Ecological Stewardship

A Common Reference for Ecosystem Management

Volume III

- **Public Expectations, Values, and Law**
- **Social and Cultural Dimensions**
- **Economic Dimensions**
- **Information and Data Management**

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A practical reference for scientists and resource managers

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Ecological and Resource Economics as Ecosystem Management Tools

Stephen Farber and Dennis Bradley

Key questions addressed in this chapter

- *A review of traditional economic analysis of inefficient use of ecosystems; the basis for why we need ecosystem management*
- *A review of traditional economic methods for valuation of ecosystem services; the basis for establishing and evaluating trade-offs, or costs and benefits, in ecosystem management*
- *A review of traditional economic instruments for correcting inefficiencies*
- *An analysis of shortcomings of traditional ecosystem management models*
- *The contributions of Ecological Economics to ecosystem management issues*

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1 INTRODUCTION AND SUMMARY OF KEY ISSUES

Economic pressures on ecosystems will only intensify in the future. Increased population levels, settlement patterns, and increased incomes will raise the demands for ecosystem resources and their services. The pressure to transform ecosystem natural assets into marketable commodities, whether by harvesting and mining resources or altering landscapes through development, is likely to be enormous. Ecosystem management must establish means of assuring that these natural assets are used in a manner that provides high returns to human welfare, and of sustaining their abilities to continue generating valuable product and service flows. Effective management requires understanding of the ecological processes underlying natural asset structures and processes, as well as the economic factors lying behind the values of these ecosystems under various use scenarios.

The major dilemmas in managing ecosystems, beyond understanding how these ecosystems work, will include understanding of the economic dependence on ecosystems of various types and qualities, recognition and evaluation of competing and complementary multiple uses of ecosystems, and development of appropriate decision processes and management instruments. These dilemmas will be confounded by the fact that many values of ecosystems will not be narrowly economic in the sense of merely providing commodities for economic use. Active and passive uses of ecosystems *in situ* will require values of preservation of natural systems at some minimum requisite levels of health and integrity. Maintaining ecosystem resilience as insurance against dramatic, irreversible changes induced by economic activities will require valuing ecosystem conditions as option values for future uses. Ecosystem complexity and complex connections to economic systems will require full accounting for the values of ecosystem conditions possibly far removed from local circumstances.

While amicable resolution may be the most desirable solution to ecosystem use conflicts, it may not be possible. When resolution is not possible, rational management of ecosystems requires some notion of accounting for the plusses and minuses of various options, analysis of the distribution of effects of these options, mechanisms to identify and arrive at the most desirable solutions, and a means of enforcing those solutions. These are daunting tasks. Ecological and resource economics have some skills and insights to offer to ecosystem management.

The purpose of this chapter is to present the potential contributions that conventional resource and environmental economics can make to the task of managing ecosystems and human economies. Environmental management is both a scientific and economic problem. Conventional economics has been useful in establishing valuation methods for ecosystem management and pinpointing various sources of inefficient use of natural systems. However, ecological economics is concerned that it has not adequately incorporated biophysical realities and complexities into understanding ecosystem values and processes, and for relying too extensively on individualistic valuations for what are largely social conflict problems. The assumption of ecological economics is that ecosystems are too complex and our knowledge too limited to permit substantial interventions in these systems without doing substantial harm to the structures and process of those systems. This perspective places a greater management emphasis on preservation of ecosystem health and integrity and focuses policy on human adaptations to ecosystem constraint. While there is nothing inherently inconsistent between conventional and ecological economics, the emphasis on issues and the policy suggestions differ somewhat.

The purpose of Section 2 of this chapter is to provide the ecosystem manager with a framework for considering why ecosystem management is necessary. It outlines the sources of failures of economic systems to achieve highest valued uses of natural assets and their associated ecosystems. It is these failures which motivate the management of ecosystem assets. Conventional economics has developed an extensive tool kit for valuing various user and non-user services and products from ecosystems. Section 3 introduces these economic valuation methods. Section 4 presents a brief explanation of how to use estimated values and impacts in making management decisions. Correcting for failures in economic systems requires the use of behavioral control instruments to achieve ecosystem management goals. Section 5 provides a case study of valuation issues applied to valuing forested ecosystem services. Section 6 addresses the range of instruments for correcting failures in achieving highest valued uses of natural assets. Contributions from the field of ecological economics are presented in Section 7, and are contrasted to traditional economics. Section 8 is a summary.

2 SOCIALLY VALUED HIGHEST USES, EFFICIENCY AND INSTITUTIONAL ARRANGEMENTS

The broad perspective that conventional economics offers to ecosystem management is the general notion of "efficiency," meaning the attainment of some goal with the least "costs," or the attainment of the highest goal for some acceptable level of "costs." However, as in all things, "the devil is in the details." For example, the manner in which traditional economics has made the general notion of cost–benefit analysis concrete enough for policy purposes has sparked meaningful debate (Kellman, 1981, Sagoff 1993, Kopp 1993).

Social systems are inefficient when they do not achieve the socially valued highest uses of man-made and natural assets. Economic value includes direct use values, indirect use values, options values, and existence and bequest values. Table 1 illustrates these four categories of economic value for forests. Direct use values include the narrowly extractive values of a forest plus direct, *in situ*, uses such as recreation or education. Indirect uses stem from, for example, reliance on forests for erosion control and hydrologic functions, in turn protecting downstream water supplies and aquatic habitats. Option values refer to potential direct and indirect uses that individuals may consider worth preserving. Existence values, which refer to simply knowing the forest exists, would include the cultural values of ecosystems. Bequest values refer to values of leaving a legacy of ecosystem capacities; i.e., a value of stewardship. Table 1 illustrates that economic values can be highly private, in the case of timber production, and highly public, in the case of flood or cultural value protection. They may also be highly localized or highly dispersed spatially and temporally. The ecosystem management dilemma is to recognize that all these values exist simultaneously, and may be held by local, regional, national, and international stakeholders in current and future generations.

Certain institutional arrangements between and among humans and their assets may tend toward the attainment of the socially highest valued uses of assets. The "privatization" hypothesis, a dominant version of this proposition, asserts that assets will gravitate toward their highest valued uses if the following conditions are satisfied:

- Asset ownership conditions are clear.
- There is unfettered freedom to voluntary exchange of assets.
- Contracts defining ownership rights and transfer conditions are inexpensive.
- Agreed upon terms of contracts are enforceable.
- Potential contracting parties have full and accurate information.

This view suggests that privatization of assets will lead to their highest valued uses, and markets are the institutions that facilitate that process. An important implication of the proposition is that natural asset management by anyone other than the immediate owner of the asset is both unnecessary and, worse, is likely to lead to inefficient use of resources.

Welfare economists have paid particular attention to the privatization argument and focused on the analysis of the social efficiency effects of various privatization and market arrangements. Considerable effort has been directed toward the classification of failures of private ownership and market arrangements to achieve highest valued asset uses. Section 2.1 below outlines these failures. These failures have been deemed so considerable in some circumstances to merit public ownership or management of natural assets.

2.1 Sources of Economic Inefficiencies in Natural Asset Use

The context of economic efficiency is social welfare. Efficiency is interpreted with respect to whether there are opportunities to change resource allocations or economic activities so that welfare gains exceed welfare losses.

2.1.1 What are Static and Dynamic Efficiencies?

Static efficiency refers to a point in time. For example, a static efficiency issue is whether to continue allowing

Table 1. Economic Values of a Forest.

Direct Use Value	Indirect Use Value	Option Value	Existence and Bequest Values
Timber products	nutrient cycling	future uses	biodiversity
Non-timber products	watershed protection		cultural
Recreation	air pollution control		
Medicines	microclimate functions		
Genetic	material	carbon storage	
Education	groundwater	recharge	
Human habitat	flood moderation		

grazing in forests. The static efficiency test is whether social welfare is higher or lower by allowing the grazing. A practical test would be whether ranchers' aggregate willingness to pay to continue grazing cattle is more or less than recreationists' willingness to pay to keep cattle out of the forests. In general, all stakeholders' monetized gains and losses should be considered.

The status quo property rights with respect to a resource or activity at the time a proposed change in use is considered will be important in determining whether willingness to pay or willingness to accept compensation are the appropriate measures of welfare changes. For example, if there is a presumed right to graze, a change from the status quo in denying that activity would require measuring the harm done to ranchers, measured as willingness to accept compensation for losses, and measuring gains to recreationists, measured by their willingness to pay to terminate grazing. Conversely, if there is no presumed right to graze, a change in status quo in allowing grazing would require measuring recreationists' willingness to accept compensation for losses, and measuring ranchers' willingness to pay to graze.

Broad social welfare considerations may weight the gains and losses of the winning and losing parties quite differently. An example is a case of environmental groups seeking to purchase grazing rights in New Mexico. The administrator of the grazing commission ruled that these groups could not purchase these rights, effectively weighting their welfare as zero in the implied social welfare function (New Mexican, November 23, 1995).

Dynamic efficiency addresses intertemporal issues. For example, allowing timber to grow may increase its welfare value. However, harvesting now and converting the net incomes into investments could possibly result in greater welfare gains in the future than would continued growth of the timber stock. Continued growth is dynamically inefficient in this case. Full consideration of the comparative welfare gains would have to account for all welfare impacts, including the welfare benefits of the forest prior to cutting, such as recreational and habitat values. Growth in recreational values over time for an intact forest would have to be considered.

A critical issue in determining dynamic efficiency is the rate at which future welfare changes are discounted. Hence, discounting becomes a key issue in testing for dynamic efficiency. The timber example would suggest discounting the welfare value of continued growth with a discount rate representing alternative returns on investments that could be made with the net incomes from the harvested timber. Although some statutes mandate discount rates and procedures, these rates and the procedures themselves may not always be appropriate for a particular efficiency analysis problem.

2.1.2 What are the Sources of Economic Inefficiency?

The "privatization" model of efficient asset use, outlined above, argues that when ownership of assets is clearly defined, exchange is voluntary, contracts are well-defined, enforceable and relatively costless, exchange is not costly, and all parties have reasonably sufficient information, then assets will tend to attain their highest individually valued uses. By implication, if individual want satisfaction is the only social value entering into social welfare, the socially highest valued uses will also be attained. In other words, private ownership and free exchange assure that opportunities will be exploited for changing activities or uses of resources in such a way that welfare gains exceed welfare losses.

The above conditions on ownership and exchange are likely to be valid for a wide range of assets, and the presumption that social welfare is commensurate with private, individual value is also likely to be valid for a wide range of circumstances. However, there are major exceptions to these conditions in some very important cases. These are generally called market or property rights failures, whereby private ownership and exchange fail to assure the socially highest valued use of assets. Failures that arise in the use of resources and the environment stem primarily from the following six sources:

1. Property rights failures
2. Spillovers or externalities
3. Public goods
4. Transactions costs
5. Immobilities and adaptability
6. Information failures and uncertainty
7. Government intervention

Each of these sources of failures is discussed below in the context of ecosystem management.

Property rights failures

Property rights require the identification of property, establishment of rules for its use and transfer, and enforcement of those rules. Environmental and resource assets are notable for difficulties in establishing well-defined and enforceable property rights, as well as for violating the condition that private values fully reflect social values. Clean air is a valued asset, but if there is no property right to clean air, users can freely expropriate it to their own use; the result is pollution. Property rights are also not fully definable if a property generates some services for which the owner cannot reap any rewards. The property generating the services may be well defined and transferable, but the services themselves may not be identifiable, transferable, or enforceable. An example is a forest or wetlands, both physically well-defined, but which produce services, such as habitat or air and water purification, that cannot be traded along with the property. There will be well-defined markets for the physical properties, but the prices will reflect only the returns from those properties captured by the owner. Market prices for these properties will underrepresent their general social value. An example is Louisiana coastal wetlands, which have market prices under $500 per acre (primarily for underlying mineral rights) but social values exceeding $10,000 per acre (Farber 1995). We cannot be assured that markets will lead to the forest or wetland attaining their highest use values. Furthermore, social welfare may include non-use values that are of no importance to potential private owners of the forest, such as spotted owl habitat, suggesting that private values would not fully reflect social values in this case.

Spillovers or externalities

Spillovers arise as the unintended consequences of economic activities of consumption or production. They are also referred to as "externalities." Spillovers can be negative or positive, and can be associated with either the production or consumption of economic goods or services. For example, the unintended siltation of a stream would be a negative production spillover associated with timber production that increased soil erosion. The unintended habitat edge enhancement from a selected forest cut would be a positive production spillover. The soil compaction of heavy recreational use would be a negative consumption spillover. The unintended vegetation enhancement of increased deer kill by hunters would be a positive consumption spillover. Given the complexity of ecosystems, there are likely to be many positive and negative spillovers associated with economic activities. Spillovers create inefficiencies in asset use because there are unintended benefits or costs associated with the consumption or production activities that are not considered by the parties making the use decisions.

