# FUNDAMENTALS OF ECONOMIC PRINCIPLES AND WILDLIFE MANAGEMENT

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***Abstract:*** This paper presents an overview of the economic fundamentals involved in wildlife management, with special consideration for cases involving harmful wildlife-human interactions. The process of benefit-cost analysis is used as a unifying platform for incorporating both theoretical and empirical issues. Topics such as external market effects and public goods are detailed in order to give the reader a theoretical foundation for understanding the economic perspective on the problems associated with defining and attaining optimally managed wildlife populations. To these principles we add practical considerations for measuring the costs and benefits associated with wildlife populations. Different categories of wildlife values, such as use and nonuse values, and alternative methodologies for their measurement are described. The paper concludes with a discussion of areas for improving data collection and value estimation so that the goals and perspectives of economists and wildlife managers can be further integrated.

***Key words:*** benefit cost analysis, economic value, wildlife management.

The purpose of this paper is to discuss, in general terms, an economic approach to addressing the problems associated with human-wildlife conflicts. We say *general* in that many of the remaining articles explore the specifics of the problems and factors that we introduce. It should be noted that, from an economic perspective, human-wildlife conflicts are just 1 part of a larger problem, wildlife management. Thus, we will broaden our focus to include the optimal management of wildlife, of which human-wildlife conflicts are an integral part. Indeed, a somewhat naïve and trivial solution to minimizing the conflicts surrounding human-wildlife interaction would be to reduce the amount of wildlife. Yet, in addition to the valuable roles wildlife plays in protecting and enhancing ecosystems, people value wildlife for both consumptive and nonconsumptive uses. While some of these values are captured in markets, others are not.

An example of consumptive and nonconsumptive values from wildlife is found in Loomis et al. (1989a). They estimate the value of California deer for hunting purposes to be approximately US$230 million and for viewing purposes roughly US$34.5 million. Moreover, the U.S. Fish and Wildlife Service suggests that more than 76 million Americans engage in "nonconsumptive practices" such as viewing or photographing wildlife (Ohio Division of Wildlife (ODNR 1998)). Hence, any discussion of resolving the problems associated with human-wildlife conflict must include the potential trade-offs associated with the proposed solutions that are likely to impact the value these resources generate. Furthermore, such a discussion must acknowledge that the interaction between human and market systems and the natural environment flows in both directions (Fig. 1).

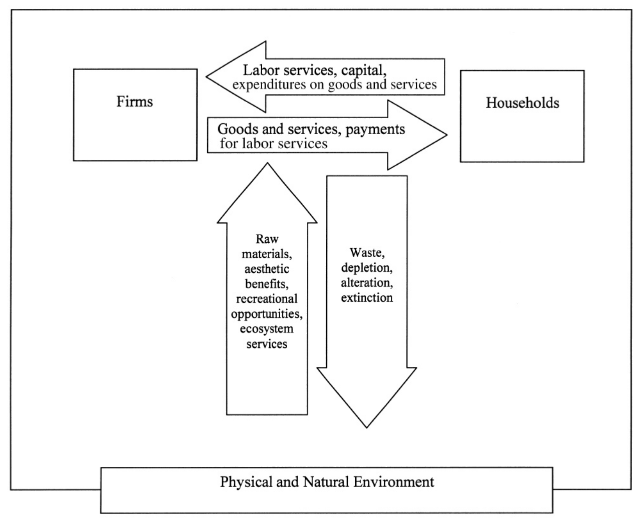


**Fig. 1. Interaction between markets and the natural environment. Adapted from Kahn (1998) and Baumol and Blinder (1997).**

One method that helps inform us of the relative trade-offs of alternative wildlife management strategies is Benefit-Cost Analysis. Benefit-Costs Analysis (BCA) provides for a systematic enumeration of the gains (benefits) and losses (costs) of particular decisions, in common units, for comparison purposes. Reed et al. (1982) provide 1 of the few BCAs of human-wildlife conflict in their research on reducing deer-vehicle collisions (DVCs). In their analysis, the benefits of reducing DVCs are associated with lower deer mortality and vehicle damage, while the costs of reducing these events include fence installation and maintenance. Conceptually, Reed et al. account for 2 of the 3 economic aspects that Keith and Lyon (1985) outline as necessary requirements of public wildlife management decisions, including: (1) "…value to users of increases or decreases in wildlife populations," and (2) "the costs of providing increments of wildlife populations through habitat manipulations and/or other management alternatives." Keith and Lyon also suggest such analyses account for the relationships between current and future wildlife populations.

