may be unavoidable and even useful, as plausible scenarios, but need to be continuously questioned as new knowledge becomes available, and replaced when they found to be implausible or irrelevant in a particular context.

This article provides an overview of the various kinds of potential watershed services, their biophysical characteristics, the kinds of instruments that have been used to create incentives for their provision, and the information used to support them. This is followed by discussion of their inherent assumptions, and challenges faced in implementing them. The article concludes by identifying the kinds of scientific information needed to identify economic trade-offs, to inform stakeholder negotiations, and to support decision-making.

Defining watershed services

The ability to link payments to the level of service provision requires an identification and quantification of benefits actually provided, in a specific context so as to be able to determine their economic significance. This section outlines the various types of watershed services and biophysical characteristics that need to be considered in doing this.

Watershed services are products of ecosystem functions or processes that provide different kinds of direct and indirect streams of benefits to humans, in the following general categories:

- Provision of freshwater for:
    - consumptive uses (drinking, domestic, agricultural and industrial), and
    - non-consumptive uses (hydropower generation, cooling water and navigation).
- Flow regulation and filtration, key aspects of which are the control of mean surface runoff, peak or flood flows, base or dry season flow, and erosion and sediment load, as well as recharge of groundwater and soil moisture (UN FAO, 2002). Benefits of these may include:
    - water storage in soils, wetlands and floodplains which can buffer flood flows and drought;
    - control of erosion and sedimentation which, in excess, can have adverse effects on aquatic life, irrigation canals, dams, and navigation. Below normal flows of sediment downstream from dams can have adverse effects on coastal areas where it provides protection from erosion and nourishes the development of mangroves, both of which can reduce storm damage;
    - maintenance of river channels, wetlands, riparian habitats, fisheries, and other wildlife habitat that may be important for hunting, migratory birds, rice cultivation, and fertilization of floodplains;
    - maintenance of mangroves, estuaries, and coastal zone processes, which often rely on seasonal pulses of freshwater inputs, and are critical habitats for fisheries as well as for other marine life;

- control of the level of groundwater tables that may have adverse effects on agriculture by bringing salinity to the surface; and
- maintenance of water quality, which may be impacted by inputs of nutrients and organic matter, pathogens, pesticides and other persistent organic pollutants, salinity, heavy metals, and changes in the thermal regime.

- Supporting services that may include:
  - maintenance of natural flow and disturbance regimes as drivers of ecosystem processes, which also supports ecosystem resilience. Resilience in turn, provides some measure of insurance against the uncertain effects of a change in conditions, for which thresholds are generally uncertain.
  - support for cultural values which may include aesthetic qualities that support tourism and recreational uses, and support for ways of life.

These various kinds of services are interdependent, in that there is a trade-off between provision of freshwater for direct uses, and on the regulatory and supporting services that insure continued provision. Therefore, a fundamental objective should be to achieve an acceptable or optimal balance between these trade-offs.

The above list only represents the kinds of benefits that watersheds may provide. However, benefits cannot be considered "services" unless they also have economic significance. Therefore, site specific assessments are necessary to identify benefits that are provided in a specific context, and the scale at which they can be detected. This then provides a basis for identifying the economic significance to various stakeholders, so as to be able to identify potential "buyers," actions required to insure that benefits continue to be provided, and the levels of compensation needed to create an economic incentive for potential "sellers". This is further discussed in the section on challenges to implementation,

Absent an independent, transparent and place-based process of assessment, payment initiatives are often based on myths or general beliefs regarding watershed services and their values. These can in turn lead to ineffective or partial actions, inefficient allocation of resources, and often to placing an inappropriate share of the blame for watershed degradation and water scarcity on marginal groups in remote upland areas - all of which can exacerbate rather than solve problems, or create entirely new ones (Kaimowitz, 2001). Myths regarding land and water relationships fall into 3 general categories:

- *Inappropriate generalizations* from one site to another, and in particular, application of knowledge from wet temperate to arid and tropical zones.
- *Forests and water myths* – e.g., that forests significantly reduce or prevent flooding and increase dry season flows. Whether or not either of these occur 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, and the scale at which increased or decreased flows of water are significant. For example, forests may significantly reduce flooding in a localized area, but tend to have an insignificant impact or to be averaged out at larger scales, where runoff is received at different rates from many different sources in the upper watershed. The types of

vegetation that occur, the depth of roots relative to water tables, and whether there is ground cover, will also play a significant role in the amount of evapotranspiration that occurs, and levels of stream flow. Soil that has been compacted as a result of previous management activities, the presence of roads, and other construction associated with development, can also reduce infiltration and change drainage patterns, regardless of whether trees are planted (Calder, 1999).

- *Erosion myths* – that soil conservation 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, though it may have onsite benefits for the farmers (UN FAO, 2002). Another fallacy regarding erosion at a landscape level is that it can be determined from plot level measurements, which assumes that all eroded soil has a negative impact, regardless of where it is deposited, and that agriculture is the main culprit (Swallow, Garrity et al. 2001). There is considerable evidence that erosion rates are significantly higher on roads and foot paths and marginal areas such as forest margins and steep slopes (Bruijnzeel, 2004). A recent study in an upper catchment in Northern Thailand found that unpaved roads produce as much sediment as agricultural land despite the fact that these roads occupy less than one tenth of the area occupied by agriculture (Ziegler, Giambelluca et al. 2004).