Public goods

These are goods, services or actions that if made available to one person can be fully used by others without diminishing their usefulness to anyone. Furthermore, it is difficult to exclude persons from their use. These conditions are referred to as "rivalry" and "excludability" conditions for public goods. They are likely to be present in many natural resource and ecosystem management circumstances. An example would be a wildlife habitat that enhanced wildlife viability, or a reforestation that would provide uncongested recreational opportunities. Conversely, public "bads" are those goods, services, or actions that have negative consequences to recipients and, if received by one person, would have undiminished consequences for others also. Air pollution and habitat destruction are examples of public bads. These are public types of spillovers that when imposed on one party become equally imposed on others.

Public goods and services would be underprotected with the privatization model. For example, private forest owners are unlikely to consider habitat protection or aquatic health as major factors in making timber management decisions. Although the wildlife that would result may have a value to the public, the private landowner will find it difficult to recoup that value. Individual members of the public are unlikely to be willing to pay for this habitat protection, hoping to free-ride on purchases by others. Interesting means of dealing with this problem are to proscribe private activities, convert private to public ownership, or establish means of collecting revenues from the public at large for compensating private landowners for providing these values. This is in contrast to private "bads," such as negative spillovers, where the harmed parties are discrete in number and free-riding on others purchases is not possible. In this case, private transactions, as suggested by Coase (1960) would provide adequate remedy and eliminate inefficiencies under certain conditions.

A good or service may jointly have both private and public goods characteristics. Forests yield goods that are clearly private, such as timber, and services that are clearly public, such as air quality, habitat, and natural heritage. Allowing the private use to dominate forest management may result in inefficiencies for the provision of the public good components of the forest. This does not always have to be the case, as some arguments for sustainable forest management suggest. An efficient management scheme would jointly consider the benefits and adverse impacts of all uses. Privatization of the forests and rangelands would exclude public goods characteristics from consideration.

Transaction costs

Transaction costs refer to the costs of making contracts or exchanges. They make trading of property rights more costly. Costs can be so high as to prevent some trades that would enhance welfare efficiency. The result is that privatization and markets would not result in the attainment of highest use value of assets. Property rights may be well-defined and enforceable, but trading those rights may be costly. In the rancher-farmer case where cattle trample farm crops, the value of the crops damaged may exceed the value of the cattle feeding on the crops, so an economically efficient allocation of resources would be to raise fewer cattle and more corn on the land. However, the legal costs of attaining any reallocation of rights to that effect may be so high that the farmer cannot procure the necessary rights. In this case, who has the initial rights is critical in establishing the socially efficient use of the land.

Transaction costs are important in explaining why privately negotiated remedies for private or public spillovers are not forthcoming many large cases. Private arrangements become costly as the numbers of persons increase, and the tendency to "fee ride" on agreements increases.

Immobilities and adaptability

Economic inefficiencies arise when resources are not easily transferable between uses, so fail to attain their highest valued uses. For example, while there may be a shortage of logging personnel in Southeast forests, there may be a surplus in Northwest forests. The reluctance of families to move between markets reduces efficient use of labor and natural resources. Contractual rigidities of property rights may impede mobility of resources. This may be particularly relevant when new information is obtained about the value of a resource. For example, logging contracts may be set prior to the discovery of an endangered species.

Immobilities and inadaptabilities can increase the costs associated with negative spillovers when the recipients or generators of the spillovers will not change their behaviors or locations. For example, a negative spillover may be most cheaply remedied by the movement of the recipient away from the source. Forest recreationists may disturb the grazing of cattle, and it may be cheaper to ask recreationists not to use the resource than to interfere with grazing. Salmon fishermen dependent on low-yielding, polluted streams may continue to live near and fish those streams. If their immobility is explained by the fact that they value their current location relative to other feasible locations more than the spillover costs of low harvest that they bear, their current location is economically efficient; i.e., the cost of relocation exceeds the value of the spillover damages. However, if their immobility is explained by lack of information or poor access to capital markets to finance a relocation, their immobility creates economic inefficiencies through a failure of markets to function properly in allocating persons across locations with different levels of spillovers.

Information failures and uncertainty

Markets may fail to assure that resources attain their highest valued uses if information about those uses is limited. Lack of information and misinformation are both sources of economic inefficiency. Information failures can magnify problems already arising from other inefficiencies. For example, spillover costs to ranchers from recreational use of forests could be moderated if ranchers or recreationists were aware of alternative options. There are recognized remedies to information failures, such as advertising. However, there may be credibility problems associated with some information sources, particularly when sources are identified as having a large stake in a particular outcome, as in the case of used car salesmen.

Uncertainty is a type of information failure stemming from the inability to predict exactly the values of resources or activities. Uncertainty may impede the development and functioning of a market, creating inefficiencies by restricting exchanges. Informational assymetries can arise when buyers have more information than sellers, or vice versa. This situation can arise in timber sales on public lands, when private buyers have more information about timber conditions than the public agencies selling the timber. In this case, adverse selection would result in "skimming the cream."

Another perverse result of uncertainty is associated with the inability of sellers to completely determine the terms of use. This is referred to as "moral hazard," whereby the availability of the good perversely increases the likelihood of its use. An example would be publicly subsidized livestock food supplement programs under severe weather conditions, a form of insurance. The perverse incentive reduces the rancher's incentive to assure adequate winter forage. Or publicly subsidized coastal storm insurance induces overbuilding of coastal areas.

Government intervention

Direct or indirect government interventions in markets can induce their own failures and economic inefficiencies. Of course, some interventions are for the purpose of correcting other failures. For example, regulating pollution corrects for the inadequate allowance for spillovers in private decisions. Direct interventions would include restrictions on prices or quantities traded on markets or mandating behaviors. Indirect interventions arise when the government subsidizes, or taxes at levels below full public costs, particular resource uses or activities. For example, the public construction of logging roads in public forests is a subsidy to the timber industry; or the permitting of grazing on public lands at prices below the costs of providing the service is a subsidy to the cattle industry. More indirectly, cattle and grain price supports indirectly complement one another in inducing more pressure for cattle grazing on both private and public lands. Demands for ski areas in national forests are increased by various public policies that keep energy and transportation costs low. Macroeconomic policies that perpetuate high unemployment reduces the mobility of labor from low valued to high valued uses, possibly contributing to excessive logging pressures. Interest rate deductions for home mortgages lower the cost of home ownership, inducing demand for larger homes, resulting in higher timber demands. Monetary policies affect interest rates, which impact economic growth and the demands for natural resource assets.

3 MEASURING WELFARE IMPROVEMENTS

Valuation is a critical task of ecosystem management when private decision-making fails to achieve highest valued use of the ecosystem. Social welfare norms are the basis for valuations. These norms are sometimes explicit and carefully specified in institutions such as statutes and their resulting regulations. For example, banning PCB's or ozone-depleting chemicals is a very explicit statement of social norms. Norms become explicit through judicial decisions, whether they are interpreting intent or administering common law. However, it is more often the case that social norms and values are poorly specified, or only specified on a case-by-case basis when circumstances arise where conflicting values are at stake. For example, the Endangered Species Act appeared to establish well-defined values for every species; the value of each species is potentially infinite when it is near extinction. The implications of this valuation have set in motion many attempts to change the values implied for species. Another example is wetlands.

When norms are poorly specified, management is difficult since no well-defined metric can be used to measure efficiency or inefficiency in management. The result is often a groping toward understanding of implied values through marginal decisions, awaiting political repercussions, and reformulating the implied value system after society realizes the implications of prior decisions.

Cost–benefit analysis is frequently used as a social norm for management decisions. The practical implementation of cost–benefit analysis as a management method raises justifiable concern. A practical version of cost–benefit analysis uses money as the metric for measuring costs and benefits of changes in ecosystem resource use. A variety of techniques have been developed to establish this monetary metric.

The purpose of this section of the chapter is to outline economic methods for valuing ecosystem changes using monetary metrics. The underlying economic principle is that some changes in individual welfare can be monetized, and that this monetization reflects what a person is *willing to pay or accept* for a welfare change. Monetization of values associated with marketed products and services of ecosystems appears straightforward. However, as noted above, many ecosystem values cannot be ascertained directly by observing market behaviors, so economists have attempted to take monetary valuation into non-traditional areas.

3.1. Sources Of Value From Ecosystems

From an economic perspective, ecosystems are natural assets, providing flows of materials and services valued in the human economy. Some of these flows are used directly in production or consumption activities. These would include minerals and timber, waste disposal, the fertility and structure of soils, etc. These direct use values could be further subdivided into active uses, such as fishing, and passive uses, such as birdwatching. They create direct benefits to users by enhancing the productivity of economic activities, enhancing the quality of life, or by allowing for reduced use of more costly alternatives, such as the use of man-made fertilizers, greater fishing effort or increased recreational transport costs. They may have preservation values insofar as individuals wish to maintain the options for future personal use. These values may also be considered "Instrumental" values, as the natural assets are being used as instruments for economic or life support purposes. In contrast to these Instrumental values, ecosystems have Non-Instrumental values, related to aesthetic, moral and spiritual, and cultural purposes. For example, moral values may relate to some perceived obligation to steward a resource for future generations, or to avoid destruction of species, an act which could be considered immoral. Cultural values relate to the importance of ecosystems in broad social and community contexts. For example, the value of acequias in northern New Mexico exceed the value of the water itself (an Instrumental Value) and include the importance of the process of preserving and tending these canals for maintaining communal relations in Hispanic communities (Tarlock 1992). The

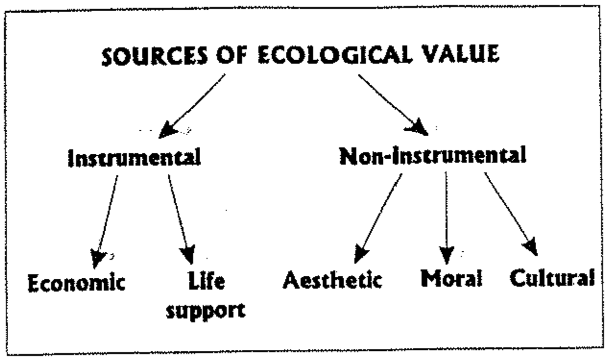

Fig. 1. Sources of ecological value.

"Value Tree" (Fig. 1) reflects these values of ecosystems (Bengston and Xu 1995):

Each of these values is identifiable, albeit perhaps unquantifiable. They are certainly not equally monetizable. Instrumental economic values are the most amenable to monetization. Life support values are infinite, by definition. Aesthetic values may be monetizable, while moral and cultural values may transcend mundane economic valuation.

3.2 General Concepts of Monetary Valuation of Welfare Changes

To begin the valuation discussion, it should be obvious that a person is willing to pay at least $P for a good they are observed purchasing at a price of $P. Using $P as a measure of the monetary equivalent of the welfare gain obtained by having the good would be a minimum measure of this gain to the person. It would also be a minimum measure of the loss to the person if they are denied the good. If there is a market for the good, and that market is reasonably competitive, the observed price of a good would roughly reflect the value of that good to someone in society. The point is that when there are well-functioning markets for goods, the market prices of those goods reflect the value of marginal units of the good to the marginal buyers in the market. In other words, price is a legitimate basis for valuation of individual welfare.

The use of these market values to reflect social values may be inappropriate in several important cases. First, if there are spillover effects of the production and use of a good, the social value of a good may be more or less than the observed market price. The price of beef may be $P per pound, but if each pound does $Y damage to water and range resources, the social value is only $P – $Y per pound. Second, markets may not function well, or are non-existent for some types of uses. This is the case for public goods, as discussed above. For example, there will not be well-functioning markets for aesthetic or spiritual properties of forests or for endangered species.

The failure of markets to adequately reflect the values of ecosystem uses prompts the necessity for non-market, or pseudo-market techniques for valuing those uses when monetizing welfare changes. These techniques can generally be divided into direct and indirect techniques. They can be further divided according to whether behavior is observed or hypothetical (Freeman 1994). Direct methods involve obtaining information directly about the use itself. If a well-defined market exists, or can be simulated effectively, observing prices that people pay for a good is a direct, observed method. For example, the price of milk almost fully reflects its value (except for adverse spillover effects from cattle management). If people are directly asked how much they would value a good, it would be a direct, hypothetical method. For example, one could ask a person, "What would you be willing to pay for a clear view of the Grand Canyon?"