The importance of accounting for both the temporal and spatial dimensions associated with wildlife populations and management decisions are critical. As Tietenberg (1997) notes, wildlife populations, like other resource stocks, are viewed by economists as composite assets that provide benefits to humans over time, and as with other assets, we wish to optimize their value. From an economic perspective, then, the primary goal of wildlife management is to find the population size that maximizes the net benefits from the resource, where net benefits are defined as the difference between total costs and total benefits to all affected parties of using the resource.[1] Once the optimal population size has been determined, a secondary goal is to evaluate alternative measures for achieving this size. Together, these actions synthesize 2 general strands of research in the economics of wildlife management: works that focus on estimating the value of additional wildlife resources (e.g., Hammack and Brown 1974, Keith and Lyon 1985, Loomis et al. 1989a, Bockstael and McConnell 1999) and works that estimate costs that wildlife impose on society (e.g., Reed et al. 1982, Conover 1994, Romin and Bissonette 1996a).

The next section will highlight the economic consequences associated with human-wildlife conflict by presenting some general cost and damages estimates. This section will also include a discussion of some of the difficulties associated with managing wildlife resources, including problems associated with externalities, open-access, and public goods. The third section will discuss the conceptual aspects of using net benefits as the criterion to judge the economic desirability of alternative management schemes. How we define benefits, what counts as a cost, and the importance of including the time element are all addressed. The fourth section presents some empirical issues and estimates for valuing and costing wildlife resources. In particular, several environmental resource valuation techniques designed to capture the nonmarket value of wildlife resources are discussed. The last section concludes with a discussion of potential policies for achieving the desired objectives and calls for coordinated efforts between researchers within different disciplines and both local and state wildlife agencies.

Before discussing the costs and damages associated with human-wildlife conflicts, it is important to acknowledge a few assumptions we, as economists, make. First, "value" is defined in an anthropocentric context. That is, a resource has value only to the extent that people care about it. This is in contrast to defining value in a biocentric context, which basically gives all creatures in the ecosystem equal standing (Loomis 1993). Second, the value of a resource, even in economics terms, can include both market and nonmarket activities, as well as both use and nonuse values.[2] Finally, while the term "human-wildlife conflict" includes both the impact of wildlife on humans and the impact of humans on wildlife, our primary focus is on the impact of wildlife on humans. It should be noted, though, that this approach does not exclude the value that people place on damages to wildlife. Indeed, such aesthetic or moral values do fit within the anthropocentric realm.

## COSTS, DAMAGES, AND MARKET FAILURE IN THE MANAGEMENT OF WILDLIFE

Only a cursory familiarization with the literature on human-wildlife interaction is needed to appreciate what these interactions cost society (e.g., Conover 1994, Conover et al. 1995, Cook and Daggett 1995, Romin and Bissonette 1996). One of the most complete evaluations of the costs wildlife impose on the United States was by Conover et al. (1995). In their summary of wildlife-human conflicts nationwide (including human illness and fatalities from wildlife-related diseases, bites or attacks; animal-vehicle collisions; and wildlife damage to agricultural production, households, and timber production) the authors estimated that roughly 415 deaths and 75,000 injuries or illnesses occurred each year from wildlife-related disease, attacks, and collisions. Overall, Conover et al. suggested that the total economic loss from wildlife-related damages was approximately US$3 billion annually.[3] To put this figure in perspective, the annual budget of the federal government for conservation and land management programs is less than US$5 billion (U.S. Bureau of the Census 1996).