In sum, land use and hydrology interact in complex ways, in which forests are one of several factors that should be considered in the context of the entire flow regime. Even in small sub-basins, there may be complex interactions between various biophysical aspects and between them and human management systems that make it difficult to predict management outcomes. Whether any of the above presumptions hold therefore depends on the outcome of multiple interactions, and is therefore highly contextual or site specific.

Not to be overlooked are increases in water scarcity brought about by increases in demand, both upstream and downstream. For example, a case study in Thailand suggests that dry season flows have diminished primarily because of a dramatic increase, both downstream and upstream in dry season cultivation and irrigation of soybeans, by those who own paddy fields. However, the focus of regulation intended to address the problem has been on the more vulnerable farmers, who are dependent on rain-fed slopes in areas where significant forest cover remains, who have the least significant impacts on hydrology, and who are generally regarded as guardians of resources rather than as legitimate users (Walker, 2003).

Even when land and water links can be demonstrated and quantified, a recent literature review on the subject raises questions about whether the magnitude of damages or benefits is likely to be economically significant, when considering just the relationship between land use and hydrology. This will largely depend on downstream economic interests that rely on water, and the scale at which impacts are significant (Aylward, 2004). However, there are other offsite values associated with land use practices, in addition to those associated with freshwater flows that may have significance when combined (e.g., biodiversity protection, carbon storage, recreational values). Therefore, it is important to consider how watershed protection contributes to basin-wide management objectives.

Types of market-based instruments used to create incentives for protecting watershed services

As mentioned above, initiatives to develop market-based incentives for the provision of ecosystem services associated with freshwater, have been made possible by a general recognition that regulations alone are inadequate for achieving this. Also, by the increased threats, or the increased perception of threats, that has led to an increase in Willingness-to-Pay by beneficiaries of these services, (Landell-Mills and Porras, 2002). The main concerns addressed in the initiatives reviewed by IIED have been maintenance of dry season flows, protection of water quality, and control of sedimentation (Landell-Mills and Porras, 2002). However, absent the use of appropriate economic instruments and supporting institutional arrangements among beneficiaries and providers, i.e., buyers and sellers of services, these values are hypothetical. This section describes the various kinds of economic instruments used to protect watershed services, discusses factors that need to be considered in their selection, and provides examples of their use.

Payment arrangements for watershed services may take various forms depending on the nature of the service, what is required to provide it, whether and how it is possible to limit access to benefits to those who pay the costs of provision, the scale of relevant ecosystem processes that support it, and more generally, on the geographical and historical context. They may range from informal, community based initiatives, to more formal contracts between individual parties, and to complex arrangements among multiple parties facilitated by intermediary organizations. They may also include a mix of complementary market-based, regulatory and policy incentives that are more likely to become necessary at larger scales, when threats are beyond the response capacity of individual communities (Rose, 2002), and when multiple services are involved. The government may play different kinds of roles depending on the type of arrangement, ranging from the enforcement of contractual agreements, to the creation of regulatory incentives, monitoring compliance, contracting with service providers, providing technical assistance, and identification of priority conservation areas. Some of these roles may also be filled by non-governmental organizations, which often have the advantage of greater flexibility, and the ability to act more expediently. NGOs may also play the role of advocates on behalf of less powerful constituencies, so as to create political pressure that may be necessary for governments to recognize their rights and respond to their concerns.

Actual design of payment arrangements will also reflect policy decisions as to who should pay costs and who is entitled to benefits. An example of a policy decision would be whether or not farmers should be entitled to the value of farmland when it is converted to urban uses, when this is driven by public investments in infrastructure, and the extent to which they should be compensated for any restrictions on land use conversion (Buist et al., 1995).

Individual economic instruments can roughly be placed in the following general categories, and are often used in combination:

Voluntary Contractual Arrangements (VCA) – these typically involve the negotiation and agreement of a contract in which resource users, who benefit from watershed services, compensate upstream landowners for the costs of adopting management actions needed to insure provision. Intermediary organizations such as landowner associations, are often necessary as a way to reduce transaction costs associated with the need for agreement and collaboration among numerous downstream beneficiaries and landowners dispersed over large upper watershed areas. An important consideration is the relative power of stakeholders at the bargaining table, what information is available to them, and whether any significant stakeholders have been excluded. Another is whether land users have some form of property right or tenure security, without which they may not have authority to enter into contractual agreements and therefore have access to the benefits. VCAs are more straightforward when negotiated among individual parties, such as the case of an agreement between the La Esperanza Hydropower Company and the Monteverde Conservation League in Costa Rica, which is the sole owner of the forested area upstream from the plant (Rojas and Aylward, 2002). Agreements among numerous parties will also require that more consideration be given to the establishment of decision-making entities for purposes of allocating funds to priority conservation measures. This may take the form of a trust fund, such as FONAG (Fondo del Agua), a trust fund established in Quito Ecuador to protect 2 upstream ecological reserves. This fund is overseen by a stakeholder board, and allocates pooled funds and in kind support received from municipal entities that provide water and electric services, from NGOs, and from private sources (Echavarria, 2002, Echavarria, 2003).

