

## chapter six

# Policies for encouraging forest restoration

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### 6.1 Introduction

Throughout the 20th century, many countries created national parks, forests, nature reserves, and sanctuaries to provide benefits that are underproduced on private lands. Private lands are now especially valuable for providing ecological services that public lands cannot provide, due to the increasing demands for all uses and the political and economic conflicts associated with allocating public lands between competing uses (e.g., recreation, watershed protection, biodiversity conservation, wildlife habitat, commodity production) (Kline et al. 2000). In many countries, the supply of public lands may not be adequate to ensure desirable flows of beneficial ecosystem services. Furthermore, because many ecological processes cross ownership borders, enhancing the flow of benefits requires management at a broader, landscape scale and with the participation of both public and private landowners (Kline et al. 2000; Gottfried et al. 1996; Boyd and Wainger 2002a; Johnston et al. 2002).

Many ecological services are public goods (goods or services for which it is hard or impossible to exclude from benefiting those who do not contribute to paying the costs of producing the good) or externalities (costs or benefits that are inflicted on or received by others, that are not reflected in market transactions). Ecological services typically are underproduced on private lands because the landowners bear the costs of restoring, preserving, and managing their lands for ecological services as well as the opportunity costs of foregoing alternative income-producing activities, while all members of the community enjoy the benefits for free. As a result, governments worldwide have been experimenting with policies and programs to encourage forest restoration, including a combination of activities on public lands and regulations and incentives to private landowners.

Deciding on the best combination of policies for encouraging restoration requires determining which ecosystem services are amenable to market solutions, which require

government intervention, and which require a combination of government and market approaches. Ecosystem services that can be efficiently supplied by markets must provide goods or services with some commercial value or for which a commercial value can be attached. Unless the landowner can appropriate some of that value, however, market solutions are infeasible and government intervention is required (Heal 1999).

Ecosystem services are frequently a combination of public (if provided for one, they are provided for all) and private goods. The extent to which they are public goods determines the necessity for government intervention. Watershed restoration projects provide an example of mixed public and private goods. Water quality is a public good because if it is produced for one user of the watershed, it is produced for all, no matter who bears the cost. Because some individuals and communities can be excluded from consuming water, it can also be considered a private good (Heal 1999). Indeed, during the recent droughts that have inflicted the western and southeastern U.S., it was not uncommon to see news accounts of communities selling water and water rights to other communities with shortages of water. Likewise, the protective role of forest ecosystems in a watershed produces public goods (e.g., biodiversity, carbon sequestration, wildlife habitat), whose production may or may not conflict with private goods produced from the ecosystem such as timber.

The situation becomes even more complicated when considering ecological processes that produce ecosystem services at a landscape scale. Individual owners acting alone are unlikely to produce the socially optimal amount of ecosystem services. Just as prices in a perfectly competitive market are determined by the interactions of all buyers and sellers in a market, the mix of commodities and services produced from an ecosystem or watershed depends on the spatial pattern of land-use decisions made by all the landowners in that ecosystem. These "economies of configuration" (Gottfried et al. 1996) suggest that policies and programs designed and implemented at a landscape level rather than with individual ownerships are more likely to produce optimal quantities of ecosystem services, unless the effects of landowners' decisions can be separated spatially.

Gottfried et al. (1996) analyzed the ability of markets to create optimal landscapes and demonstrated that market forces in decentralized, unregulated economies are inadequate for optimizing ecological services at the landscape level. An optimal market solution would require that all public and private landowners compensate each other for the production of all externalities, both positive and negative. Unfortunately, this optimal situation requires intimate, quantifiable information, knowledge, and monitoring of the interrelationships within and between ecosystems and between ecosystems and economic systems, a level of knowledge that is not yet possible. Even if possible, implementation would likely be prohibitively expensive.