On the other hand, one can obtain some value-relevant information indirectly. If we observe people paying for some goods that are intimately related to the uses we wish to value, that would be an indirect, observed method. For example, if property prices are higher near cleaner streams, these prices reflect something about the value placed on clean streams, although the stream does not have a price of its own. Or if we observe people incurring costs to avoid an undesirable circumstance, such as boiling contaminated water, these costs would indirectly reflect at least one of the benefits of clean water. These direct and indirect techniques are discussed below. There is a considerable literature on these methods (Freeman 1994, Kopp and Smith 1993).

3.2.1 Observed-direct Methods for Valuing Ecosystem Uses

This is an appropriate technique when there are actual prices for the use of an ecosystem or its elemental materials, services, and processes. If we can establish the price of use, that price would reflect use value at the margin of use. For example, a grazing fee of $X per animal, with *no restrictions* on use, would result in a rancher grazing cattle to the point at which the value of grazing an additional animal, in terms of net revenue (revenues minus costs of production, exclusive of the fee), just equals $X. This would measure the welfare increase to ranchers for one more animal, or welfare decrease for one less animal. This means the $X underestimates the average profit on all the animals the rancher grazes since the last animal grazed will be the least profitable.

When there is an opportunity to vary usage prices, the valuation problems cited above can be mitigated. For example, if grazing fees or duck stamp prices are varied (Bishop and Heberlein 1979), we can observe the changes in usage. This allows the valuation of various usage levels and resulting valuation of large changes in usage. Any spillover damages to the ecosystem that are not accounted for in rancher net revenues must be subtracted to obtain the net social benefits of use. Even this simple valuation procedure has its complications. If use restrictions apply, such as limits on the number of cattle permitted, the established fee of $X would not reflect the value to the rancher of an additional animal. If the rancher would have grazed more than the limited number of animals, the value of grazing an additional animal exceeds $X, but we cannot determine by how much. However, it is most important to recognize that the value of grazing to the rancher is not the market price of the cattle, but the net profit from cattle sales.

It is important to warn against potential misuses of observed valuations. One of the most notable mistakes (intentional or inadvertent) is to equate the market value of a good produced using an ecosystem with the value of the ecosystem services to that good. For example, someone may argue that a steer grazing on public land has a market value of $1,000, therefore, the value of land for grazing is $1,000. This is completely erroneous. There are costs associated with raising the steer and taking it to market, exclusive of grazing. Supposing these costs are $600 per steer, the implied value of the land for grazing is only $400 per steer. It is this latter value that must be compared to the value of other uses, such as recreation or cultural values, to assure highest and best use of resources. Similarly, a $1,000 tree which costs $600 to grow and bring to market has only a net value of $400, which is the implied, or stumpage, value of the tree. Net profit, or producer surplus, is the correct measure of welfare loss and gain from natural resource use in economic production.

3.2.2 Hypothetical-direct Methods for Valuing Ecosystem Uses

This method obtains a measure of value directly from individuals. However, it involves creating a hypothetical situation in which the individual provides some information about value. The primary method is referred to as "contingent valuation," or CV, because it obtains a valuation contingent upon a hypothetical scenario. CV is potentially useful when we cannot observe actual behaviors that reveal valuations. This is the especially the case for non-use type values, where people value something even when they do not directly use it.

The CV method appears quite straightforward: simply ask people directly what something is worth to them. There are many potential bias pitfalls, where bias refers to the method revealing something that was not intended, or not revealing something that was intended. For example, a strategic bias can stem from respondents wanting their responses to impact policy, so may give higher or lower willingness to pay or willingness to accept values than they truly hold. Outlining the advantages and disadvantages of the CV method is beyond the scope of this chapter. The reader is referred to Mitchell and Carson (1989), Freeman (1993), Kopp and Smith (1993), and Cummings, et al. (1986). In spite of these pitfalls, it is a potentially useful method for valuation. It is allowable in courts for natural resource damage assessments.

CV methods remain about the only viable means of measuring non-use values. There is considerable controversy about their validity and reliability in valuation (Hausman 1993). Freeman (1993) provides a good analysis of applications and issues.

3.2.3 Observed-indirect Methods for Valuing Ecosystem Uses

When use of the good or service being valued is coupled with the use of other goods or services for which people directly pay, we can often impute the value of the good in question from the observed prices and quantities of the related purchased goods. For example, people have to travel to a park to use it, or have to buy or lease property at a lakeshore to live there. How much they spend to travel or how much more they spend to live next to the lakeshore can be used to indirectly reflect the value of the park or the lakeshore.

The travel cost (TC) method has been used extensively to value recreation sites or qualitative changes in those sites. The costs people incur to access a site can be used to establish demand functions for those sites. Variations on the TC have been used to assess changes in the qualities of sites. If there are differences in site qualities, demand would depend upon those qualities. Implied values of site quality can be inferred from the changes in demands due to quality changes. For example, Smith et al. (1983) estimate the implied value of water quality enhancements at U.S. Army Corps of Engineers lakes in this manner.

The random utility (RU) model of choice behavior has been developed as a useful alternative to the TC (Kopp and Smith 1993). The RU model formally derives a demand for site visits, which can be estimated with data. It is a useful method when individual behaviors are available, when there are substitute sites of varying quality, and when individuals visit some but not necessarily all sites.

The hedonic method is another type of valuation method that relies on the use of other goods or services in conjunction with the resource or ecosystem being valued. Studies in a variety of circumstances have shown that people are willing to pay more for properties where amenities are high, or will accept lower wages where job risks are lower (Freeman, 1993). These relations are the basis for measuring the value of location amenities or job risks. The procedure is to observe circumstances where amenities or risks vary, control for other price determining factors, and establish the empirical relation between amenities or risks and the prices of the associated goods (properties or jobs) while holding those other factors constant.

The averting expenditures method is another valuation technique that relies on information about the use of goods or services in conjunction with the resource or ecosystem being valued. This method presumes that people will incur expenses to avoid adverse effects stemming from the loss of valued resources or ecosystem services. The hedonic model is of this type when the concern is loss of amenities; for example, people will pay more to live further from a noxious or hazardous location. More generally, there are many types of averting expenditures: recreationists will travel further to a nice site than an unpleasant site; households will incur bottled water costs to replace degraded drinking sources.

The marginal productivity method is an indirect method involving determination of the indirect effects of resource or ecosystem change on the economy and people. A classic example is to establish the increase in fisheries harvests when the quantity or quality of wetlands increases. The procedure is to estimate a fish harvest production function, where inputs include fishing effort and wetlands conditions (Farber 1995). The marginal effect of a wetlands enhancement can then be estimated. The resulting increase in fish catch would have a value to society, and that value would reflect the implied value of the wetlands that yielded the increase.

3.2.4 Hypothetical-indirect Methods for Valuing Ecosystem Uses

The direct hypothetical methods outlined above explicitly asked a person how much they valued some ecosystem good, service, or condition. Valuation could be inferred indirectly from hypothetical questions posed to persons. Conjoint analysis (CA) is a variation on the CV method that may become increasingly useful in valuing complex ecosystems. The basic principle of CA is to present the respondent with a choice between two sets of "goods," where the goods have multiple attributes. When one of the attributes is a price of the entire good, appropriate experimental designs allow the determination of the marginal value of each of the attributes. For example, if we wish to value water quality, fishing abundance, and aesthetic value of a forest ecosystem, we could design an experiment in which these three dimensions are varied along with a price that respondents would have to pay for access. Respondents are asked to choose or rank among the choices presented. The marginal value of each of these three qualities could be established using statistical procedures. Examples of applications are cited in Freeman (1993).

3.3 Valuation of Welfare Changes Versus Impact Analysis

An important distinction must be made between valuation of the welfare changes resulting from ecosystem management options and the analysis of the impacts of those options. Welfare change refers to how much better or worse the public is under the various management options. We may try to monetize those changes. Impacts refer to how those options may alter general economic conditions, primarily employment and spending. Welfare change and impact are not the same, as the terms are used by economists. An example will illustrate the difference. Suppose an option for forest use is to increase the timber cut. Impacts of this option would include increased local employment and spending, increased national timber supply, and reduced local recreational activities. Net employment impact would add increased timber jobs and reduced recreation-related jobs. These are "impacts" and do not represent welfare gains and losses in the sense of monetized costs and benefits associated with the option. Welfare gains and losses, or the costs and benefits of this option, refer to the improvements or reductions in public well-being. The welfare gains from increased timber supply would be measured by reduced timber prices to buyers and increased incomes in timber-related industries. The welfare losses would be measured by lost recreational enjoyment, increased costs of finding recreational alternatives, and reductions in recreation related incomes. Options that have positive impacts on, say, jobs may have negative net welfare results; i.e., the costs of the option exceed the benefits. Cost–benefit analyses are designed to measure the net welfare effects of decisions, not impacts.

4 USING VALUES AND IMPACTS IN MANAGEMENT DECISIONS

Management decisions are typically going to be made based on traditional cost–benefit analysis, economic development, and equity. This section suggests means of incorporating these factors in management decisions. Cost–benefit analyses use the consumer and producer surplus valuations noted above to consider the net monetized welfare gains or losses from a decision. Deriving those values can be a useful exercise in understanding the multiple values of ecosystem change. These values must include non-user as well as user values: multiple use of ecosystems must include non-use as well.

Bottom-line net benefits or costs are not enough to make the hard management decisions. All decisions will have different effects on the various stakeholder groups. These decisions will ultimately be political in nature, requiring an understanding of exactly who gains and who loses. Decision-based cost–benefit analysis must maintain an accounting of gains and losses by stakeholder groups. For example, a decision on a forest cut will have recreational groups pitted against local loggers and timber firms. These groups may differ geographically, and even temporally if recreational use is sustainable and logging is not. Ecosystem management problems often involve geographically widespread and diverse benefits, but highly localized costs. Knowing the magnitudes of the gains and losses of stakeholders helps in weighing the equity issues.

Economic growth and development are factors that will invariably enter management decisions. Jobs and spending impacts will be considered in addition to cost–benefit analyses. When this is the case, the manager must be careful to consider the net impacts of decisions. Some jobs may be gained, and some lost; some spending will increase and some will decrease. Timber jobs will be saved or created, but opportunities for recreational industry jobs may be lost. Making matters worse is the likelihood that some jobs may not last long; and jobs lost may have been long-lasting. A timber cut that creates a short blip in a large number of employment opportunities may result in a loss of a smaller number of perpetual jobs in recreation. The boom-bust of the one-hit harvest employment may create public service and public welfare problems during the bust period. How can we compare the short term large gains with the long term, but smaller losses? Using discounting of jobs may help. Another method is to confront the community with the issue in the context of its visioning process.

It is bad policy making when options are judged on the basis of the significance of an industry to a region. The important issue is how much *change* will occur as a result of the management option; i.e., what *difference* will the option make to whatever is important, be it welfare, jobs, or spending. To say that X persons are employed in an industry and earn $Y is irrelevant to considering the option; the critical issue is how many *more or less* persons and how much *more or less* income will result from the option. Of course, public debate frequently becomes focused on significance because the numbers are likely to be considerably greater than the changes that will be induced by the management option.

5 CASE STUDY: VALUING FOREST SERVICES

To provide the reader with insight into issues of valuation, this section works through an example of valuing forest services. The first problem in valuation is to define the services rendered by an ecosystem. In general, these services include provision of goods, such as timber and fish, maintenance of life support systems and cycles, such as nutrient and hydrologic cycles, and non-instrumental values, such as aesthetics and culture. An elementary breakout of services provided by forest ecosystems might include (Myers 1997):

- Extractive materials (timber, fuelwood, minerals, etc.)
- Extractive species (wildlife, wildfowl, fish, nuts, mushrooms, etc.)
- Hydrologic cycling (flood control, runoff control, water quality, etc.)
- Nutrient cycling (soil fertility, forage, habitat, etc.)
- Soil creation
- Sediment retention
- Local and global climate moderation
- Biodiversity
- Pest and disease control
- Landscape value (aesthetic, cultural, spiritual, etc.)