One of the largest components of these damages is deer-vehicle collisions. The National Highway Traffic Safety Administration suggested that 120 human deaths occurred from DVCs in 1990 and estimated that the national cost in motorist loss of life and injury is nearly US$200 million annually. In terms of lost wildlife, Romin and Bissonette (1996) estimated that more than 500,000 deer are killed each year in DVCs. Cook and Daggett (1995) estimated that vehicle collisions account for 500 moose deaths in Alaska every year, and over a 5-year period more than 200 black bear deaths occurred from animal-vehicle collisions in Florida and Pennsylva-

[1] We note that the parties that realize the benefits from a given wildlife population or wildlife policy may be different than those individuals that bear its costs. Further, both benefits and costs may be distributed temporally and spatially.

[2] "Use" values include those values associated with the tangible uses of the resources while "nonuse" values account for the intangible uses, such as leaving the resource for future generations. Further discussion of these concepts is provided in a later section.

[3] From a regulatory perspective, the United States Department of Agriculture Wildlife Services Program spent more than US$26 million in 1988 on efforts to reduce the damages from wildlife and spent an additional US$11 million on administration-related expenditures (Rollins and Briggs 1996).

nia. Endangered species are also lost in animal-vehicle collisions. For example, since 1979 more than 54% of the endangered Florida panther population and 65 of the 300 endangered Florida Key deer have been killed via collisions with vehicles (Cook and Daggett 1995). Finally, more than 20 million small animals (e.g., rabbits, badgers, reptiles, dogs, cat, and birds) are killed each year due to collisions with vehicles, including aircraft (Cook and Daggett 1995).

In addition to the costs associated with collisions with vehicles, wildlife-related crop damages are a substantial component of the overall costs of human-wildlife conflict. Estimates of annual wildlife-related crop damage in the U.S. range from US$464 million in 1994 (Conover et. al. 1995) to US$533 million in 1989 (Wywialowski 1994). At the state-level, Forster and Hitzhusen (1997) estimated that Ohio farmers lost approximately US$46 million from wildlife-related crop damage.

While these costs and damages may seem excessive and beg for immediate management intervention, optimal management strategies must consider both the costs and benefits of wildlife resources. Indeed, many states explicitly express their objectives of managing wildlife populations with consideration of both the benefits and costs. For example, in Ohio the goal for deer management includes maximizing the recreational opportunities such as hunting, viewing, and photographing within the context of minimizing conflicts with agriculture, motor travel, and other areas of human endeavor (Ohio Department of Natural Resources 1998). Similarly, in Wisconsin the deer management policy states that, "…regulations shall be designed to maintain a herd in balance with its range and at population levels reasonably compatible with agricultural and forest management objectives…"(Creed et al. 1984). These principles capture the idea of optimal management strategy, yet in practice 3 potential problems arise that can inhibit local or state officials from managing these resources optimally. These include externalities, open access externalities, and public goods.

### Externalities and Open-Access

Many states manage wildlife resources in terms of geographical areas. For instance, in Wisconsin deer resources are managed in "management units," with 96 such units in the state and each unit averaging approximately 1,500 km$^2$. Similarly, Ohio manages its wildlife populations at the county level. One problem with this management scheme is that wildlife resources are not physically confined to either management units or county boundaries. As such, actions officials take at the management unit level are likely to impose costs (or benefits) on the adjacent areas. In essence, the management authorities in 1 unit will base their optimal population targets on the benefits and costs that they incur but will not account for or "internalize" any residual effects these actions impose on surrounding areas. These residual effects often translate into costs and benefits imposed on others. When decisions by 1 county or management unit are made without consideration of the external costs or benefits these decisions may impose on others, each county or management unit may achieve their economically efficient population levels, but the state as a whole may be operating at an inefficient level. This suggests that some intervention at the state level should coordinate population targets across inter-county or management units. Clearly, the distributional effects of policy on overall efficiency should play a role in policy design.