Incentives to individuals (e.g., taxes or subsidies) for restoration will usually suffer from scale problems (Gottfried et al. 1996) because the location of each ownership within a landscape determines, to a large extent, the production of a landscape's mix of goods and services, and landowners jointly (but in different ways) affect the landscape's ecological functions. Therefore, in landscapes with multiple ownerships that have spatially varied impacts on the ecosystem, markets will most likely fail to provide adequate ecosystem services, even in the presence of traditional methods of internalizing externalities (e.g., taxes and subsidies). This suggests that a combination of government and market interventions will usually be required to ensure successful and efficient ecosystem restoration efforts across a landscape (Gottfried et al. 1996).

## 6.2 *Costs and benefits of restoration policy options*

Governments have used a variety of mechanisms to encourage ecosystem restoration. These include tax incentives, subsidies, and cost-share programs; purchase of conservation

easements; fee-simple purchases; regulations restricting land use; tradable development rights; and cooperative/collective efforts (Alberini and Segerson 2002; Boyd and Simpson 1999; Cubbage et al. 1993; Granskog et al. 2002; OECD 1999). Table 6.1 provides a qualitative overview of the relative costs and benefits of alternative restoration policies, which are elaborated in this section.

The fundamental decision for policymakers is whether the government has both the ability and will to impose unwanted costs on landowners through either mandatory or voluntary approaches to encourage restoration. With mandatory approaches, governments impose net costs on landowners, who are therefore worse off financially than in the absence of the policy. Landowners will participate in voluntary programs if the total amount of benefits they receive from the land and government (both financial and nonfinancial) are at least as high as they receive without participating. For voluntary approaches to succeed, the landowner must perceive some gain or at least no net loss from participating (Alberini and Segerson 2002).

No matter which mechanism is used, the social cost of restoring and preserving ecosystems is the value of the lost income from whatever economic activities are foregone as a result of restoring and managing the land for ecological services. In other words, the social cost of restoration is the difference between the value of the land in its highest and best private use and its value following restoration. This cost is constant for individual properties no matter which type of policy or program is used to encourage restoration (Boyd and Simpson 1999). However, these opportunity costs will vary with the options provided to landowners to choose which parcels of their lands will be used for restoration and which for economic activities.

Available policy and program options, however, differ in the size of the transaction costs associated with the institutional, organizational, and informational requirements for implementing the policy. They also differ in who pays for the restoration of ecological services that benefit the whole society. If restoration is mandated by regulations that restrict and/or require certain activities on private lands, the private landowners pay. The costs of outright (fee-simple) purchase and purchase of conservation easements on private lands by the government are paid by taxpayers. Developers and future landowners pay under systems of tradable development rights. Under voluntary tax incentive and cost-sharing programs, both the landowner and taxpayers bear the costs. As Boyd and Simpson (1999) state,

Reasonable persons may differ regarding which groups are more morally deserving of bearing or escaping the burden of payment. But someone must pay.

### 6.2.1 *Mandatory approaches*

Mandatory (or regulatory) approaches use laws and policies to either dictate specific land management actions or otherwise limit or control how landowners (private and public) manage their lands. Granskog et al. (2002) list 13 federal statutes and 6 types of state laws regulating forest management in the southern U.S., while a 1991 survey identified 359 local ordinances regulating forestry activities in the eastern U.S. (Cubbage et al. 1993). Forest and land-use regulations vary considerably across Europe, but most European countries have stringent laws regulating timber harvesting, environmental protection, forest conversion, and restoration. In contrast to the U.S., however, most private lands in Europe are required to allow public access (Cubbage et al. 1993). Regulation is sometimes viewed as an efficient way of producing or protecting ecological services, because, in contrast to other approaches, fewer intervening institutions (e.g., tax assessment and collection agencies or markets) are usually required. Monitoring and enforcement costs,

Table 6.1 Summary of Qualitative Levels of Costs and Benefits of Alternative Restoration Tools