This list is arranged with the most obvious, concrete, and potentially marketable services listed first. These are highly instrumental values associated with direct use in the economy. The list includes a complex set of services associated with a functioning ecosystem. Before addressing valuation methods and examples, it is important to note that social attitudes toward the values of ecosystems may be shifting over time. For example, Booth (1994) notes that public arguments for protection of old-growth forests have been shifting from focusing on instrumental values, such as recreation, to more natural integrity arguments, such as preservation of ecosystems, watersheds, and species. These arguments were couched in terms of wilderness being valuable in its own right. These trends may reflect emergence of existence and option values, a de-emphasis on instrumental and narrowly economic values, or a recognition of the complexity of ecosystems and the necessity of preserving the wide range of their services.

5.1 Extractive Materials

The first category's values can be reasonably estimated using traditional market based values, such as stumpage rates for timber, net prices for minerals, and net price for fuelwood. Net price reflects market value minus the cost of harvest and bringing to market. The value of the resource alone is its net price, not the revenue collectable from its sale, since a portion of that revenue covers costs of human inputs. Commercial materials extraction valuation is not so simple when large tracts are involved. When potential extraction or harvest is so great that market prices for related commodities are affected, these impacts must be valued also. For example, a massive timber cut could lower consumer prices, creating consumer values. This requires an estimate of how much prices would fall.

5.2 Extractive Species

The second category's values are more complex. These values originate in the hunting, fishing, and harvesting activities, both commercial and recreational. Commercial values are measured as the net profits from the extractive activities. These profits are the difference between market prices and costs of harvest and bringing to market; i.e., revenues minus costs, not just the revenues from sales. Recreation values have both commercial and consumer value components. The recreation industry obtains profits from forest related recreation, both extractive and non-extractive. These profits would be included in the value of forest services. It is most important in measuring commercial values to recognize that profits are the measure of value, and not simply sales revenues. The valuation argument is that commercial vendors are worse off or better off according to whether their profits are diminished or enhanced, not by what happens to their revenues.

Recreational consumer valuation is more complicated than commercial valuation. This is because values are not directly observable from direct market purchases. A range of valuation methods is available. The travel cost method (TC) recreates the cost of visitations to sites using distances and time for travel. It then creates a demand function for the site or sites, and this is used to estimate the consumer surpluses associated with site visitations. It is important to note that the cost of travel to the site, including transportation and time costs, reflects the cost of visitation, and the benefits of visitation must be greater for someone to decide to visit. The tricky valuation issue is that the value of the site for recreation is the difference between the benefits and costs of visitation. This is a measure of what would be lost to the user if the site were not available; i.e., if the benefit was $100 and the visitation cost $80, the measure of value is only $20. It is often the case that public debate will use the visitation cost measures to reflect the recreational values of sites. This is inappropriate in the sense that any attempt to determine how much better or worse off society is with or without a site should use the $20 measure. The travel cost method uses observations over a large number of visitors over a wide region to establish the benefit measure ($100) and subtracts the cost of visitation ($80) to estimate the value ($20). The $20 is the consumer surplus associated with the site.

A more direct valuation procedure is the contingent value method (CV). It asks respondents what they would be willing to pay for something, above and beyond what is already paid. It is a direct measure of consumer surplus.

The TC and CV can be used to estimate the value of a site for any type of travel related recreation, fishing, hunting, hiking, birdwatching, etc., as well as the value of changes in site qualities. Both valuations are important. While typical ecosystem management issues are changes in ecosystems that improve or degrade certain services, knowing the value of a site is useful in gauging the magnitude of values at risk from any change. The valuations of ecosystem changes to recreational users are complicated by the fact that these changes will induce increases or decreases in the values of visitations as well as the number of visitations. Both effects, value changes per visit and visitation rate changes, must be estimated.

The classic recreational values for forest use are "Unit Day" values, which are TC and CV derived measures of consumer surplus associated with a one-day activity. A wide array of activities have been valued on a unit day basis, including camping, picnicking, swimming, cold water and warm water fishing, etc. These values can be useful in estimating the values of management options that alter usage rates. For example, a day of cold water fishing has a unit day value of roughly $30 per day per user (Walsh et al. 1990). A forest management decision leading to a decrease in fishing day visits can be evaluated with this type of number. It is not useful in evaluating changes in the quality of experience; i.e., the unit day value would fall to something less than $30.

The following are some examples of the use of TC in recreational valuation. A simplified method and illustration of using TC for recreational valuation is provided by Bowes and Krutilla (1989, pp. 215-247). They have used easily available data from the US Forest Service to value visitations to the White Mountain National Forest. Statistical regressions were run to establish visitation demand functions. Their estimates of consumer surplus values for access to this site ranged from $13 to $16 per person per visit, depending on the primary purpose of the visit. These estimates are on the low side since they could not estimate the time costs of travel. Assuming 1.5 days per visit, these values translate into $8.70 to $10.80 per recreation visit day (RVD). Based on 2.4 million annual RVD's (1983) the TC based estimate of the value of access to the site was $20.9 to $25.9 million annually. The net value of the forest for recreation would be these values minus whatever costs were necessary to provide these recreational services.

Costs are not simple to estimate for a multiple-use forest. Bowes and Krutilla estimate the net value of forest for recreation to be roughly $26 per acre per year, with a discounted present value of $650 per acre using a 4 percent discount rate. This value can be updated to current dollar values by using a price index for adjustment. They also estimated the changes in value (consumer surplus) from changes in the quality of the forest. For example, they suggest that a 10-percent improvement in scenic quality of a site will increase valuation for picnicking, fishing and hunting, or hiking by 6 to 8 percent.

The above valuation information can be used in several ways. Bowes and Krutilla (1989) estimated that, under the most productive conditions, the present value of the forest for timber after deducting management costs, was at most $100 per acre using a 4 percent discount rate. This means that if the ecosystem management choice is between recreation and timber, the recreation value is over six times as great; recreation is thus the most efficient use of the forest. However, management issues are seldom either/or decisions. It is more likely that the issue will be more versus less timbering. In this case, the efficiency question is whether the change in recreational value is greater than the change in timber value. A cut that would reduce scenic quality by 10 percent would reduce the recreational value of the forest by $39 to $52 per acre.

One of the major theoretical and practical problems in implementing the TC is how to incorporate multiple destination trips. Travel costs are complicated in this case. Mendelsohn et al. (1992) have proposed a method of grouping sites for purposes of estimating trip demand functions. Using the example of Bryce Canyon National Park, they estimate the consumer surplus from a typical single destination user to be approximately $10 per person per trip, and from a typical multiple destination user to be $17 per person per trip. In other words, multiple destination users place higher values on Bryce visits than single destination users. They note that Sorg et al. (1985) found multiple-destination, cold-water fishermen in Idaho placed higher valuations on sites than did single-destination users.

Both sites and activities can be valued. For example, Vaughan and Russell (1982) have valued a fishing day using a variant of TC. Their varying parameters model allows them to estimate the value of fishing days under different site qualities; in their case, by primary species supported by a site. For example, they estimate the value of a trout fishing day to lie between $16 and $24 per day (1979 dollars); and the value of a catfish fishing day to lie between $10 and $16 per day. These values can be indexed up to current dollar values using a price index. These are consumer surplus estimates; i.e, the value of the fishing benefits above the costs of travel. They note that these estimates are similar to contingent valuation estimates. These valuations can be useful if an ecosystem management decision will result in the increase or decrease of fishing visits. A decision that will degrade streams and result in reduced angler visits can use these types of values to estimate the costs of that decision.

Vaughan and Russell (1982) use their model to estimate the marginal value of a fish caught. For example, the value of one more trout caught per angler was $0.45 per fish per angler; and the value of a catfish was $0.31 per fish per angler (1979 dollars). This measure of value is also useful in valuing the costs or benefits of ecosystem management decisions. For example, a decision that will improve streams and increase catch can be valued. The per day (unit) values and the per fish values can both be used if the decision changes visitation rates as well as the catch experience. These valuations are related to forest ecosystem management if stream quality is impacted by management decisions, as in the case of sediment and nutrient releases from timber and road cut practices. Of course, a coupling between physical inputs into streams and fish catch experience must be made.

The CV method has been used to estimate the loss in values from elk hunting in response to a proposed timber cut and associated road construction. A CV study for Gallatin National Forest established the willingness to pay for varying probabilities of elk sightings. While the existing consumer surplus associated with hunting the site ranged from $317 to $376 per trip, the value of an elk hunting trip to this site increased $108 per trip for double the chances of an elk sighting (and likely kill) (Loomis 1993). This information was used to establish that the proposed timber cut, which would reduce trophy elk populations 5% each decade, would result in a loss of present value of $405,000 (1978 dollars) in elk hunting alone.

TC and CV methods can be combined for useful valuation procedures. For example, Layman et al. (1996) estimate the value of a salmon sport fishery under different management options by surveying how visitations to the Gulkana River, Alaska, would change under different conditions. This hypothetical travel cost method establishes how the travel cost based demand function for site visits changes under different hypothetical conditions. For example, they estimate the current consumer surplus per day for salmon fishing on this river to be roughly $32 per person per day, using 60 percent of the wage rate as a measure of travel time costs. However, if the harvest rate was double the current rate, this value increases to roughly $44 per person per day. This increased abundance could result if commercial catch was limited, so this type of valuation is useful for estimating the recreational value of reduced commercial catch. A reduction in commercial catch that would double recreational catch would have a recreational value of $12 per person per day. Knowing the number of fishing days per year and lost commercial profits would allow a comparison of recreational value created with commercial value lost. The study evaluated other fishery management options, such as increasing bag limits.

Hedonic models of travel costs have also been used to value site qualities and can be useful in valuing changes in qualities as a result of management decisions. Recreationists should be willing to travel further for higher quality experiences. Observing how much more people are willing to pay, in travel and time costs, to visit higher quality sites provides a measure of the value of that higher quality. Englin and Mendelsohn (1991) and Wilman (1984) have developed and applied this method. For example, Wilman uses the method to value forest vegetative characteristics on deer hunting. A measure of deer habitat quality and probability of hunting success (a measure of forage) as well as aesthetic visual variables reflected site quality. A timber sale in the Black Hills was designed to improve deer habitat. The sale would increase forage and reduce travel costs to high quality hunting sites. The consumer surplus value of this sale was estimated by Wilman to be between $63 and $71 per year per hunter from one of the study towns (1980 dollars). There were 844 Black Hills hunters from that town, implying an annual aggregate benefit of the sale of between $53,000 and $60,000 just to hunters from that town. Values to hunters from other towns were estimated also. Of course, this consumer surplus value of the sale would diminish over time if the stand regenerated, but may be partially offset by an increase in hunters.

Englin and Mendelsohn (1991) used the hedonic travel method to value wilderness recreational site attributes, such as old-growth, campgrounds, and views, in Washington wilderness areas. For example, they estimated the willingness to pay for an additional mile of old-growth forest was $2.61 per mile per trip, and the value of an additional campground was $7.10 per site per trip (1990 dollars). Given the existing miles of old-growth, they estimated the consumer surplus from the existing configuration of old-growth forest to be $72 per trip per year. The consumer surplus for all existing campgrounds was $180 per trip per year. Since there were 125,000 trips per year into the studied wilderness areas, the value of existing old-growth was $9 million per year. If this configuration was maintained indefinitely, the present value of old-growth in these wilderness areas would be $225 million. On the margin, the loss of 1 mile of old-growth would result in the loss of $362,250 per year ($2.61 × 125,000) and a present value of that loss would be $8.2 million.

Recreational uses may be in conflict. Bikers, hikers, campers, boaters, hunters, and fishermen can create conflicting uses. A wilderness example of conflict is instream flow conditions for fishing and rafting. In a study by Naeser and Smith (1995), fishermen using the Arkansas River claimed that fishing value is optimal at flow levels around 450 cfs, while boaters claimed the highest value at flow levels between 1100 and 1500 cfs. A contingent valuation survey of anglers revealed a willingness to pay of $2 per person per day to fish the Arkansas. This is a consumer surplus above actual expenditures, which were roughly $30 per day. A similar survey revealed that boaters were willing to pay $3.50 per person per day to boat the Arkansas. This is the boating consumer surplus above actual expenditures, which were $51 per person per day. The management question is what level of flow to maintain at what times. Knowing what the visitation rates would be for each use at different times of the year would allow a comparison of aggregate fishing and boating consumer surpluses. An efficient ecosystem management rule focused only on consumer surpluses would manage flow rates to support that activity with the greatest surplus at that time. Of course, other flow-related uses, such as drinking water and irrigation supplies, would have to be incorporated also. Also, flow effects on long-term river quality and ecosystem health would have to be considered. Sustained high flows may alter fisheries habitat, reducing the consumer surplus from fishing the river. This would be an additional cost of supporting boating flows.