Transfer Payments (TP) – these are payments made to land owners as compensation for the costs of adhering to specified management practices. This is a hybrid approach, in that payments are usually made on the basis of VCAs with landowners but differ in that the contract is normally made between landowners and the government, for purposes of achieving broader policy objectives rather than directly with downstream users, in response their specific concerns. When voluntary, landowner participation will depend on whether payments are sufficient to offset their opportunity costs, which may be revealed by allocating them on the basis of bidding systems. It is also a mechanism often used to allocate funds collected through various sources, which may include User Fees, conservation donors, and general tax revenues. Transfer Payments may also be funded through the sale of marketable permits (described below) to those who have higher costs of regulatory compliance. The best known examples of TPs are the U.S. Conservation Reserve Program and similar initiatives in some countries in the European Union, in which farmers are compensated for conservation measures based on a number of criteria that include water quality (USDA, 2000). An arrangement established by irrigation farmers in the Cauca Valley in Colombia, in which they voluntarily pay additional water use fees to support watershed conservation, could also be considered a system of transfer payments. This is because the farmers themselves do not have the authority to directly fund watershed management activities. Instead, the fees are paid to a governmental entity - the Corporación Autonóma Regional del Valle del Cauca – which uses them to implement existing watershed management plans (Echavarría, 2002).

Acquisition – This approach encompasses many forms, ranging from the acquisition of rights to water and land parcels, to the acquisition of partial rights through easements that restrict uses, and leasing, in which rights are acquired for a specific period of time. In the context of watershed protection, it is common for governments or non-governmental entities such as land trusts, to acquire development rights on particular parcels of land so as to prevent their conversion to other uses, while owners may retain rights to occupancy and/or other specified uses. An example is the purchase of land and conservation easements in critical areas that cover the Edwards Aquifer, which supplies drinking water to 1.5 million people in Austin and San Antonio Texas (U.S.) (Trust for Public Land, 2001). It is also one of the instruments used to implement the New York City Watershed Agreement, in which the city invests in upstream watershed protection measures (Perrot-Maître and Davis, 2001, Catskill Corporation, 2001, New York Department of Environmental Protection, 2001).

Tradable Development Rights (TDRs), are a form of acquisition that shifts the cost of acquiring development rights to developers and future residents. Using this approach, developers who acquire the rights from areas designated for conservation are granted permits to build at higher densities than would otherwise be allowed, in areas specifically designated for development. The more successful initiatives are generally part of a comprehensive regional plan that justifies the designation of conservation and development areas. The best known example of the use of TDRs for the purpose of insuring the provision of freshwater may be the New Jersey Pinelands Development Credit Program in the US. A key concern is that the Pinelands occupy an area of sandy soil that covers a very large aquifer, and where massive development is otherwise expected (Collins and Russell, 1988) (New Jersey Pinelands Commission). A related approach is the U.S. Wetland Mitigation Banking program, in which wetland restoration may be funded through the sale of credits to developers, who may be required to purchase these as compensation for development impacts on wetlands that cannot be mitigated. The sale of credits also provides a way of concentrating wetlands restoration efforts in areas where they can be most beneficial (Liebesman and Plott, 1998, Salzman and Ruhl, 2002). A survey conducted in 2000 identified over 230 operating wetland mitigation banks, over half of which were commercial, and which involved the restoration of 16,500 ha in exchange for the development of 9500 ha (IWR, 2000, National Research Council, 2001).

Marketable Permit Systems (MPS) – These are similar in concept to TDRs, but, in this context, are applied to trades between point and non-point sources of pollution. Once allowable levels of pollution and resource use have been established as a matter of policy and regulation, MPSs provide a way to reduce the overall costs of compliance by allowing those who have higher compliance costs to purchase credits from those whose costs are lower, usually through intermediary organizations that register and verify credits. They may also be structured such that proceeds from the sale of permits are used to fund transfer payments, thereby shifting the cost of payments from the public to the private sector. 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 emission of pollutants. The second instead offers credits to those who reduce emissions or resource use beyond legal requirements or engage in specific conservation practices. Among the key policy issues is the choice of method for allocating initial rights to shares of the total allowable emissions, and rules regarding the transfer of those rights (Tietenberg, 2002). For example, rights may be allocated on the basis of existing uses, lotteries, auctions, or rules that define who is eligible. 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. A third policy issue is for stakeholders to reach an agreement as to what constitutes an equitable allocation of the burden of emissions reduction among point and non point sources, so as to avoid violating the "polluter pays" principle (King, 2003).

In the context of watersheds, several programs are under development in the U.S., though very few trades have actually taken place to date. Those few include one among non-point sources in the Dillon Reservoir, and one in Minnesota, in which the Minnesota Pollution Control Agency permitted the Rahr Malting Company to construct a downstream facility that would increase biological oxygen demand, in exchange for funding reductions of upstream non-point sources of phosphorus. However, trades are expected in a number of other basins as estimates of Total Maximum Daily Loads (TMDLs) are completed (Nutrient Net, 2004). TMDLs define permissible levels of emissions that are consistent with achieving water quality standards, and provide a basis for allocation of the burden of emission reductions.. In the Tar Pamlico basin in the state of North Carolina, an association of point source dischargers has purchased credits from the state that have been banked for use when their cap is exceeded, and that are intended for future use in funding Best Management Practices (NC NPSMP, 2004). A credit trading program is also being used for salinity reduction in New South Wales in Australia. In this case, an association of farmers purchases salinity credits from the State Forests agency of NSW, which in turn, contracts with upstream landholders to plant trees, which also provide carbon storage benefits (Perrot-Maître and Davis, 2001, State Forests of New South Wales, 2001, Sundstrom, 2001).