Policies	Costs					Who Pays?	Benefits
	Transactions	Monitoring	Enforcement	Opportunity	Political		
<b>Mandatory</b>							
Land-use regulations	Medium	High	High	High	High	Landowners	High
Compensatory mitigation	High	High	High	High	Medium	Landowners and developers	Low-medium
Mitigation banking	High	Medium	Medium	Low	Low	Landowners and developers	Medium
In-lieu mitigation fees	Medium	High	High	Low	Medium	Landowners and developers	Low
Fee-simple purchase	High	Low	Low	High	High	Taxpayers	High
Easements	High	Medium	Medium	High	High	Taxpayers	High
<b>Voluntary</b>							
Subsidies/cost-share	Medium	Medium	Medium	Medium	Low	Taxpayers and landowners	Low-medium
Tax credits	Medium	Medium	Medium	Medium	Low	Taxpayers and landowners	Low-medium
Fee-simple purchase	High	Low	Low	Low	Medium	Taxpayers	Low-medium
Easements	High	High	High	Low	Medium	Taxpayers	Low-medium
Cooperative-collective actions	High	High	High	Low	Low	Landowners	Medium-high

however, are potentially very large, and regulations can often be politically difficult to pass and implement as landowners may resist relinquishing rights to how they manage their land.

Among the most common regulatory approaches for mandating restoration and preservation of ecosystems are the so-called compensatory mitigation regulations, also referred to as "no-net-loss" policies. Mitigation is the creation, enhancement, or restoration of habitats or ecosystems in response to an action by the landowner that impacts negatively on the services provided by the ecosystem. Although the U.S. initiated compensatory mitigation in 1972 (Clean Water Act: 33U.S.C. 1344 [1972]), the European Union and its member states have also recently implemented a variety of no-net-loss policies (Ledoux et al. 2000). Compensatory mitigation regulations have been primarily directed at wetlands, but they are also applicable to a wide variety of ecosystems and habitats (Fernandez 1999; Ledoux et al. 2000).

The three general mechanisms for implementing compensatory mitigation actions are landowner or permittee-responsible, mitigation banks, and in-lieu fees (NRC 2001). Early compensatory mitigation actions in the U.S. encouraged on-site mitigation under the direct responsibility of the landowner or permittee and allowed developers to simultaneously develop and attempt restoration on the same site. Critics argued that on-site mitigation produced scattered small islands of restored ecosystems that were too small to adequately compensate for the loss of landscape-level ecological services, and that temporal losses of wetland functions were too large when developers were allowed to mitigate contemporaneously with development. The early mitigation efforts were also criticized for failing to protect high value wetlands, to enforce minimum mitigation requirements for replacement habitats, and to enforce and monitor compliance (Ledoux et al. 2000; NRC 1992, 2001; Race 1985; Race and Fonseca 1996; Reppert 1992; Roberts 1993; Zedler 1996).

Mitigation banking and in-lieu fee programs were developed in response to problems with individual on-site, postproject, landowner-responsible mitigation. Private firms began producing wetland credits for sale in the early 1990s, and by 1995 U.S. agencies issued guidelines for approving the sale of wetland mitigation credits (Fernandez and Karp 1999). Mitigation banking provides for advanced compensation of unavoidable habitat loss by creating, restoring, or enhancing large areas in the same watershed prior to future development and as a precondition for future development. Mitigation banks are usually relatively large blocks of restored, created, or enhanced ecosystems that compensate for ecological impacts from particular development projects, but also act as a repository with credits available for compensating future development projects (Reppert 1992).

When developers or landowners want to develop an environmentally sensitive area, they are required to have credits in hand from investing in a completed restoration site. Credits are denominated as Habitat Units (HU), which are calculated as the product of the number of species or ecological functions per ha times the number of ha being developed. The number of credits required depends on the success of the restoration project (Fernandez and Karp 1999). In essence, mitigation banking sets up a market in which restored ecosystem values and functions are quantified as credits, deposited into an account, and later purchased by developers or landowners when regulations require compensation for authorized losses of ecosystem function (Ledoux et al. 2000).

Mitigation banks offer several advantages over individual on-site, postproject mitigation. First, failure rates should be lower because temporal losses of ecosystem function and uncertainty over the success of mitigation efforts are reduced when restoration occurs prior to implementation of the development project. Second, it is usually ecologically advantageous to have larger, consolidated mitigation sites instead of the smaller, isolated

islands of restored ecosystems that individual on-site mitigation tends to produce. Third, economies of scale (financial, regulatory, ecological, scientific) associated with mitigation banking should lead to more cost-effective and superior mitigation projects (NRC 2001).