Conflicting uses should not necessarily be resolved based upon size of related industry arguments. For example, Arkansas River commercial rafters spend \$51 per day compared to \$30 per day for anglers, and commercial rafting revenues are \$30 million per year compared to fishing related revenues of only \$3 million per year (Naeser and Smith 1995). These numbers alone suggest favoring instream flow conditions for rafting. However, these expenditures do not reflect profits or consumer surpluses, nor do they reflect how spending, profits, or consumer surpluses would differ under different instream flow conditions. Profits generated per fishing day may exceed those from boating; and restricting favorable rafting flow days may simply result in more intensive use on high flow days with no revenue effects.

5.3. Biodiversity

The biodiversity value of forest ecosystems and associated species has many components, including potential medical value, particular species value, resilience and integrity value, and pure existence values. The valuation of conservation of a forest ecosystem and associated species is represented by the contingent valuation study of Hagen et al. (1992). Depending upon a variety of assumptions, they estimated the value of conservation of all spotted owl related old-growth forests in the Pacific Northwest to lie between \$48 and \$144 per household per year in the states of California, Oregon and Washington. Mead et al. (1990) had estimated the economic cost to this region of reduced timber harvests that would result from old-growth preservation on public lands. This cost was due to reduced timber supply driving up prices, resulting in losses in timber market consumer surplus. This loss was estimated by Hagen et al. (1992) to be \$3.39 per household per year in the three state study region. The value of harvest loss was dwarfed by the value of conservation.

The old-growth example illustrates the potential usefulness of establishing the costs associated with a decision. If benefits of old-growth preservation are unreliable or unavailable, knowing the costs provides some gauge against which to make a management decision. In the above case, the cost of foreclosing timbering of old-growth forest on public lands was only \$3.39 per household in the study region. Benefits would have to be at least that to make a conservation decision most efficient. Montgomery et al. (1994) establish an even more accurate estimate of the opportunity costs of conserving old-growth forest for spotted owl habitat. Noting that saving species is uncertain, they estimate the costs of various probabilities of saving the spotted owl. These costs result from consumer and producer surplus losses accompanying necessary reductions in timbering of old-growth forest habitat. Their assessment of one conservation strategy that would provide a 0.9 probability of owl survival suggests a net welfare loss of \$33 billion per year (1990 dollars). They disaggregate these costs by region, since reductions in harvest in the Pacific Northwest will be partially offset by increased harvests in other regions. Disaggregation by industry groups suggests gains to private stumpage suppliers (\$25 billion), since the conservation plan studied is limited to public lands, and losses to intermediate and finished goods producers (−\$63 billion). In addition to evaluating the conservation costs, they estimate welfare costs of varying probabilities of owl survival. For example, a strategy that would provide a 0.8 probability of survival has a welfare cost of \$21 billion. This means that increasing the probability of survival from roughly 0.8 to 0.9 requires an additional welfare costs of \$12 billion per year.

5.4 Local and Global Climate Moderation

Climate services of forested ecosystems are of considerable value. These services include localized microclimate effects as well as global effects. The avoided cost method of valuing these services is illustrated for the carbon sequestration value of forests. For example, Sedjo and Solomon (1989) estimate that a forest will annually sequester 6.2 tons of carbon per hectare during its growth phase. A 30-year rotation period results in total sequestration of 186 tons per hectare, and a 50-year rotation results in total sequestration of 310 tons per hectare over its life. The National Academy of Sciences (1991) has estimated engineering control costs in the United States of roughly \$36 per ton of carbon for low cost options (excluding options such as energy conservation that result in cost savings). Increasing the forest rotation period from 30 to 50 years would sequester 124 more tons of carbon per hectare. This would result in a total carbon alternative control cost savings of \$4464 per hectare. The present value of these savings would be smaller than this; for example, assuming the 6.2 tons per hectare sequestration annually, waiting 20 years to cut a growing forest would result in a present value cost savings of \$3033 per hectare using a 4% discount rate.

5.5 Sediment Retention

The value of forests for soil retention can be estimated through effects on enhancing stream quality, lengthening dam and reservoir lifetimes, reducing needs for dredging, and reducing damage to mechanical equipment used in water resource development projects. For example, the useful life of a hydroelectric dam would be shortened resulting in increased electricity costs, if forests were harvested. Southgate and Macke (1989) have estimated that increased dredging costs and reduced hydroelectric dam lifetimes due to sedimentation from agriculture and silviculture in an Ecuadoran watershed result in economic losses. The present value of benefits of various conservation programs range from $15 to $39 million (1989 dollars).

5.6 Other Non-Instrumental Values

While the above values are generally concrete, instrumental types of values, there may be non-instrumental reasons for placing values on ecosystem conditions. For example, Sanders et al. (1990) used CV techniques to establish both recreational use values and non-use values, including option values (payments to preserve the future opportunity to use a resource), existence values (payments for preservation in the absence of any interest in use) and bequest values (payments for use by future generations) of a set of potential wild and scenic rivers in Colorado. The non-use values ranged from $32 per household for protecting three rivers to $82 per household for protecting 15 (1983 dollars). Non-use values were roughly four times as great as the recreational use values. Excluding non-use values would seriously underestimate the value of preservation of the potential wild and scenic rivers in their study.

6 CORRECTING FOR INEFFICIENCIES

While some would argue that private ownership of ecosystems and their resources is adequate to attain their efficient use, the structural failures within private ownership based economies outlined above suggest inefficiencies that may need to be remedied. These inefficiencies result in either too little or too much activity or resource use compared to what would be required for attaining highest and best use of the ecosystem. There is a wide array of methods available to correct for these inefficiencies. These include directly regulating the activities or resource uses, or correcting the causal failing factors that lie behind those activities and uses. For example, society could correct for pollution by proscribing or prescribing certain behaviors, placing prices or subsidies on those behaviors, or more fully defining and enforcing the assignment of property rights violated by the behavior.

We can refer to methods of correcting for inefficiencies as instruments. Instruments generally have either monetary or physical dimensions, or both. For example, forest harvest permits may be granted stipulating the size of trees to be harvested, that harvesting must comply with water regulations, and noting which violations of harvest conditions will result in penalty assessments. Instruments differ considerably in the extent to which they constrain behavior, or intervene in detailed decisions of affected parties. They may range from simply providing information to dictating technologies. The selection of instruments is complicated, depending upon costs of administration, compatibility with interventionist goals and human behavior, costs to affected parties, and political constraints. In some circumstances, instruments themselves may have greater welfare or political costs than the welfare benefits obtained from their utilization.

The costs of instrument utilization have become a major focus of attention when attempting to correct inefficiencies. This is particularly true of what are termed "command-and-control" instruments. These instruments prescribe in some detail the behaviors that must be satisfied, such as the number and times during which cattle can graze an ecosystem, or dictating what trees must be cut and how they must be cut. The purpose of such intensive command-and-control is often to assure that intervention goals are attained. Uncertainty in the attainment of a goal is a major factor for several reasons. First, more subtle instruments may leave too much room for behavioral discretion to assure effects. Second, there may be some circumstances where considerable ecosystem damage can be done if behavior exceeds certain bounds. This is why society bans certain extremely harmful substances. It would also be the case for activities that can irreversibly alter ecosystems, such as some types of logging activity. Finally, the command-and-control may provide for clear and verifiable behaviors, making it clear to regulated parties what they are to do, and making it clear to the regulator when the regulated party has violated the standards. For example, it is clear whether a firm has an electrostatic precipitator on a stack, albeit not so clear that it is used fully. The costs of such intensive and intrusive control are that more efficient means of attaining the same ecosystem goal may be undiscovered or disallowed. This is a particular problem when the same "one size fits all" control is used across a wide variety of circumstances; for example, permitting a fixed number of cattle per acre of wilderness across all wildernesses, when the capacity or ecosystem sensitivities vary.

The target of any control instrument is typically some well-identifiable element or behavior in a chain of activities eventually leading to the environmental or resource problem of immediate concern. For example, the number of cattle each rancher is allowed to graze is regulated when the ultimate concern is the condition of the range or forest. An alternative to the command-and-control instrument in this case is to require that certain range or forest quality conditions be met. The usefulness of this performance-oriented instrument would depend upon the cost of enforcing it and the possibility that its failure could result in irreversible ecosystem degradation. Another alternative would be to simply set a price for each animal or tree and let the user determine the use. This has the same potential failings as the pure performance instrument.

Economists and others have expressed increasing concerns about highly centralized, dictatorial, and interventionist command-and-control instruments. While they may be necessary in some circumstances, such as outright bans on the production and use of highly toxic materials, they are not always necessary to attain goals. There is increasing interest in decentralized instruments that achieve the same goals but at lower social or political cost. Lower costs result from their less intrusive, controlling nature, and from their ability to be flexible enough to allow variations in behaviors appropriate to different circumstances. For example, grazing fees as a control instrument can be tailored to different cases, reflecting the costs of grazing in different ecosystem contexts. Management control instruments that are not command-and-control would include the following:

1. Negotiation instruments: bargaining, conflict resolution
2. Liability instruments: fines and penalties, injunction and sentencing
3. Pricing instruments: taxes, fees, deposit-refund, performance bonds, or subsidies
4. Market emulation instruments: tradable permits

Each of these will be discussed below in the ecosystem management context.

6.1 Negotiation Instruments

This type of instrument is included for its general use in reaching highest-valued uses without heavy-handed intervention. Its success relies on both the compatibility of private and public goals, and the ability of parties to reach mutual agreement on transfers of property rights. Active intervention is limited to the extent that parties will find a mutually agreed upon allocation of use to the ecosystem under conflict. It relies on the theoretical Coase (1960) notion that if a higher valued use exists, it will be attained through private parties' willingness to pay and accept compensation for transfers of rights and uses.

There may be impediments to negotiated settlements. Costs of reaching agreements may exceed any gains. Facilitation of negotiating processes may be the proper role for the ecosystem manager in such cases. This may be the problem when there are many parties involved, such as recreationists versus ranchers in forest grazing conflicts; or when there could be a wide range in the distribution of welfare resulting from particular outcomes, such as environmentalists versus loggers in an endangered species conflict. In extreme cases, negotiated conflict resolution is not possible.

Negotiated private settlements may be feasible but may not fully attain highest use of an ecosystem. Highest use requires consideration of all uses and their values. Private negotiations may exclude some valued uses or costs, especially when there are spillovers or public-goods type values involved. In these cases, private highest uses will not reflect highest social values, making private negotiations inefficient. Examples would include rancher–recreationist negotiations for forest use when private negotiations fail to consider downstream water quality impacts; or negotiations over timber harvest rates when global carbon sequestration benefits of forests are not considered.

6.2 Liability Instruments

The role of these instruments is to enforce explicitly proscribed activities, or to enforce predetermined rights. For example, if management rules are that cattle cannot be within a certain distance of a stream, violation of this sanction incurs a penalty. Or if a farmer's land is trespassed upon by wandering cattle, the farmer has a right to compensation. The liability instrument extracts a penalty, monetary or otherwise, for doing something disallowed. Therefore, it differs slightly from a tax that must be paid in order to engage in the same activity, the latter being interpreted as a price that must be paid for doing something that is allowed.

Liability instruments rely on the rational calculation of individuals to realize that violations incur penalties, but only if they are apprehended and actually penalized. The rational individual will consider both the magnitude of the penalty, the probability of being apprehended, and the probability they will be penalized if found guilty. High-potential penalties have two contrasting effects as an incentive instrument. On the one hand, they raise the expected costs of violation if enforcement remains constant; but they may reduce the likelihood that an administering body, such as a court, would actually levy such a fine. The net result of high penalties is indeterminate. Furthermore, managing agencies may consider a high penalty a substitute for enforcement, thinking the high fine will discourage behavior, and reduce enforcement activity; i.e., higher fines and fewer rangers. This may result in lower probabilities of apprehension and lead to greater violations than if lower fines and more enforcement were used.

6.3 Pricing Instruments

These instruments define acceptable activities and place a price on those activities. Where activities are undesirable, the proper instrument is a positive price. This would include a tax or fee. It would include a deposit required, with a refund when proven that an adverse activity was remediated. It would include performance bonds, which would be posted, and drawn upon when there was proof of unremediated damage. There are typically legal distinctions between taxes and fees, where the latter are generally for purposes of compensating for services rendered by public agencies or for values received from the public, such as the case of severance fees. They all require an accurate definition of the priced activity, or the conditions for refunds or withdrawals from the bond fund.