Certification and labeling (CL) – creates an incentive to adhere to specified management practices by providing consumers with the information necessary to make their choices of products more consistent with the values they place on ecosystem services. This may increase the market share of a product, and/or result in a price premium. However, producers are not always the ones who benefit from price premiums given the number of actors between them and those who ultimately purchase the products. This arrangement requires intermediary organizations to establish standards for labeling and to certify practices, all of which has a cost. An example is the Salmon Safe initiative, which certifies and promotes wines and other agricultural products from Oregon farms and vineyards that have adhered to management practices designed to protect water quality and salmon populations (Salmon Safe, 2003).

In addition to the sale of permits and voluntary payments based on contractual agreements, are a number of different kinds of sources of funds, ranging from user fees, to taxes, donations, and earmarked proceeds from sales of specific products. The appropriateness of particular sources will depend on whether or not it is possible to limit benefits to those who pay for them, and the scale at which benefits are detectable and significant.

For example, *User Fees* (UF) involve charges to users of a resource, based on the principle of "User Pays", which is only feasible when it is possible to limit benefits to those who pay for them. To be considered an instrument for protecting services, funds must also be specifically designated for conservation purposes. Intermediaries usually play a role in the collection of funds and their disbursement to landowners. These are often added to fees already paid by users for delivery of water. Examples of this approach include the cases of New York City, Quito Ecuador, Cauca Valley Colombia, where charges are added to existing water charges, and are designated specifically for funding upstream conservation measures (New York Department of Environmental Protection, 2001, Perrot-Maître and Davis, 2001, Echavarría, 2002, Echavarria, 2002). Fees may also be differentiated so as to support policy objectives of giving priorities to particular uses. For example, under the South African water law, water licensing charges vary by sector and do not apply to a certain amount of water that is reserved for basic human and ecosystem needs (DWAF, 1999a).

When it is not possible to limit access to benefits to specific users, *taxes* may be more appropriate. Transfer payments, such as the US Conservation Reserve program, tend to be funded through general tax revenue. In Costa Rica, the National Fund for Forest Financing (FONAFIFO), a program of payments for ecosystem services that includes protection of watersheds, is in part funded by a fuel tax, in addition to payments from beneficiaries. In Colombia, watershed management is in part funded through a 6% tax on the revenue of large hydroelectric plants. This tax also supports political decentralization, as it is a key source of funding for regional authorities, who have the primary authority for watershed management. This revenue is divided among the autonomous regional corporations and municipal governments. A portion of the allocation to the autonomous regional corporations goes to a general fund that supports management of watersheds in which there is no hydropower facility. In addition, 1% of funds invested by towns in water projects must be invested in watershed protection.

Donations from external sources may be important for addressing impacts associated with actions of external actors, such as those associated with timber concessions and other uses of government owned land, and with macroeconomic policies. Funding from external donors may also be necessary to finance the high transaction costs faced up front, in the beginning stages of an initiative, associated with feasibility studies, assessment activities, and capacity building, generally done through intermediary organizations.

Earmarked proceeds from the sale of specific products that are associated with special places and symbols, may simply be designated for the support of conservation objectives that have broad benefits, which may also influence consumer purchase decisions and have other benefits for the company or organization. For example, in the state of Maryland (in the U.S.), for an additional fee, car owners have an option of obtaining Chesapeake Bay license plates, which supports the Chesapeake Bay Trust – an entity created by the state legislature to support Bay restoration. The University of Maryland also recently began to designate 2% of the proceeds from the sale of items associated with the "Fear the Turtle" slogan towards the protection of the terrapin – the school mascot and official state reptile (Hsieh, 2003). The terrapin is generally regarded as an "ambassador" for the Chesapeake Bay ecosystem, as it relies on it's water quality, shorelines, salt marshes and tidal rivers, and has provided the basis for extensive public education and outreach regarding comprehensive approaches to conservation (Dunlap, 2003, Golder et al., 2000).

Most arrangements will consist of a package of various instruments. For example, TPs may be made through VCAs with sellers, using funds derived from UFs. Individual arrangements may also be part of a comprehensive plan intended to protect multiple services, which, in addition to those associated with freshwater, may include other benefits of forests, such as carbon storage, aesthetic values, and biodiversity. These are the four services for which forest owners who adhere to approved management plans are compensated in Costa Rica's FONAFIFO program, which is then able to sell them to different kinds of buyers. For example, hydroelectric companies and municipalities may pay for watershed benefits, tourism agencies for landscape beauty, and foreign energy companies may purchase carbon offsets. State Forests of New South Wales has also initiated an Environmental Services Scheme in which landowners are to be compensated through a credit scheme for multiple benefits of forests, including carbon sequestration, biodiversity, soil conservation, in addition to protection of water quality and offsetting the rise in salinity levels (State Forests of New South Wales, 2004). This approach may also be necessary when watershed benefits alone are not sufficient to offset the opportunity costs of forgone land uses (Landell-Mills and Porras, 2002).

Fundamental to all of these instruments is the need for a consistent set of criteria and a transparent process for decision-making with respect to the establishment of priorities for allocation of the funds. It is also important to keep in mind that market-based mechanisms are not a substitute for regulations, which are often necessary to create an incentive to find more cost effective ways of achieving compliance through the use of market mechanisms. For example, in the well-known case of New York City, the incentive for the city to negotiate with upper watershed communities and to pay for upper watershed conservation measures was created by new regulations that would have otherwise required much higher expenditures to construct a filtration plant (Perrot-Maître and Davis, 2001). Trading schemes rely directly on regulatory standards or caps, without which there would be no incentive for using market based approaches.