Another problem with managing wildlife resources is that many of these resources reside on private lands. As such, decisions by the private landowner with respect to maintaining suitable habitats for these resources may not be optimal for society. In other words, the private landowner's actions impose costs on society.[4] For example, Rollins and Briggs (1996) discussed private decision-making by agricultural landowners near Horicon Marsh in Wisconsin regarding whether to passively provide forage for Canada geese (*Branta Canadensis*) as the geese migrate across their land. These geese provide large recreational benefits to hunters (Bishop and Heberlein 1979), yet the costs accrue to agricultural producers via foregone production. Hence, private decisions by landowners around Horicon Marsh may not acknowledge the benefits these geese provide to Wisconsin hunters. From society's perspective, the benefits of having the geese available for hunting may outweigh the costs the geese impose on agriculture. Hence, when the incidences of costs and benefits from a particular resource management decision differ, the solution attained through private decision-making is likely to be inefficient. We can conclude that the problems associated with both fugitive resources and externalities complicate the management process, and suggest that efforts to achieve the socially optimal population size require a higher level of government involvement.

As the term implies, a resource that is considered open-access, meaning it is open to uncontrolled access by potential users of the resource (Field 1997), often leads to users imposing external costs on other users and nonusers of the resource. A classic example includes a fishery where the anglers continue to fish without considering the impacts of their actions on others, i.e., less fish for other anglers. In effect, individuals engage in their own private-decision making process without regard to the external costs they impose on others. With an open-access resource, the costs are in

[4] Alternatively, decisions by the federal, state, or local governments with respect to resource usage could impose costs on private landowners. For a detailed discussion how of public agencies impose costs on private landowners with respect to wildlife management, see Lueck (1995).

the form of a reduction in resources, resource usage, or resource quality for others. Finally, since the resource is open-access, 1 user cannot restrict another user from the resource. From an economic perspective, the market alone will not utilize the resource efficiently and thus some level of government intervention is required.

### Public Goods

Another difficulty with managing resources at the local level is that many of these resources have public good characteristics. A public good is a good that is both nonrivial and nonexcludable. That is, 1 person's consumption does not diminish the amount left over for others to consumer (nonrivial) nor does 1 person's consumption inhibit others from consuming the good (nonexcludable). Thus, once a public good such as wildlife services are provided to 1 unit or locality, another unit can consume them without cost.

While at first glance public goods may not seem like a problem, consider the case of endangered species that reside in such a unit or county. The benefits the species provide to society may extend far beyond the borders of the unit or county. Indeed, such benefits can conceivably extend over political and national boundaries and across generations. Yet, the costs of maintaining these resources are likely to fall upon the authority under which the resource resides. While it is likely that the value to society is much greater than the costs of maintaining the resource, appropriating the required funds for maintaining the resource from members of society that value it may be difficult. This difficulty arises because once the resource is provided to 1, everyone can "use" it regardless of his or her part in helping to provide the resource. This phenomenon, known as "free-riding", takes place because of the assumption that other members will provide the resource. If enough members engage in this "free-riding," the funds required to provide the resource may be inadequate and thus the resource may be under-provided. While this problem does provide a role for additional government involvement, such as public financing for the good, government action may not resolve the problem fully. As Loomis (1993) pointed out, the provision of additional public goods still involves opportunity costs, costs that may consist of tax monies that would otherwise provide alternative services that people value. As such, these are additional trade-offs whose costs and benefits need to be evaluated to determine the overall impact on net benefits.

## CONCEPTUAL ISSUES ASSOCIATED WITH NET BENEFITS: BENEFITS, COSTS, AND TIME

The attractiveness of BCA is that it can inform policy makers of the benefits and costs of alternative resource uses to society and provide a criterion – net benefits – by which to judge alternatives. We begin this section with a definition of benefits and costs, and then discuss some issues involved with combining these 2 factors into an estimate of net benefits. We conclude with a discussion of the dynamic nature of wildlife resources and the importance of acknowledging the distribution of costs and benefits across time.