Economic efficiency principles would dictate that the taxes or fees be set at levels that reflect the opportunity costs associated with the priced activity. For example, if cattle are allowed to graze in a forest, and this activity does $X in damage per animal, the proper price would be $X per animal. If an animal consumes $Y of fodder on public lands, a severance-like fee would be $Y. Animals consuming $Y of fodder and doing $X in additional damages would bear a price of $X+$Y. The efficiency principle is that when the user is forced to bear a price equivalent to the opportunity costs of the resource, the user will only do so if the activity value to the user exceeds that cost. If so, the highest use value lies in grazing; if not, the highest use value lies elsewhere.

Although simple theoretically, in practice the determination of opportunity cost is complicated. Furthermore, there may be some public benefits from the users activity that do not accrue to the user and would not be considered by the user in deciding whether to pay the price. If positive spillover benefits were $Z per animal, this problem can be solved by offering a price of $X+$Y–$Z to the user. Evaluation of X, Y, and Z requires use of valuation methods outlined above.

This efficiency pricing differs from pricing to attain maximum revenues. If maximum revenues are desired, prices for access would be set considering demand responses; e.g., based on cattle grazed or trees felled and responses of those quantities to prices. The revenue maximizing price may be greater or less than the efficiency price. For example, if the highest price a rancher will pay to graze cattle is less than the damage it would do, it is not efficient to graze cattle, but denying grazing would mean revenues lost. Ecosystem use pricing is frequently based on revenues and not on the principles of inducing highest and best use.

Where activities are desirable, the opposite type of instrument may be appropriate. Subsidies of some activities may be required to attain the highest use values. This would be appropriate where positive spillovers from activities occur. Subsidies can also be used to reward a user for not undertaking an activity that is harmful. For example, timber firms can be paid not to harvest old growth forests; or ranchers can be paid not to graze their cattle on public lands. The danger of using subsidies to encourage termination of certain behaviors is that it may create a problem of moral hazard. Recall the classic moral hazard problem where availability of health insurance discourages healthy behavior. Subsidies to discourage "unhealthy" behaviors may induce people to engage in more "unhealthy" behavior simply to receive a larger subsidy. Paying ranchers not to graze their cattle may induce them to claim they would have grazed even more cattle than if there were no subsidy. Paying farmers not to farm some lands induces them to claim they would have farmed more land.

Performance bonds, or other escrow type instruments, require upfront obligations of funds with reimbursements or reduced obligations depending on some performance criteria. An example is bonding requirements for surface mining and construction. Applications could be generalized; for example, bonding of ranchers for grazing cattle on public lands. A variant of this instrument is deposit-refund systems. An example is bottle deposits. The criteria for the use of these types of instruments is that reimbursement conditions be clearly set up-front. This is more difficult in the case of ecosystem management than bottle returns or construction, since performance criteria are more complex.

6.4 Market Emulation Instruments

These instruments achieve regulatory goals through the use of market-like mechanisms. The markets are primarily to allow the shifting of responsibility for goal attainment from high cost to low cost compliance entities. The classic example is tradable emissions allowances or emissions reduction credits. The regulator sets the total volume of allowances or reductions, allocates these across sources through issuance of permits, then allows the purchase and sale of these permits. Success of such an instrument requires clear establishment of the rights the permits entitle their bearers, the ability to monitor compliance with permit transfer conditions, and a well-functioning market with enough traders to avoid monopolistic behaviors.

For example, tradable grazing or timbering rights may be successful where there are an adequate number of potential traders, possibly including environmental groups. It must be clear that a permit allows the grazing of so many cattle for a specified time, and the number of cattle and grazing periods are verifiable.

An interesting feature of tradable permits is that allowing buyers to take permits off the market is a method of attaining highest valued uses of resources. For example, allowing environmentalists or recreationists to buy grazing rights would let them bid against ranchers for access to public resources. If environmentalists are willing to bid more for the rights than the ranchers, the resource passes from a lower valued use in grazing to a higher valued use in preservation. There is no technical reason why environmentalists or recreationists should not receive a share of the initial allocations. Initial allocations are critical to the scheme primarily because they determine the final distribution of welfare resulting from the scheme. For example, giving all the rights to the ranchers gives them all the initial assets, and they will trade only if it enhances their wealth. The welfare of the environmentalists or recreationists is only enhanced by their ability to buy permits at less than their value to them.

Initial allocations can change final outcomes, hence efficiency in the use of the ecosystem, for two reasons. First, allocations determine initial wealth and, possibly, the ability to bid or willingness to sell. For example, giving the rancher more initial permits increases real wealth and, perhaps, the ability to finance further purchases; a banker may be more willing to make a loan to the rancher. Second, there are transaction costs associated with trades in the permit market. Unless it is very well organized, such as commodity exchange markets, buyers and sellers may have difficulty finding one another. This reduces the volume of trades, and the ability of resources to transfer to their highest valued uses.

7 THE ROLE OF ECOLOGICAL ECONOMICS (EE) IN ECOSYSTEM MANAGEMENT

Ecosystems are complex combinations of interconnected structures and processes: forests are trees, soils, streams, birds, nutrient cycles, carbon cycles, etc. To human economies, they are assets yielding various services. Yet they yield up their services in complex and often unpredictable ways. We try to manage them to serve our purposes, often without adequately considering their complexities. We use our values to extract services, often without adequately considering natural system values. We often weigh one use against another, without adequately knowing how the structures and processes yielding those uses are connected in a natural system. We simply do not pay enough attention to the rules and laws of the natural systems we seek to exploit. Ecological economics has emerged as an interdisciplinary attempt to address these deficiencies.

Although it is somewhat difficult to exactly pinpoint the origins of ecological economics, it is reasonable to suggest it was spawned by the recognition that natural processes place impossibility rules on human economies. One of the first spokespersons for this view was Boulding (1966), whose "Spaceship Earth" image was based on the notion that "the earth has become a single spaceship, without unlimited reservoirs of anything, either for extraction or for pollution, and in which, therefore, man must find his place in a cyclical ecological system which is capable of continuous reproduction of material form even though it cannot escape having inputs of energy." Ayres (1978) took the First Law of Thermodynamics (conservation of energy and matter) and implemented a mass-balance approach to the analysis of economic processes. Georgescu-Roegen (1971) more formally considered the implications of the Second Law of Thermodynamics (disorder, or usable energy, increases in a closed energy system) to economics. He argued that the economic process converts low-entropy resources into high-entropy waste, and that the economic process is not an isolated, circular affair of spending in which the economy can merrily proceed without facing constraints from natural systems. It is anchored in a material base, which is subject to definite constraints. Entropy and material–energy constraints place an ultimate limit to growth, suggesting an ultimate absolute scarcity of usable resources.

The implications of these biophysical constraints for human economies are profound. They imply an ultimate carrying capacity of the ecosystem for human economic activity, population, and quality of life, whereby the only salvation is requisite changes in consumption patterns or technological changes in the use of earth's materials and energy supplies. Daly (1977) advanced these lines of argument by proposing the necessity and desirability of a "steady-state" economy in which the stocks of people and their artifacts (cars, buildings, etc.) are maintained at some desired, sufficient levels by low rates of throughputs of matter and energy. Growth in artifacts would be replaced by qualitative "development" in enhancing quality of life.

While these arguments were framed in larger, macro scales, they have analogous implications for smaller scale ecological–economic management. The essence of the arguments is that human economies are bound by their ecosystems and its biophysical rules. Economies are dependent on matter and energy flows from ecosystems. The conversion of low-entropy, highly organized ecosystem resources, into high-entropy waste irreversibly alters the ecosystem. The non-isolated, non-circular character of economic processes requires that considerable attention be paid to the feedbacks from economies to the ecosystems in which they are embedded; i.e., there may be unexpected costs associated with altering ecosystem structures and processes. Simple ecological examples abound: deforestation reduces tree cover and litter, which reduces soil nutrient stocks and flows, changes soil temperatures and cohesiveness, increases rain impact of soils and reduces water uptake, causing erosion which fills in streams, which reduces stream capacity, which causes more flooding, which creates costs to the economy, and so on. Another striking example is the effects of grazing on western pine forest systems. Biologists are recognizing that grazing of forest undergrowth grasses has increased pine seedling growth and resulted in an extensive carpet of incendiary pine needles that shifts fire regimes from beneficial surface to devastating crown fires. Furthermore, terpenes in these needles interfere with the bacteria that convert nitrogen in dead wood into usable nutrients. The needles create a dense mass that is less water permeable, blocking pathways for water into the ground. This would be consistent with observed slowdowns in flows from springs. The economic activity of grazing has had a dramatic, adverse impact on forest structure as well as hydrologic and nutrient cycles; the cost of grazing may be far greater than we have imagined.

The work in ecological economics has been intended to heighten concerns about the sustainability of traditional economic growth and ecosystem uses, and to make us think much more deeply about the implications of our economic actions on ecosystems and, therefore, on economies. This is not to say the conventional economics cannot address these direct and indirect impacts of economic activities; it is to say that persons more skilled in understanding natural systems can contribute significantly to the economic analysis of trade-offs and costs and benefits of various resource management options.

The traditional exploitative ecosystem–economic model has been unidirectional, seeking services and materials from ecosystems according to their values to the economic system; forests are valued for their trees. But economies are merely sub-ecosystems embedded in a larger ecosystem. Economic possibilities are constrained by the rules of their larger ecosystems. These rules include feedback processes; while using their parent ecosystems, economies have impacts on these ecosystems through alterations in their structures and processes. Cutting trees and building roads change forest hydrology, soil properties, and nutrient cycles. Ecological economics highlights these feedback processes; i.e., makes the ecosystem–economic models more ecologically valid.

Stemming from the simple unidirectional model of economies and ecosystems is the naive notion that the ecosystem can be "managed" for human use. This notion is in spite of increasing recognition of the complexities and unknowns in the ecosystems we seek to manage. Recognition of feedback processes and their often unknown effects suggests management at another point in the ecologic–economic system: managing human economies. Conventional economics has recognized that human economies can create harmful spillovers in the process of doing what economies do. Corrective measures, such as proscriptions, taxes, penalties, etc. have been suggested. Ecosystem management must exploit the adaptability of human economies; huge homes requiring massive quantities of trees will be less desirable if their prices are higher. Ecological economics attaches a high level of significance to management solutions involving the adaptability of human economies to given ecosystem conditions and processes.

Greater recognition of interrelations between economies and ecosystems results in greater depth of understanding of the trade-offs in ecosystem use; deforestation is not simply the choice between cutting trees and seeing a rare bird. Far more complex events occur when ecosystem structures and processes are altered; fisheries and climate may be permanently altered. Far more values are at stake than simple use values of trees. Cultural values associated with intact forests are on par with values of huge homes made from timber. Ecological economics addresses squarely the complex trade-offs in services rendered from ecosystem assets. This is not to say that conventional economics is blind to these interrelations, or does not have the tools to address them; but they are the primary focus of ecological economics.

Biophysical constraints also have implications for human valuation of ecosystems and their services. Human values will always be a part of ecosystem valuation for ecosystem management purposes. Humans can hardly escape placing their own values on actions that require sacrifices to themselves. However, biophysical constraints may place values at odds with natural system values. Part of the valuation problem is one of full accounting for values. A tree is more than a tree; it is a tree plus a nutrient cycler, soil retainer, and hydrology and climate stabilizer. The value of a tree is its timber plus its other values. Timber markets may suggest one value, but a full ecological accounting suggests another. The economy may evaluate the use of land for pasture at twice its value for sustainable forestry. Yet the ecological value as a forest may be ten times that of the land's value in agriculture, as measured by their respective primary productivities. Which value is it most appropriate to use in deciding whether to deforest the land for agriculture? Natural system values are at odds with economic values. This issue of valuation is a controversial sticking point between ecologists and economists. However, all would agree that the issue implies that even narrowly economic valuation consider the full ecological implications, and the economic values of irreversibly altering ecosystem structures and processes. Surely, in the forest-agriculture case, narrowly economic valuations and purely ecological valuations would converge.

7.1 What Is the Ecological Economics (EE) Perspective?

Humans and their economies are parts of larger natural ecosystems and coevolve with those natural systems. There is a material and energy basis for the relations between human economies and their ecosystems. This basis defines economic as well as social structures and processes. Economies possess general ecosystem properties, such as dynamism, evolution, integrity, stability, and resilience. What makes humans and their economies unique as a sub-ecosystem is their ability, through wilful effort, ignorance, and human-designed tools, to dramatically restructure and reform processes in their ecosystems to such a degree that human welfare can be significantly diminished or enhanced. There are many factual examples (World Commission on Environment and Development 1987, Goudie 1994). Some types of economic activities, and the welfare that originates from them, would not be sustainable if they substantially adversely impact natural systems.