In general, benefits will be more tangible, and contractual agreements more feasible, at smaller scales, where links between causes and effects can be more readily established, 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 service, or of multiple services, there will be a greater need for government involvement. Larger scales also offer a larger pool of buyers and sellers but are harder to tailor to local conditions (Rose, 2002).

Challenges to Implementation

Implementation of market-based payment arrangements for watershed services will be difficult unless beneficiaries are both able and willing to pay for them. Willingness to pay will generally depend on perceptions and beliefs regarding the benefits of ecosystem services. In the long if not the short term, this may require a demonstration that there are measurable benefits that support livelihoods and overall well-being. It will also require the development of institutional arrangements that insure access to benefits by those who pay for them, over the relevant period of time. By providing compensation for inevitable trade-offs among conflicting interests and objectives, market-based instruments will often be one element in a more comprehensive management strategy. This section begins with an overview of the difficulties of demonstrating measurable benefits, and the use of scientific and other information to evaluate trade-offs. It concludes with a discussion of the various kinds of institutional arrangements needed to support payment initiatives.

Demonstrating measurable benefits

Assessment and verification of the benefits of ecosystem services, and the effectiveness of management actions taken to insure continued provision, are critical to building and maintaining stakeholder confidence and willingness to pay for them. In general, this aspect has received less attention than the identification of able buyers and systems for collecting payments (Pagiola et al., 2002). However, it can require gathering of data over time and significant commitments of funding and expertise that are not available until an initiative is well established. Rules of thumb may be sufficient in the initial phase, providing that initiatives are designed to include monitoring and assessment that allows them to be improved over time.

Key questions that should be answered in assessments, or for which they should at least provide a working hypothesis are:

1. What ecosystem functions support the provision of specific benefits and what are their key parameters, or, how can they be measured or approximated? How does land use in the watershed affect these functions?
2. What is the direction and magnitude of changes in parameters of interest? and
3. At what spatial and temporal scales can these changes be detected?

Investigation of changes in the water balance and the impacts of these changes on the flow regime, provide a framework for quantifying changes in watershed conditions relative to management objectives. These objectives, or points of interest, may include:

- Total flow yield,
- Attenuation of peak or flood flows,
- Maintenance of minimum low or dry season flows,
- Protection of water quality,
- Recharge of groundwater, and
- Protection of biodiversity

(Tognetti et al., 2004).

The most significant changes in the water balance are linked to extreme and randomly timed events. For example, most sediment and pollutants are transported during storms, that also produce the periodic high flows necessary for maintenance of channels, riparian areas, wetlands and coastal mangroves. Such events are the dominant process in upstream hillslope areas, which make up 70-80% of catchment areas, have tightly coupled land and water interactions, and have greater variability in discharge. This is in contrast with downstream fluvial processes that are more continuous and are dominated by regular flood pulses and movement of bedload within the channel as well as cumulative impacts of hillslope processes (Gomi et al., 2002). Therefore, it is more important to identify known ranges and patterns of variability and uncertainty than to identify average values. These ranges can then be used to put lower bounds on unknown thresholds of resilience – which refers to the range of variability within which changes can be adjusted to (Tognetti et al., 2004).

Human uses also play a significant role in the hydrological cycle, and should be accounted for so that their cumulative impacts can be distinguished from natural variation, and so as to be able to distinguish physical from economic causes of scarcity - which has implications for the kinds of measures that will be effective. Economic causes of scarcity imply the need for changes in allocation among human uses. Physical causes of scarcity imply the need for changes in the flow regime, which refers to allocation between ecosystems and human uses, and changes in land use practices that affect the timing and routing of flows. This should be accounted for on a seasonal basis - to the extent water is limiting during dry periods, there will be generally be a greater Willingness-to-Pay for it (Tognetti et al., 2004).

Basin-wide seasonal Actual Evapotranspiration (AET) is a principal component of the water balance. It is also a key source of uncertainty, because it is a function of numerous variables that include climatic factors, vegetation and land use. Given the heterogeneity of landscapes, this is difficult to estimate. However, it is important to consider land use practices and landscape characteristics such as riparian zones, that have disproportionate impacts on levels of runoff and sedimentation, and that may account for significant differences in AET (van Noordwijk et al., 2004).

Because of such heterogeneity, land use impacts on flows of water and sediment are generally best addressed at small scales, at level of hillslopes and patches, at which they can be detected. With some exceptions (e.g., suspended sediment and microbial contamination), water quality impacts, diversions, and impacts of large infrastructure are more appropriately addressed at basin-wide scales, which allows for consideration of a broader range of trade-offs..

The measurement of benefits should ultimately provide a basis for estimating relationships between the extent of changes in land use and the amount of benefits provided, so as to provide justification for payment amounts. Given inherent uncertainties, it may be sufficient to rank the relative values of land uses and practices for achieving desired outcomes, and to identify opportunity costs and conflicts associated with changes in land use practices. As a general rule, greater precision and more data will be needed when it is unclear whether or not benefits exceed costs. However, this depends also on what costs and benefits are considered – which is largely a political and institutional question.

Identifying and evaluating trade-offs

Identifying the economic significance of changes in watershed processes provides a basis for identifying actual or potential trade-offs between meeting various objectives. This will require consideration of downstream land uses and stakeholder vulnerability, which depends on what options they have, and on their opportunity costs. Just as with biophysical processes in watersheds - vulnerability, options available to stakeholders, and conflicts, become more transparent under extreme conditions. Impacts may be both positive and negative, depending on what is measured and valued.. For example, whether floods increase or decrease welfare will depend on the presence of development in flood plains, and valued habitats that rely on periodic floods.