The third approach to compensatory mitigation involves the use of in-lieu fees as a payment to natural resource managers for implementing specific or general restoration projects. U.S. agencies require the sponsors of in-lieu fee accounts to enter into agreements similar to banking instruments to define appropriate conditions for in-lieu fee mitigation.

Historically, U.S. agencies have generally preferred on-site mitigation to off-site mitigation. For example, 75% of compensatory mitigation projects in 1998 were implemented on-site, while 9% occurred through mitigation banks and the remainder through other mechanisms such as in-lieu fees and in-kind exchanges (NRC 2001). However, when off-site mitigation is approved, U.S. agencies usually prefer mitigation banks to in-lieu fees as long as there are credits available from an approved mitigation bank in the service area. This is based on criticisms that in-lieu fee programs were allowing compensation outside the impacted watershed, funds were being used for nonmitigation activities, programs resulted in out-of-kind compensation, and preservation was too often allowed as a compensatory action. Nevertheless, in-lieu fees are preferred when in-kind mitigation is not available from a bank or the bank only provides preservation credits and the in-lieu fee arrangement offers in-kind restoration (NRC 2001).

The National Research Council (NRC 2001) recently completed a thorough review of restoration projects in the U.S. required under the Clean Water Act (CWA) Section 404. Success was spotty. The NRC found some sites meeting established criteria and showing promise of becoming fully functional ecosystems and watersheds. Others were never initiated, or if initiated, were poorly designed, carelessly implemented, or both. Compensation sites were often placed in landscapes that did not provide enough hydrological impacts and/or associated communities to adequately compensate for the loss of ecological services. At most sites, the absence of long-term monitoring and legal and financial assurances for long-term protection suggest that the mitigation efforts may not be sustainable over time (NRC 2001).

Based on this analysis, the NRC recommended that the U.S. mitigation programs develop and implement a watershed-based approach. The recommended approach would use watershed assessments to determine existing and reference conditions, incorporate results from these assessments into resource management planning, and foster collaboration among landowners in a watershed. Some state and federal agencies have experimented with a process called Advanced Identification (ADID) to plan mitigation sites. The ADID process sets watershed restoration priorities and designs mitigation strategies by assessing the functions and values of sites in a watershed and identifying the most degraded, least valuable sites for future development and the more valuable sites for restoration. Since the public (rather than the developer) pays for the substantial costs that are incurred, ADID can be considered a subsidy to landowners to produce public benefits while satisfying the constraint of no-net-loss criteria (Fernandez and Karp 1999).

Because on-site, permittee-responsible mitigation will likely continue to dominate mitigation efforts in the near future (due in large part to the scarcity of mitigation banks and in-lieu fee programs), the NRC recommended that agencies establish and enforce clear compliance requirements to assure that projects are initiated at least concurrently with the development action, science-based design criteria and adaptive management are used to implement and construct mitigation projects, performance standards are specified and attained before permits are approved, and the permittee provides a government or nonprofit stewardship organization with an easement on or title to the restored site and cash payments large enough to ensure long-term monitoring, management, and maintenance of the site (NRC 2001).

In Austria, Weiss (2000) compared the use of regulations, subsidies, and an educational program to promote mountain forest restoration. Although the regulations (Forest Act of 1987) require the landowner to bear the cost of restoring and ensuring the existence of mountain forests under a polluter-pays principle, it also requires downstream and downhill beneficiaries to pay for additional costs to implement measures or management actions for specific protective functions such as control of avalanches or soil erosion. However, to avoid the transactions costs of resolving conflicts over who owes whom what, the regulations were rarely implemented. Instead, the forest authorities switched primarily to forest restoration subsidies, which are politically more palatable, less conflictive, and therefore less costly to implement.