The wilful effort to extract useful things from natural systems is motivated by the satisfaction of basic biological needs and the seemingly limitless search for pleasure through consumption of goods. The magnitude of potential impact on their own welfare through effects on natural systems requires that human decisions be guided by some notion of the value of their actions and the value of their impacts on ecosystems, either in terms of benefits of use or costs of abuse. Some concept of value is required for rational evaluation of human economic activity within natural systems (Page 1992).

Both the structures and processes of natural systems have identifiable instrumental value to the human economy. These narrow use values my be reflected by the summation of individual values, to the extent they are private. However, natural systems also have aesthetic, moral and cultural values (Sagoff 1988). These values are more intrinsic and unmeasurable using traditional human preferences. They may not be reflected in the simple summation across social members of individual values, since they are social and not wholly private.

Valuation is made more complicated by the fact that our natural environment helps shape values through establishing social and economic relations, aesthetic standards, and culture. If so, our decisions now about the natural environment will shape future value systems, making values endogenous and, therefore, a weak guide to behavior. A way out of this dilemma may be to base valuations of natural systems on a social vision of what a society would like to see itself be (Page 1992). The value of natural systems is then based on their ability to achieve that vision. The management dilemma in implementing this valuation is to organize a method for establishing this visioning process. This requires a collaborative visioning effort. Informed, participative visioning is a critical precursor to ecosystem management under the EE perspective.

7.2 What is the Contrast Between the Prevailing Management Paradigm and that Proposed by Ecological Economics?

To understand the management implication of the EE framework, it is useful to contrast it to a characterization of the current management paradigm. The two paradigms differ on the primacy given to human economies versus natural ecosystems. The Prevailing Management Paradigm focuses on how humans can manage ecosystems for instrumental purposes of optimizing human economic wealth. This wealth is typically measured in the value of utility-enhancing things and actions. This value is frequently measured by individual "willingness to pay" or "willingness to accept" monetary compensation for gains or losses, and by summing across independent individuals. Preferences are typically taken as given and immutable, and the manipulation of natural systems for human benefit addresses those preferences. This management paradigm approaches uncertainty about natural systems by either denying or opting in favor of human economies. If not denying the uncertainty, the optimistic argument is given that natural processes are either reversible with enough time and engineering skill, or economic systems can find human-made replacements for lost ecosystem materials and services. The prevailing issue of this paradigm is "How can we use the ecosystem to more effectively enhance human wealth and welfare?"

An alternative to this paradigm is suggested if we weight more highly the fact that ecosystems are critical to social survival, we are ignorant about how ecosystems work, we are uncertain about the full potential value of natural ecosystems to the economy, and we are ignorant about preferences of future generations. EE, using what we may term an Ecological Economic Stewardship Paradigm, would ask the following management questions:

1. What does society wish to become and what does it value?
2. What is the requisite health of an ecosystem relative to that social objective?
3. What set of human economic artifacts, structures, and processes is feasible within that requisite healthy ecosystem?
4. How can we use the adaptability of human economies to assure they meet their own welfare needs as well as the needs for preservation of a healthy ecosystem?

This perspective first requires a collaborative social dialogue to establish what society would like itself to become and how it will value things. People and societies value ecosystems for many reasons, not only those reflecting economic need. Due to the interconnections among all dimensions of social action, apparently non-economic reasons may nonetheless have material economic consequences. We value ecosystems because they are necessary for life, and are places of symbolic and aesthetic inspiration. A more complete spectrum of values, when integrated in a suitable socialization process, can provide individuals and societies with both the constitutional and institutional support for long term views.

7.3 What Is Critical Knowledge Under the Ecological Economic Stewardship Paradigm?

The EE Stewardship framework requires scientific knowledge of both how natural ecosystems respond to economic activity, as well as how economic activity responds to ecosystem changes. A seemingly useful analytical construct at this boundary is a full ecological–economic, input-output matrix. Flows of material, energy, nutrients, etc., between the economic and ecological systems would be quantified and impacts of one system on the other would be established. Such a model has been well developed for the economy alone, and ecologists have established energy flow models for ecosystems. However, little progress has been made in coupling these two separate models in any meaningful practical way (Isard 1972, Daly 1968, Cumberland 1987, Costanza and Hannon 1989). A potentially useful coupling is currently being undertaken at the University of Maryland (Bockstael et al. 1995). An ecosystem model of the Patuxent, Maryland watershed has been developed, where flows of nutrients and energy flow between spatial cells. Economic land uses are predicted, with the ecosystem configuration being an input to that prediction. Land use then feeds back to the ecosystem through runoffs based on land use. The system is dynamic and can be used in a practical way to predict land use and ecosystem configuration. This modelling of the ecologic–economic interaction is useful for purposes of foreseeing implications of management decisions, and thereby valuing different options. These implications can then be used as information in collaborative decision settings.

A related example of regional ecologic–economic analysis is the study of the Baltic Sea and its surrounding agricultural, fishery, and industrial "watershed." Folke et al. (1991) emphasize the interdependence of past urban/industrial/agricultural development on environmental goods and services, as well as on ecosystem support functions. They relate the increase in industrial production and related environmental problems in the region to several factors, especially fossil fuel usage. In 1900 annual energy consumption in Baltic Europe was 9 tons per square km and 0.25 ton per person. By 1984 this had increased to 284 tons and 5 tons respectively. Industrial production has increased by 5 to 15 times since World War II, and population has increased 4 times since the mid-1900s. The cumulative effects of this activity on the Baltic Sea's food-web, in conjunction with an intense increase in fishing pressure, severely reduced the productivity of the marine ecosystem. While the catch had increased 10-fold in the last 50 years, the catch per unit effort declined to less than half that of 1955. While less than 2 percent of the surface of the Baltic was required to produce the 1900 catch, about 85 percent of this area is now required. Based on declines in fishery stocks and other toxicity problems, the gray seal population declined from 40,000 in 1940 to 1,500 currently.

7.3.1 Scale and Mix of Human Economic Activity

The appropriate "Scale" and "Mix" of human economic activity relative to the natural ecosystem are critical issues at the ecological–economic interface (Daly 1992). The carrying capacity of an ecosystem has been proposed to address appropriate scale, and has been mechanistically applied in some circumstances

(Ehrlich 1994, Hardin 1991). A single number, for example the number of humans, is meaningless since human innovation and biological evolution may interact to moderate potentially adverse welfare effects of natural system changes. Also, a level of human welfare must be specified to make the concept operational. A general index of the physical intensity of the human economy relative to the natural system would be useful. Vitousek et al.'s (1986) estimation that humans appropriate 40 percent of the net terrestrial primary production of the biosphere is striking, although we do not know what level of appropriation places the health of the natural system at risk. Recent attention has moved toward the notion that an appropriate scale of economic activities would preserve the resilience of the life-support systems on which they depend (Arrow et al. 1995). Resilience is the ability of the ecosystem to take shocks without making catastrophic changes in structure or processes. In this perspective, indicators of loss of resilience would be used to measure whether the scale or mix of economic activities is "too large."

A more micro issue at the economy–ecosystem interface is the production relation between natural systems and human or human-made capital. First, in a pure production framework, natural systems can be viewed as natural capital (Jansson et al. 1994, Bradley and Xu 1994), which is combined with economic and social capital to generate welfare. Considerable social policy energy has been expended in arguing that enhancements in natural capital reduce the need for human or human-made capital. Instances in which natural systems and human-based capital are complementary are most certainly cases where sustainability of the natural system is valuable, if not critical, to human economies in the most instrumental sense. For example, fishing boats have no value without fishing stock. On the margin, a larger fish stock increases the productivity of human and human-made capital. On the margin, labor is more productive the cleaner the air and water. These relations are the bases for the proposition that jobs and the quality of ecosystems are positively linked (Templet and Farber 1994).

7.3.2 Indicators of Sustainable Economic Health

Second, indicators of sustainable economic health are critical under EE Stewardship. Sustaining a flow of income (welfare) requires the maintenance of the source of income, which is wealth (capital). Using the analogy of natural capital, measures of sustainable economic health require the subtraction from traditional economic income an amount necessary to replace any net degradations in the quality of natural capital. The presumption is that these degradations are reversible through investments from the economic sector to the natural sector. This is a very strong assumption in circumstances where there is no replacement for degraded natural system processes or structures. Practical examples include full welfare indicators (Daly and Cobb 1989), and integrated adjustments of National Economic Accounts, or Sector Accounts (agriculture, forestry, fishery, etc.), for natural ecosytem degradation (Van Dieren 1995, Repetto et al. 1989). This accounting for loss in natural capital is applicable at any spatial scale.

Useful attempts have been made to improve the existing system of national or regional accounts by the addition of satellite accounts that connect the flows of economic products to the resource stocks supporting those flows. These stocks include traditional items such as forests and minerals as economic assets, as well as their ability to provide various environmental services; for example, soil erosion protection, soil water retention, and climate. The primary concern is the strucure and scale of economic activity and its dependence and impact on Nature. Repetto et al. (1989) is preeminent in this area. He notes that there is a "dangerous asymmetry in the way we measure, and hence, the way we think about, the value of natural resources." While we recognize that if a level of income is only maintained by drawing down the stock of capital on which it is based, one would soon have no income, natural resources have not been considered in the same fashion. However, there has been considerable progress in this area, again due largely to Repetto's efforts. While the concerns that originally motivated the development of national accounts, the need to recognize and ameliorate major fluctuations in the business cycle, are still a central concern of most governments, questions concerning the sustainability of the natural world have now assumed major importance.

As an example, a more holistic measure of Indonesia's Net Domestic Product for the period 1971 to 1984 shows that instead of achieving an apparent annual growth of about 7.1 percent, correcting for resource depletion would reduce this annual growth to only 4 percent. These adjustments only consider a few of the commodities produced; a full accounting would certainly show a larger gap. Moreover, other indicators are similarly biased. Gross versus Net Investment showed that for a number of years Net Investment was actually negative. This implies that instead of growing capital stock, consumption merely used up the principal. Some of the years that net investment grew were due to the fortunate but hardly repeatable discovery of large but exhaustible petroleum reserves.

The integrated accounts approach seeks to measure sustainable economic welfare by subtracting the loss in potential productivity of ecosystem degradation, or the cost of ecosystem remediation, from positive values of the economy's production of useful goods and services. Valuation of that natural capital loss is made from a purely anthropogenic, current or discounted future generations perspective. The full welfare indicators go further in proposing to measure a more general concept of welfare than that represented by economic consumption opportunities, including such factors as income inequities and crime rates. Non-integrated accounts include, side by side, both traditional economic accounts and some physical indicators of natural system conditions (Bradley and Xu 1994).

The concept of ecosystem health, derived largely from its long-standing use in medicine, has recently been proposed both as a an integrative standard embodying the ultimate goal of ecosystem management, and as a source of criteria to assess the effectiveness of specific options (Costanza et al 1992). Defining health operationally is not easy. Any integrative standard such as health unavoidably involves normative considerations. Health is entirely a human construct, with specific ends and purposes in mind. Costanza (1992) focused on developing an operational standard of ecosystem health as a desired endpoint of ecosystem management. The analogy between ecosystem and human health is justified by the fact that both ecosystems and humans are complex systems that achieve a functional balance among the structures and processes of which they are composed. While physicians possess a relatively well-specified model of a 'healthy' individual, as well as a compendium of diseases, their symptoms, and other diagnostic tools, no such model or compendium is yet available to practice "ecological medicine."

Hannon et al. (1993) proposed and developed a physical standard for the maximum sustainable production of an ecological system incorporating thermodynamic principles. They chose for purposes of demonstration the maximum level of plant production from ecosystems similar in geology and climate to the indigenous tall-grass prairies of central and southern Illinois. The authors proposed that the annual entropy produced by a climax tall-grass prairie is the maximum sustainable system possible, given its annual nutrient, water, temperature, and sunlight gradients and potentials. They assume that a climax plant community, as a result of evolution, has adapted to fully use the material and energetic potentials of a particular site. The energy production of such a system is assumed a maximum on the justification that if there were any unutilized low-entropy potentials, some species or community would grow until it fully exploited that potential. They calculate and compare this 'sustainable annual entropy maximum' to agricultural production systems of local Amish farmer's corn rotations and to 'modern' farmer corn-soybean rotations. Their estimates take into account the great differences between the Amish and modern techniques in the consumption of fossil fuel, fertilizer, and pesticides. They conclude that the Amish system is sustainable because its entropy production is roughly 80 percent that of a tall-grass prairie. In contrast, the modern farm system is not sustainable since it exceeds the prairie entropy production by at least 25 percent. Similar estimates for various ecosystems that comprise an economic region would be extremely useful in estimating the extent to which the economy is sustainable.