An example of an extensive assessment that was done regarding relationships between land use and hydrology is a study of the Arenal basin of Costa Rica, which examined the marginal values of changes in flows of water and sedimentation for a downstream hydroelectric plant. In this particular case, it was found that, while sediment from pasture compared with forested areas did have a cost in terms of lost hydroelectric production, ranging from $35 to $75/ha, this was exceeded by the benefits to the facility of increased water yield from pasture areas, which ranged from $250 to $1,100, depending on the type of forest area cleared - the greatest yield of water appeared to be associated with fragmented cloud forest areas which have the highest rates of interception of precipitation (Aylward and Echeverria 2001). This (counter-intuitive) result occurs in part because the Arenal reservoir is an interannual regulation reservoir, in which hydroelectric production depends on total flows, and is therefore largely independent of dry season flows. Given that the reservoir also contains a large amount of dead storage space, sedimentation also provides a benefit – when it fills this space and displaces water upward, which makes more water available for production.

Given the high economic values associated with forest clearing in the Arenal study, ranching was also found to produce higher net present values than was offered by the government for reforestation and thus to be more economically efficient. On the other hand, these costs and benefits were not uniformly distributed. 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 depended 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 who, not coincidentally, also tend to disproportionately occupy the steepest slopes (Aylward and Fernández González 1998). The scale of the study did not permit it to answer questions regarding the more localized impacts of forest clearing on landowners residing within the upstream area.

In contrast with the results at Arenal are those of another Costa Rican reservoir that is part of the La Esperanza hydroelectric facility, which is smaller, depends on dry season flows, and has higher costs associated with sedimentation. In that case, payments from the hydroelectric company to the Monteverde Conservation League, the sole owner of the upper watershed area, are simply based on the value to the company of reducing uncertainty that would accompany any change in upstream land use (Rojas and Aylward 2002).

Because of natural variability, landscape heterogeneity and human activities, it will be difficult or impossible to unequivocally establish links between land use and water yield at the watershed scale, even with a full analysis of land and water relationships. For example, in the case of Arenal, there are a number of land uses and vegetation types that include pasture, forest with different canopy heights, cloud forest with different degrees of fragmentation, all of which are changing over time, are a key parameter in estimating water balances, and are therefore a major source of uncertainty. A field study in the Arenal basin found the highest capture of precipitation in fragmented primary cloud forest, of 12% compared with a negative gain of 11% in intact high primary forest. However, during the dry season, yields were positive for all types of forest, when water availability is most limiting and therefore has higher economic value – with gains ranging from 15 to 53%, (Fallas 1996). Other potentially significant sources of variability in the interception of precipitation in cloud forests, often not accounted for, are the position of the slope in relation to moist winds and storm intensity, which affects the amount of water held in the forest canopy (Bruijnzeel 2001). Although water quantity impacts associated with forest clearance are often ambiguous, water quality impacts can generally be expected to be negative (Aylward, 2002).

Recent reviews of the scientific knowledge base that has supported many initiatives in Costa Rica suggests that management is generally limited by the lack of reliable and precise information on forest water linkages (Pagiola 2002; Rojas and Aylward 2004). Instead, most are based on conventional wisdom, secondary sources of information, and also, selective references to more balanced literature reviews on forest hydrology. In other words, regardless of the source material, information used in decision-making tends to invariably support statements that protection of forests will increase water yields (Rojas and Aylward 2004). In some cases, such as in the Arenal basin (Castro and Barrantes, 1998) and in Heredia (Castro and Salazar, 2000) the values of watershed protection are calculated based on the opportunity cost of returning cleared land to forest cover, with no attempt to model and assess links between land use and hydrology, and to estimate the marginal values of water in specific consumption and production activities. In other words, estimates of opportunity costs of land under alternative uses is used to infer the benefits of forest hydrological services. In the latter case, payments are also justified based on the statement that “Costa Rican society positively correlates the presence of forest with the supply of hydrological services”.

Faced with these and other kinds of uncertainty, such as data that is rarely if ever complete, reliance of models on average values, the random timing and disproportionate significance of extreme events - decisions to protect watershed services are often based on the costs of alternative courses of action rather than on the value of changes in land use practices. For example, in the New York City case, the decision to invest in upstream conservation and upgrading of infrastructure was based on the avoided cost of building a filtration plant that would have otherwise been required to meet a new regulatory standard designed to insure safe drinking water. However, the relationship between changes in land use practices and resulting water quality remains the subject of ongoing research.

Other factors that have been used to inform the analysis of trade-offs and to justify decisions and influence WTP have included:

- costs of implementing management plans,
- regulatory costs avoided,
- reduction of uncertainty associated with proposed changes in land use;
- individual WTP in the form of user fees and purchase of certified products;
- political WTP as indicated in national budget allocations of tax revenue;
- landowner WTA compensation or cost of supplying the service; and
- consistency with comprehensive management plans.

Given the period of time that may be required to detect the results of changes in management practices, and the impossibility of obtaining complete information, it will usually be necessary to begin with rules of thumb. These can provide a basis for constructing plausible scenarios, or serve as working hypotheses, to be tested in an adaptive approach to management, in which they are continuously questioned and revised as new information becomes available and in response to unanticipated consequences.. Initiatives are likely to be more effective in the long term when uncertainty is made explicit and initiatives are also informed by a process of monitoring and assessment, which are essential as a feedback component in the design of any initiative.