### 6.2.2 *Voluntary approaches*

Voluntary approaches for environmental policy and programs have become increasingly popular throughout the world (OECD 1999). In many countries, landowner resentment toward increasing regulation of private property has driven governments to initiate policies and programs that utilize positive incentives to encourage cooperation with government agencies, or that appeal to landowners' sense of shared responsibility toward environmental stewardship, in order to encourage voluntary, collective restoration projects (Gottfried et al. 1996; Hodge and McNally 2000). The increased flexibility of voluntary approaches for both landowners and government agencies may result in cost savings. Landowners may choose the most efficient and cost-effective means to restore ecosystems for their particular site rather than implementing dictated restoration strategies under mandatory approaches. In addition, enforcement and monitoring costs should be lower when landowners volunteer to participate. Critics suggest, however, that relying on voluntary approaches may result in the restoration of the lowest quality lands that would not necessarily produce the highest ecological and social benefits.

Landowners or firms may voluntarily comply with environmental regulations and programs for a number of reasons. The combined financial incentives and value of improved environmental services to the landowner may be sufficiently high to offset the costs of participating. Individuals or firms may be motivated to voluntarily comply in order to project a positive image as responsible environmental stewards with their neighbors, customers, or communities. Fear of stricter regulations in the future may also motivate voluntary compliance or participation (Arora and Cason 1996; Kline et al. 2000). However, voluntary approaches based on exemptions from existing regulations or taxes probably provide greater and more credible incentives than threats to enact new regulations or taxes should voluntary approaches fail. Several authors have suggested that voluntary approaches backed by a strong regulatory framework are more likely to succeed (Alberini and Segerson 2002; Khanna 2001).

Voluntary participation in ecosystem restoration, preservation, and conservation occurs through unilateral and bilateral government programs, individual or collective initiatives, or both. The most common are unilateral government programs in which the government offers financial incentives, such as tax credits or subsidies, in exchange for certain land management actions by landowners. The second type consists of bilateral agreements between a government agency (or nonprofit organization) and landowners (either individually or as groups or communities). Conservation easements and nonmandatory, fee-simple land purchases are examples of this second type. The third type, unilateral action by one or more landowners with or without government involvement, includes collective/community action and individual efforts at environmental stewardship.

Tax incentives, financial subsidies, or cost-sharing schemes reduce private costs of restoration and increase the likelihood that the benefits to the landowner will exceed the

private costs. By using positive incentives, the government attempts to guarantee the landowner a net benefit level from participating that is at least as high as without participating. This "carrot" approach has been used extensively throughout the U.S. and Europe to induce farmers and other landowners to adopt conservation practices, retire environmentally fragile lands, and restore degraded ecosystems (Cooper and Keim 1996; Cabbage et al. 1993; Granskog et al. 2002).

In the U.S., a number of federal income tax incentives and cost-share programs have been directed at improving forest management and encouraging reforestation and ecosystem restoration. These include the Forestry Incentive Program (FIP), Conservation Reserve Program (CRP), Wetlands Reserve Program (WRP), Stewardship Incentives Program (SIP), Environmental Quality Incentives Program (EQIP), and the Wildlife Habitat Incentives Program (Cabbage et al. 1993; Granskog et al. 2002). An equally wide variety of similar programs exist under the European Union and its member states (Cabbage et al. 1993; Hodge and McNally 2000; Ledoux et al. 2000; McCarthy et al. 2003; OECD 1999; Terstad 1999; Weiss 2000; Whitby and Saunders 1996).

As always, a central problem for policymakers is determining the level of subsidy or incentive that encourages the optimal level of participation. Another critical problem with these programs concerns the ability to target the sites and elicit participation from landowners that are most crucial for maximizing the benefits associated with ecosystem restoration. Obviously, landowners vary significantly in terms of their likelihood of participation and the benefits available from restoring ecosystems on their individual sites. A large literature has developed that analyzes the characteristics of landowners that affect their likelihood of participation. The most important factors include income, personal values, tract size, residence, long-term plans, knowledge of management options and benefits, tax policies, available capital, and resource commodity values (Birch 1996; Bliss et al. 1997; de Steiguer 1984; English et al. 1997; Nagubadi et al. 1996; Wicker 2002).