7.3.3 A Concept of Value

Third, some concept of value must be established since any human decisions are going to be based psychologically on values gained versus values lost. Values stem from moral systems. Leopold (1949) has suggested a moral system that would imply sustainability of natural systems: "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise." The implication of this moral system is that values are based on the extent to which these properties of ecosystems, including humans, are preserved and enhanced. Basic human physical and biological needs would have high value, insofar as they reflect basic human health. Beyond basic physical and biological needs of human economies, preferences can be viewed as molded by a complex of social, genetic, and natural forces. EE suggests that preferences are mutable and adaptable beyond basic needs. An economy that did not satisfy basic needs would not be healthy, adaptable, resilient, or possess any of the desirable properties of a healthy ecosystem. Such an economy would not have the ability to adjust to changes in natural systems.

Valuations of ecosystem structures and processes should include both utilitarian and other types of values, such as social, moral, and existence values. However, at a minimum they should reflect a full valuation of all utilitarian values. For example, a comprehensive paper by Costanza et al. (1989) applies a range of empirical techniques to estimate the ecological, economic, and other social values of Louisiana's coastal marshes. In a real sense, it exemplifies precisely the kind of integrated or cross-disciplinary approach that an EE requires. They apply sound principles of ecology, economics, and political science to estimate a lower bound and bracket for wetland values. They applied two different techniques, Willingness to Pay (WTP) and Energy Analysis (EA). WTP techniques were applied to four categories of wetland benefits: commercial fishing and trapping, recreation, and storm protection. The first two categories required estimates of the per acre marginal productivity of the wetlands for shrimp, menhaden, oysters, blue crab, and muskrat and nutria furs. Estimating this marginal product required separating the effect of human effort from the effect of the wetland's intrinsic productivity. An extensive canvas of fisheries and trapping data and aquatic ecosystem science for the entire Gulf Coast was required. A travel cost method using questionnaires was their primary technique for establishing recreational values. Recreational fisherman, boaters, hunters, and photographers were questioned over a one-year period to estimate individual household WTP. These estimates were then combined with an independent survey of the total recreational saltwater fishing population in order to calculate a total recreational WTP. Hurricane protection values were estimated by relating expected annual storm damage to distance from the shore. The authors assumed that people would be WTP for the estimated reductions in damage. Estimated rates of shoreline recession with and without wetland protection efforts were also required.

EA compared the biological productivity of the wetlands versus adjacent coastal waters to measure their total contributory value. Primary plant production, the basis of the food chain, was converted to an annual economic value in terms of the equivalent fossil fuel energy costs to replace this natural plant production. These annual values, assuming that such values would materialize over an infinite series of annual payments, were then discounted to the present using 3 and 8 percent. The controversies surrounding the matter of discounting were also given considerable discussion. The WTP approach estimated the total per acre present net economic values of Louisiana's wetlands between $2,400 and $9,000 per acre. Their conservative estimates using EA were between $6,400 and $17,000 per acre.

Establishing requisite economic adaptations for sustainability of natural systems, identifying basic human needs, and understanding how preferences can be reshaped are critical research issues necessary for managing sustainable economic and natural systems. Furthermore, aggregations of individual values may be less important in valuing ecosystems than the value that society as a whole places on them, particularly when value is relative to moral codes and to what a society wishes itself to become. Studies of the divergence between the aggregate of independent, individual valuations and joint, socially based valuations, where these individuals set a consensual value in some social decision setting, are necessary before valuing large ecosystems.

National Forest Planning might benefit from such entropic calculations. Current planning processes construct and compare several alternatives for the use of the forest over a future planning horizon. Timber harvests, grazing, and other consumptive uses are key considerations. Without some idea of what constitutes a maximum sustainable ecosystem production, planners must fall back on historical use levels. Some forest users have a huge stake in maintaining or even expanding their share of forest outputs. Other groups believe that current usage exceeds potentials.

Valuations of ecosystem services have typically been from the perspective of current generations and propose that values reflect individual preferences, as represented by the willingness to pay for these services. Extensive valuation methods have been developed by environmental economists (Freeman 1993) and have been applied to large ecosystems (Farber 1995). However, these valuation procedures may not be appropriate to valuing such services in a sustainability context. In a sustainability context, ecosystem structure and functions would be evaluated on the basis of the extent to which they contribute to the goal of economic and ecosystem health and sustainability, rather than on the basis of their immediate contribution to current economic welfare.

Valuation of ecosystems based on individual preferences can be useful where spatial scales are narrow, temporal scales are short, and values are "on the margin." However, the dramatic and potentially most serious ecosystem issues, such as global warming, are non-marginal, large spatial and temporal scale problems. Preference-based valuations appear shallow in this context. Preference based valuations are further complicated by the time-dependence of benefits from ecosystems. Traditional discounting is preference based. One justification is based on extrapolating the presumption that a unit of something is worth more to an individual today than tomorrow, to the presumption that this would also be true if it were different individuals at different points in time. To avoid this individualistic presumption, economists have suggested using rates of social time preference, which reflect how much an existing society would discount the same society's benefits in the future. The problem with even this social concept is that it places the members of the present society in a position of dictating the legacy to be passed to the future, with the weighting of future generation welfare less than the current generation. Arguments are made that discounting is appropriate because investments will be made in the present that will provide a legacy of increased productive capacity to the future, or that the future will be better off than the present. Neither of these may be the case; and if economic decisions result in irreversible destruction of ecosystem capital, they will likely not be the case.

A discounting procedure consistent with sustainability goals could be as follows. In making decisions over the management of ecosystems, those changes that would enhance or degrade the human life support capacity of the ecosystem, or that would degrade the health, integrity and resilience, would have a zero discount rate. Those ecosystem changes that impacted welfare above the threshold basic needs level would be discounted, but at the social rate of discount (Mikesell 1977).

A proposed, purely ecological valuation designed to avoid preferences altogether would value ecosytem structures and processes solely by their capacity to transform energy or matter; hence an "energy-based" valuation (Costanza 1980, Costanza et al. 1989). This valuation is extreme in placing a zero weight on human preferences, and may be too sterile to be attractive for ecosystem management, although it is consistent with measuring ecosystem value relative to the goal of preserving ecosystem processes.

7.3.4 Understanding Human Economic Adaptability

Fourth, the EE Stewardship focus requires understanding of human economic adaptability. This includes adaptability of preferences to new circumstances, noted above. In addition, this requires knowledge of tradeoffs that the human economy has available to meet human needs and wants. Knowledge about preference formation, and the speed and costs of adjustment to changes in markets for economic goods and services, is important to understand how the economy can adapt to changes in ecosystem structure and processes.

7.3.5 Institutions for sustainability

Fifth, EE Stewardship requires the use of property rights systems, laws, and institutions that are incentive compatible with sustainability norms. All economically driven incentive systems that have adverse consequences for ecosystem health, and existing institutional impediments to economic adaptability, such as farm subsidy programs and land tenure systems, have to be illuminated to portray their full ecological–economic impact. We are developing increasing knowledge about these perverse incentive systems and institutional barriers to sustaining ecosystem health (Farber 1991).

7.4 What Are the Bases for Decisions Under Uncertainty With Ecological Economic Stewardship?

We can distinguish between risk and uncertainty. Classic risk presumes that we know some probability distribution associated with events and states of the world. The concepts of expected value or most likely states are definable. Classic uncertainty presumes there is no knowledge of probabilities. As noted above, the prevailing management paradigm approaches uncertainty about natural systems by either denying it, proclaiming there will be remediable options, or opting in favor of human economies. If not denying the uncertainty, the optimistic argument is given that natural processes are either reversible with enough time and engineering skill, or economic systems can find human-made replacements for lost ecosystem materials and services. Under EE Stewardship, there is a very high cost associated with being wrong about reversibility, remediation, and mitigation of degradations in natural system health. A precautionary (Perrings 1991) or minimum regrets approach to decisions that may adversely impact natural system would opt in favor of ecosystem health protection. The cost of this decision rule may not be so high, particularly if basic human needs are not at stake and human preferences and economic structures are adaptable.

Additional management decision criteria would include estimation of impacts under worst case scenarios. The values of benefits lost would be a maximum under these scenarios. When benefits are not known or unquantifiable, a decision criterion is to consider the costs associated with preserving those benefits. Finding that timber profits are relatively low when considering a cut that is likely to have serious ecological impacts would suggest the benefits lost from denying the cut would be minimal.

8 CONCLUSIONS

Ecosystems are assets, or natural capital, which yield services to the human economy. Ecosytem management is a necessity due to the failures of private ownership and ecosytem use to arrive at decisions for uses of these assets that are appropriate for society at large; i.e., they fail to achieve the "highest and best" uses, most broadly interpreted, of ecosystem assets. These failures reside in the facts that property rights are never complete when it comes to all ecosystem services, ecosystem use is replete with spillovers and externalities, ecosystem services are frequently like public goods and subject to the availability for a wide array of persons and interest groups, costs of privately negotiated, voluntary settlements between stakeholders are often prohibitive, immobilities and nonadaptabilities of human resource use impede the most appropriate and highest valued uses, people may be unaware of all ecosytem values, and government interventions in seemingly unrelated areas (such as support prices) may cause inappropriate private decisions for some ecosystem assets.

Given an objective of "highest and best" use of ecosystem assets, most broadly interpreted to include full contemporaneous and intertemporal uses, there must be some type of valuation system for making the inevitable trade-off decisions in ecosystem management. Traditional economics offers a wide, potentially useful array of valuation methods, all directed toward determining monetary values for various uses. These valuation procedures seek measures of what individual members of society would be "willing to pay" to have more ecosystem services of a certain type, such as recreation, or what they would be "willing to accept" in compensation for denial of these services. An entire array of values, from direct use of a resource, such as timber harvest, to non-use, such as cultural values, are absolutely necessary for establishing a complete picture of ecosystem service values. These methods include both observed and hypothetical techniques. These methods do provide meaningful clues as to relative values of different ecosystem uses. However, warnings must be made that monetary valuations are not the only types of values. And individualistic valuations may differ from a more socially oriented valuation. Nevertheless, these traditional economic valuation methods help paint a picture of value that is certainly superior to simply counting trees logged or livestock grazed.

A new interdisciplinary field, Ecological Economics, has developed to address a major deficiency in traditional economics: the absence of due attention to biophysical processes in managing human economies and ecosystems. It points out that there are physical laws that constrain economic possibilities, particularly unconditional, unlimited economic growth. Economies are dependent on the ecosystems in which they are embedded. Ecosystem and economic management models must shift thinking toward maintaining natural capital and its flow of services in order to sustain even contemporaneous levels of economic activity. Full system management includes both managing ecosystems and economies. Preserving the health and integrity of ecosystems is a dominant goal emerging from this field, and addressing the adaptability of the human economy to that constraint is the emerging policy issue.

Ecological economics has sought to elevate understanding of ecosystem complexities in making ecosystem use decisions. It has sought to enroll ecologists and other physical scientists, along with economists, in understanding how natural and human systems interact. Philosophers have contributed their analyses of values to the issue of how trade-offs in uses can be evaluated; narrow monetary, individualistic willingness-based values may not be fully adequate to the task of addressing the complex trade-offs in ecosystems. An implication is that social-based valuations, such as those arrived at by visioning and meetings of stakeholders, may be superior to these individualistic, narrowly economic, valuations.

Ecosystem management should be a complex task. It involves not only managing complex natural capital systems, but it should also involve managing complex human economies. Certainly, full valuation of uses, both direct and indirect, must be evaluated in gauging the trade-offs involved in decisions. In addition to asking about the adaptability of the ecosystem to changing uses, the ecosystem manager must ask about the adaptability of the human economies connected to and dependent on these natural systems. We hope this chapter will help managers to frame problems in a more comprehensive way, and provided them with a glimpse of potentially useful tools in making their difficult decisions.

9. ADDITIONAL SOURCES OF INFORMATION

Relevant Professional Societies

International Society for Ecological Economics
PO Box 38, Solomons, MD 20688
(410) 326-0794; http://kabir.cbl.cees.edu/ISEE/ISEE home. html

Association of Environmental and Resource Economists, 1616 P Street, NW, Room 507
Washington, DC 20036.

Relevant Professional Journals

Ecological Economics
Journal of Environmental Economics and Management
Land Economics

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