Ultimately, the extent of information needed for decision-making will depend on what is required to justify an expenditure in a particular case, or, in other words, to convince the buyers. To the extent that uncertainty regarding links between actions and outcomes can be reduced - or at least made transparent so that it can be factored into decision-making, stakeholders will have greater economic incentive to participate in these initiatives and in helping to insure that objectives are actually achieved. A key challenge then is to develop greater capacity for conducting site-specific assessment of functions that support services, of threats they will be degraded or disrupted, and to use these to develop feasible and effective response options.

Development of appropriate institutional arrangements

As discussed in the introduction, willingness-to-pay is inextricably linked to confidence in the effectiveness of management actions as well as of the institutional arrangements needed to insure access to benefits by those who pay the costs of management actions. Absent such arrangements, economic value is no more than hypothetical, as there would be no incentive to take actions needed to insure provision of the service. Institutional arrangements are essentially the "rules-of-the-game" that are needed to resolve conflicts among competing demands on any limited resource so as to avoid it's depletion or degradation. They may take several different forms, addressed in this section, chief among which are property rights, which define the rights and responsibilities of all players, and have implications for the distribution of costs and benefits. A second is the decision making process, which may or may not allow for effective participation of all concerned, and the process of assessment – which includes both the gathering and dissemination to users, of information needed to support decision-making. The feasibility of these arrangements will depend largely on their transaction costs, and on the broader social and economic context in which they are embedded.

Institutional arrangements also serve the purpose of reducing transaction costs by facilitating transactions among numerous buyers and sellers, and, where necessary, collective action. For example, given that areas of high erosion tend to be marginal areas such as steep slopes, or common areas such as roads and paths, individual landowners will have little incentive to invest in improvements, which makes these areas de facto, open access (Swallow et al., 2001). In this kind of situation, provision of services may require incentives for collective action by communities. When there are numerous stakeholders dispersed over large upper watershed areas, the establishment of intermediary organizations who can negotiate on their behalf, serve as advocates for the rights or marginalized stakeholders, and provide technical assistance is an important way of reducing transaction costs and making an initiative feasible.

Rights to watershed services and responsibilities for their provision

Property rights play an important role in creating appropriate economic incentives because they determine who has access to benefits and also define responsibilities for costs, or actions needed to insure the provision of benefits. 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 et al., 1994).

Appropriateness of property regimes depends on whether the incentives they create are consistent with social objectives. For example, rights to water based on historic use or “prior appropriation”, which usually require also that the water be used in ways that are considered socially beneficial, was consistent with social objectives in the 1800s, of promoting development in the western United States. 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 the water and to develop water markets, which can provide incentives for more efficient allocation among various uses (Meinzen-Dick and Bruns, 2000). In an open access situation, the incentive is simply to consume resources before someone else does.

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 the development of new and more appropriate institutional arrangements. 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.

Allocation of costs and benefits

Equitable distribution of costs and benefits is an important aspect of feasibility in that, if significant stakeholders are excluded or disadvantaged, and regard existing rights as inequitable, they will have little incentive to cooperate in their enforcement. On the buyer side, some studies have found differences in willingness to pay 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 low prices paid for agricultural commodities, direct payments for providing services of maintaining the landscape and water quality may also be seen simply as recognition of the values of environmental services in addition to those of agricultural commodities.

Equity issues may also present an obstacle to nutrient trading between emitters of nutrients from point and non-point sources , as it has been suggested these may not be regarded as fair by point source emitters who have already invested in point source reductions (King, 2003). Agreement on Total Maximum Daily Loads (TMDLs) that equitably allocate load restrictions between point and non point sources, as a basis for the initial allocation of permits, may help to overcome this.

In a number of cases that have been evaluated, it was found that payments for watershed have disproportionately benefited those who own larger tracts of forests or forest plantations, excluding smaller and marginalized landholders, who tend to occupy the steepest slopes, who don't have large forest areas that they can be compensated for protecting, and may not have the property rights needed to gain access to benefits. They also tend to exclude activities such as agroforestry and organic farming. This suggests the need for broader criteria for allocation of payments, to as to include smaller landholders and better support livelihood strategies (Rosa et al., 2003).

Participation in Decision-making

Acceptance and cooperation of both buyers and sellers may ultimately depend on whether all of the relevant stakeholders cooperate, how funds will be spent, and whether stakeholders are able to effectively participate in allocation decisions, all of which are issues of governance. For example, 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 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). A study of water resources management in Cyprus 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 bird migration route (Koundouri et al., 2003).

A key obstacle to effective participation of stakeholders in decision-making has been the association of large water resource infrastructure with the need for highly centralized management authority. In the case of large infrastructure projects, decision-making takes place at national levels, and is largely driven by geopolitical considerations in which local stakeholders have little in any voice. Because of environmental and institutional heterogeneity, highly centralized authorities tend to have a limited capacity to respond to livelihood concerns 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, which can increase the capacity to respond to factors such as variations in rainfall and crises associated with extreme events, as well as to mediate conflicts. Transfer of rights and sharing of benefits 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 be inextricably linked with efforts to and to achieve democratic forms of decentralization, or to “pry open… local democratic space” (Kaimowitz and Ribot, 2002).