If targeting is difficult or impossible because government agents are unable to observe the specific characteristics of individual landowners, a second-best approach uses a policy "menu" that offers different financial incentives in return for providing different levels of restoration. This would essentially involve paying additional informational subsidies that exceed the minimum amount to induce participation if landowner characteristics were observable by the government agent (Alberini and Segerson 2002; Wu and Babcock 1999).

Conservation easements are the most common type of bilateral, voluntary mechanism for encouraging ecosystem restoration. Conservation easements are legally binding agreements between landowners and governments or nonprofit organizations that are effectively a form of shared ownership. The landowner relinquishes the right to certain landuses (or agrees to manage the land in certain ways) for a given period of time (often in perpetuity) in exchange for tax benefits or direct monetary compensation.

Conservation easements are usually less costly to acquire than fee-simple purchase of entire properties so that with a fixed amount of public or nonprofit funds, more land can typically be restored and preserved with conservation easements. Compared to tax incentives, subsidies, or tradable development rights, easements carry less administrative burden and cost and typically require fewer changes in property or environmental laws or regulations. Because they are voluntary, easements tend to be more palatable than direct land-use regulation to private landowners. However, the long-term monitoring and enforcement of easement contracts may entail large administrative costs; nevertheless, the use of conservation easements has grown rapidly. Because conservation easements often result in a fragmented pattern of restored ecosystems, they should be combined with other approaches that identify broader areas for restoration and protection (Granskog et al. 2002).

A growing group of scientists, land managers, and policymakers suggest that effective ecosystem restoration must be planned at a landscape or watershed level (Gottfried et al.

1996; Hodge and McNally 2000; Kline et al. 2000; NRC 2001). Successful restoration efforts will depend not only on how landowners individually react to government incentives but also whether neighboring landowners and communities participate and can work together to maximize the benefits from restoration. For example, Hodge and McNally (2001) state that given the technical constraints and costs of water quality management, "wetland restoration will not be possible unless all producers cooperate (within a specific location)."

Watershed councils are used widely throughout the Midwestern U.S. to develop and implement watershed restoration plans and to coordinate management of riparian areas (Gottfried et al. 1996). In Ecuador, fishers, shrimp aquaculturists, and a variety of landowner and conservation groups cooperate to develop agreements to improve riparian areas and water quality, such as restricting harvesting of mangroves. In Costa Rica, landowners cooperate to develop ecotourism reserves in coastal mountain regions (Gottfried et al. 1996). In the Philippines, socioeconomic factors such as a knowledge of trees and tree planting, land-use patterns and ownership, and social and community organizations were more important than ecological factors in determining the relative success of forest restoration efforts (Walters 1997).

Individuals or communities may be motivated to unilaterally initiate restoration activities by environmental stewardship, personal satisfaction, or utility gained from improving the environment. Recent research in experimental economics suggests that effective communication and high marginal payoffs are essential to achieving voluntary provision of public goods and cooperative actions within a landscape (Ledyard 1995). Therefore, successful cooperative restoration efforts will most likely require some external agent (government or nongovernmental) that facilitates communication between landowners, provides information, incurs transactions costs, and provides resources to lower the costs to individual landowners for participating (Hodge and McNally 2000).

One would expect that if incentives such as tax reductions and subsidies are high enough, most landowners in a watershed or landscape would participate in coordinated efforts at ecosystem restoration. Nevertheless, collective provision of ecological services requires significant and costly information on technical requirements for ecosystem restoration. Indeed, the transaction costs involved in acquiring information and negotiating agreements between landowners may be so high that collective restoration efforts will not succeed, even with government incentives (Hodge and McNally 2000).