A second obstacle to effective participation is the gathering and dissemination of information needed to support decision-making. Given the site-specific nature of watershed services, this presents the challenge of developing an integrated and place-based approach to assessment. Given that adequate information seldom exists in advance of an initiative, and that complete information is unobtainable, this is particularly important in the implementation phase, so as to identify gaps between policies and practices. This provides a basis for course correction, as new information becomes available and as lessons are learned, and for engaging stakeholders in identifying feasible options.

Socioeconomic and political context

Ultimately, the development of market-based arrangements for watershed ecosystem services needs to also be considered in the context of a broader trend of institutional changes in water resource management. Among the key elements of this trend are (Saleth and Dinar, 1999) (Bauer, 2004):

1. a shift from the development of new water supplies through infrastructure development, to the reallocation of existing ones, and
2. efforts to improve cost recovery both for
    a. operations and maintenance of infrastructure, and
    b. to cover the costs of conservation management and research activities.

In theory, recovery of these costs could increase the capacity of governments to deliver basic water supplies and sanitation, and the capacity of ecosystems to provide freshwater supplies. However, cost recovery faces a major obstacle in that major water users, particularly in agriculture, which accounts for approximately 70% of water withdrawals worldwide (United Nations, 2003), are also accustomed to high subsidies and pay only a fraction of the costs of operations and maintenance alone. In urban areas, inability to recover costs from those who are served, reduces the capacity of the government to extend services to the poor, who must then pay more to obtain water from trucks. Reallocation of existing supplies therefore faces significant political constraints, and may need to be linked to broader macroeconomic reforms that tend to be associated with crises or sweeping political changes, such as the end of apartheid in South Africa which, among other things, made it possible to make significant reforms in the country's water law (see Box 1).

Conversely, given the role of water in most if not all sectors of the economy, the development of payment arrangements can have implications for broader policy reforms. For example, potential spin-off benefits associated with the development of markets for watershed services include: clarification of property rights, stakeholder cooperation in other areas important to livelihoods 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.

Although development of appropriate supporting institutional arrangements can have high transaction costs, 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.

Box 1: National Water Act of South Africa

The South Africa National Water Act is among the more innovative in that it explicitly seeks to recover costs both for operation and maintenance and for conservation, management and research, as well as for meeting basic human subsistence needs. Therefore, water charges do not apply to minimal levels of water for ecosystem and human subsistence needs (DWAF, 1997). Beyond these basic needs, it requires the registration of water uses. Licensing and payment of fees are required for water service authorities, industrial uses, irrigation, and for activities that reduce streamflow, such as tree plantations. Licensing applications are evaluated for consistency with a number of criteria including individual catchment management strategies that are prepared in accordance with a national water strategy (DWAF, 1999b). Implementation has not been without problems. Among these are tensions with existing riparian rights holders, hardships associated with the withdrawal of subsidies from irrigators, and the inability of many of the poor to pay even minimal amounts for water delivery. Whether these problems can be resolved, and whether licensing can truly serve as a reallocation mechanism, may depend less on formal changes in the law than on the strength of public participation, and the bargaining power of the poor (Schreiner and van Koppen, 2000).

Conclusion

The specific services provided by watersheds will depend on how they are managed, and whether impacts have economic significance. This in turn depends on downstream uses of land and water, on the scale at which impacts can be detected, and on what options are available to stakeholders for maintaining their livelihoods. Therefore, market-based instruments will not solve all problems of watershed degradation, but are important tools that should be used as components of broader management strategy.

In the past, efforts to manage freshwater ecosystems to meet multiple objectives has been driven by the more dominant and tangible economic interests, typically navigation and hydropower (Barrow, 1998). More recently, in a somewhat similar fashion, efforts to protect watersheds using market-based instruments, have been driven by the interests of those most able to pay such as municipal water suppliers and hydroelectric facilities, and have placed emphasis on protection of forests and protected areas. This approach tends to overlook management practices that may have greater impacts than forest clearing, and to exclude the most vulnerable populations from access to benefits. A significant obstacle to cost recovery is that the most significant users, e.g., agriculture, are accustomed to subsidized prices.

The use of market-based instruments to recover the costs of watershed management activities needs to be considered in the context of basin-wide management objectives, and should be part of a package of approaches constructed to identify the relative values of multiple ecosystem services of watersheds as a basis for setting conservation priorities, and to resolve conflict among multiple uses. This can provide a basis for engaging stakeholders in identifying a broader range of management options that are consistent with meeting the broader objectives of ecosystem management to support human livelihoods and general well-being. It may also increase their confidence and willingness-to-pay and to cooperate in management activities.

Experience to date with the use of market-based instruments to cover the cost of actions needed to insure continued provision of watershed services suggests that little is known about their effectiveness for actually delivering ecosystem services. This is in part because of the time lag between management activities and their outcomes in a watershed context, and in part because less attention has been given to development of an independent and transparent process of assessment.

Given the heterogeneity and constant change in ecosystems and in human institutions, the site specific nature of watershed processes – which are dominated by randomly timed and extreme events, and the difficulty of linking multiple causes and effects, or predicting outcomes, an adaptive approach to management is required. On-going assessment to support decision-making is a critical component of such initiatives. However, assessments based on generalizations can only provide some rules of thumb and working hypotheses from which to begin. Perhaps the most significant challenge therefore is to develop the capacity for a place-based approach to assessment, which is necessary to identify ecosystem functions that support provision of valued ecosystem services in a specific context, and to select payment and institutional arrangements that are feasible and appropriate to that context. To be effective, market-based initiatives also need to be viewed as part of a long term process of building appropriate institutions, and in the context of broader issues of structural reform.

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