An example of such an approach in the U.S. is the Oregon Coastal Salmon Restoration Initiative (OCSRI). A coalition of state agencies and private interest groups developed OCSRI in order to avoid the consequences of the Pacific Northwest Coho salmon being listed as threatened under the Endangered Species Act (Kline et al. 2000). By appealing broadly to Oregonians' collective responsibility to protect salmon habitat, the initiative relied on community-based, voluntary efforts by private landowners, local interest groups such as watershed councils, and soil and water conservation districts. The actions by forest landowners to reduce timber harvests in critical riparian buffer areas were crucial for successful restoration. Apparently, a high proportion of forest landowners in Oregon are motivated to own land for protecting and enhancing habitat for threatened or endangered species, as well as timber production (Kline et al. 2000), suggesting a high potential for success. However, the owners with the largest tracts of forestland and the largest percentage of forestland were motivated primarily by timber production. Therefore, a combined policy that provides economic incentives such as tax relief or cost sharing may be required to induce cooperation of a sufficient number of landowners to achieve success (Kline et al. 2000).

Internal Drainage Boards (IDB) in the U.K. evolved from informal associations (clubs) of local landowners organized to improve land drainage into formal institutions with statutory responsibilities (established in 1976). They now act like a local government organization representing the competing interests of local residents. Although the IDBs are

well positioned to promote watershed-based collective action for restoring wetland ecosystems, several reforms are needed (Hodge and McNally 2000). The current system of using standard contracts to landowners severely limits flexibility and reduces the capacity to develop new cooperative actions by landowners. Even with institutional reform of the IDBs, collective action will be limited unless financial incentives are sufficiently high to encourage participation of adjacent farmers. Hodge and McNally (2000) recommended allowing groups of landholders to compete for restoration grants and contracts in order to promote larger cooperative restoration projects.

### *6.3 Measuring success*

A variety of criteria have been used or suggested for evaluating the success of government policies and programs for forest restoration (Cubbage et al. 1993). These include physical measures (e.g., number of ha restored), measures of ecological function (habitat, water quality, biodiversity), and socioeconomic measures (efficiency, cost effectiveness, and equity). All of these measures involve different ways of determining and comparing benefits and costs (ecological, economic, and social) of program and policy alternatives. Although measuring the direct costs of restoration programs is relatively straightforward (Boyd and Simpson 1999), measuring the benefits is more difficult and is the issue addressed here.

In addition to producing and supporting species and biological and ecological functions, forest ecosystems produce and support a wide variety of socially valuable services such as water quality, erosion and flood control, wildlife habitat, recreation, biodiversity, and aesthetics. When ecosystems are degraded and subsequently restored, the social value of the services produced also changes. Ecosystem services are the beneficial outcomes of ecosystem functions. For example, reducing runoff, erosion, and flood peaks are biophysical functions of a watershed restoration project. Examples of the services they provide include reducing damage to agriculture, water quality, buildings, and roads. Even though a restored ecosystem may have high rates of ecosystem function, it may not necessarily produce ecosystem services with high social values (Boyd and Wainger 2002b). Therefore, evaluating the effectiveness of ecosystem restoration requires more than just good ecological analyses. Evaluating the success of alternative policies and programs for restoring ecosystems requires comparing the social value of the change in ecosystem services to the costs of implementation, management, monitoring, and enforcement.

The problem of how to estimate benefits can be divided into supply and demand components (Johnston et al. 2002). The supply component consists of estimating how policy and program alternatives affect the quantities of ecosystem services and the demand side establishes how people and society value the various services. Most current governmental programs rely on purely biophysical descriptions to quantify the success of supplying ecological functions from restoration projects. Further, these estimates are often very simple, such as area treated. The demand or value side is rarely assessed (Polasky 2002). In the absence of adequate assessments of the value of the services produced by restored ecosystems, regulators and program administrators tend to use differences in the program costs as the primary criteria for success. However, if restoration efforts are evaluated solely on area restored (or treated) and cost effectiveness, most restoration will occur on the least expensive, least ecologically valuable lands. The result would likely be a migration of restored habitats and ecosystems to remote, cost-effective sites, an outcome that is unlikely to be ecologically or economically optimal (Boyd and Wainger 2002b).

Current regulatory land-use programs in the U.S. under analyze the social value of both the lost and restored ecosystems under compensatory mitigation programs (Boyd and Wainger 2002a, 2002b). Usually, mitigation decisions are based solely on requiring

restoration of an "acre for an acre" of a biophysically similar site when the landowner seeks to develop or otherwise negatively impact an ecosystem (usually wetlands). Unfortunately, this simplistic approach fails to consider how society values the destroyed and restored ecosystems. Indicators of social value include the location in the greater landscape, availability and importance of local substitutes and complements to the site, and future risks to the restored site's ability to continue to provide services.

Surveys and econometric analyses are used to develop evidence and estimates of the demand for and resulting social value of different ecosystem services (Costanza et al. 1998; Holmes et al. 2004; Kenyon and Nevin 2001; Kline et al. 2000; Loomis et al. 2000; Lupi et al. 2002; Schaberg et al. 1999). Unfortunately, these analyses tend to be difficult, costly, and incomplete; rarely do they value the full range of ecosystem services at a site. In response, researchers are beginning to develop alternative approaches by combining economic and ecologic indicators and attributes that identify potential differences in social benefits generated by restored ecosystems on a landscape level (Boyd and Wainger 2002a, 2002b; Johnston et al. 2002). These efforts require credible models of the production of ecosystem services that link models from the natural sciences (which predict how changes in ecosystems impact the services produced by those ecosystems) with economic models (which estimate the value of those services and predict landowner participation). Since the value of services from any particular site depends on both the conditions at the site as well as its location on the landscape, these models will need to operate at landscape level (Polasky 2002).

## 6.4 Conclusions

Different approaches taken by governments to encourage restoration will differ in their implementation costs (transactions costs), in who ultimately bears the cost of restoration, and in the ability to select and maintain appropriate habitats and ecosystems. Policymakers have the difficult task of making trade-offs between how much to spend, who pays, and how to ensure successful restoration. For example, fee-simple purchase by the government of lands for restoration provides the lowest risk of failure and the best targeting of ecosystem restoration. However, the costs are very high and political perils may be even higher if the government begins a process of requiring large numbers of private landowners to sell their lands to the government. In contrast, a broad-based program of tradable development rights (e.g., compensatory mitigation) may result in many restoration commitments on paper, but significant ongoing problems with compliance. Voluntary approaches tend to be more politically palatable and usually less expensive, but often result in a fragmented landscape of restored and unrestored sites with less than optimal production of ecological services.

All this suggests that there is no restoration policy panacea. All restoration policies and programs involve trade-offs. A combined approach of carrots (voluntary approaches) and sticks (mandatory or regulatory approaches) will usually be required to achieve maximum production of ecological services at the lowest cost. The exact combination of efforts needed to optimize restoration efforts will depend on local political, economic, social, and ecological conditions and require significant and difficult economic and ecologic modeling and analysis.

Perhaps the greatest unresolved policy issue, no matter which policy or combination of policies is used, concerns targeting and choosing the properties for restoration that will produce the greatest ecological benefits at the lowest societal cost. The use of a spatial, landscape, or watershed basis for planning and implementing restoration policies and programs is very complicated and difficult due to the interdependence of habitats and properties across a landscape, scientific debate over restoration priorities, and the inherent difficulties in predicting human development patterns across a landscape (Boyd and Simpson 1999).

The easiest approach to the problem of ensuring sufficient restoration across a landscape would be through public lands. However, public lands are generally too small to provide the necessary economies of configuration (Gottfried et al. 1996). Thus, creative policy and program approaches are still needed and usually require a combination of targeted voluntary and mandatory policy instruments combined with community-based, cooperative approaches between government agencies, nonprofit organizations, and landowner groups and organizations to effectively encourage the majority of landowners in a landscape or watershed to participate. Obviously, this type of approach is complicated, costly, and challenging; it requires considerable understanding of the social dynamics that promote or obstruct alternative institutional solutions to ecosystem restoration at a landscape level. Considerable social, economic, and institutional research and political support will be necessary to turn this dream into a reality.

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