### Benefits

Economists use the term "benefits" to mean the dollar value of the satisfaction obtained from the use of a good or service. It is important to identify 2 characteristics of the term "value" in economics. As noted in Field (1997) and mentioned above, value is defined from an anthropocentric context and is only meaningful relative to what people are willing and able to give up for the good or service. If people are not willing to pay or trade something to obtain a particular good, we say that good has zero value. Alternatively, if someone was willing and able to give up some amount of 1 good for another good, say a pair of tennis shoes for a hunting license, we would say that this person values the hunting license at least as much as a pair of tennis shoes.

While there may be many combinations of goods people would be willing and able to give up for any other particular good, what is useful and convenient for comparison and aggregation purposes is to define the value of a particular good to an individual as the most this individual would be willing and able to give up to obtain the good (Fig. 2). Furthermore, and again for comparison and aggregation purposes, it would be convenient to measure the trade-offs with a common metric. Economists have therefore defined the value of a good to any particular individual as the most this individual would be willing and able to pay for the good (Fig. 2). Going back to benefits, then, we see that the benefits one derives from a good or service can be measured by the (greatest) amount of income an individual would be willing (and able) to give up in order to consume the good. While income is used as a measure of the amount by which an individual is made

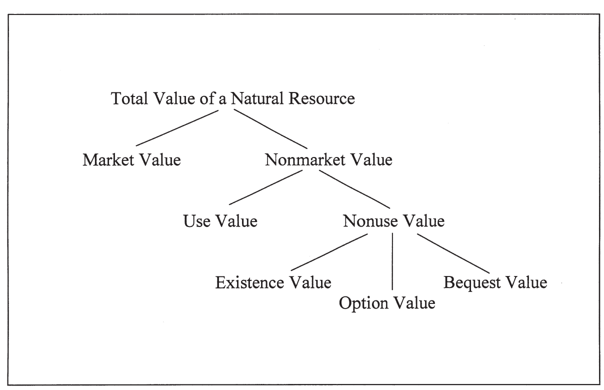


**Fig. 2. The components of value.**

better off from using the resource, it should be noted that there is nothing inherently attractive about income. Rather, income serves as a proxy for the other bundles of goods or services this income could have otherwise purchased.

These concepts may seem relatively straightforward, yet there are some common misunderstandings and misconceptions with how economists use the term value, and ultimately then, benefits. First, value is not measured by what you actually have to pay for a good, but rather by what you would be willing to pay for a good. Along these same lines, cost is not an accurate representation of value. One can easily imagine a service that costs an enormous amount, say importing sand from the Middle East to Colorado, yet is valued quite low.

As far as misconceptions, economic value is often thought of as only pertaining to those goods that are traded in the market place, i.e., market goods have value as observed by people willing to buy and sell them. Yet, goods that are not traded in the market (nonmarket goods such as clean air, clean water, or the sunset at the Grand Canyon) have economic value as well. In these latter cases, rather than obtaining these goods via trades in the market place, people are willing to give up time or other resources (including money) for the opportunity to consume them.

Finally, the economic value of a nonmarket good entails both use and nonuse values. The more commonly acknowledged "use" value consists of the tangible components of the good, such as actually traveling to the Grand Canyon to see the sunsets, view the wildlife, or hike the trails. The less commonly known "nonuse" values pertain to the intangible or indirect uses of a good. Such intangible uses, which will be discussed in more detail in the next section, include saving the good for use at some other time (option value), saving the good for future generations (bequest value), saving the good for others to use now (altruistic value), and simply saving the good for the mere sake of its existence (existence value).

One of the guiding principles of economics is that the additional benefits or satisfaction derived from subsequent units of a given resource typically decrease, i.e., benefits are decreasing at the margin. For example, the benefit realized by a hunter for the first unit of game harvested will likely exceed the benefits from the second unit harvested, and so on.[5] As the marginal benefit is the dollar value of the satisfaction derived from an additional unit of the resource, we can state that marginal benefit is a measure of the maximum willingness to pay for that unit. Hence we can assume that a hunter's willingness to pay for additional units of

[5] It is important to recognize that this assumes that the quality of a unit is held constant.

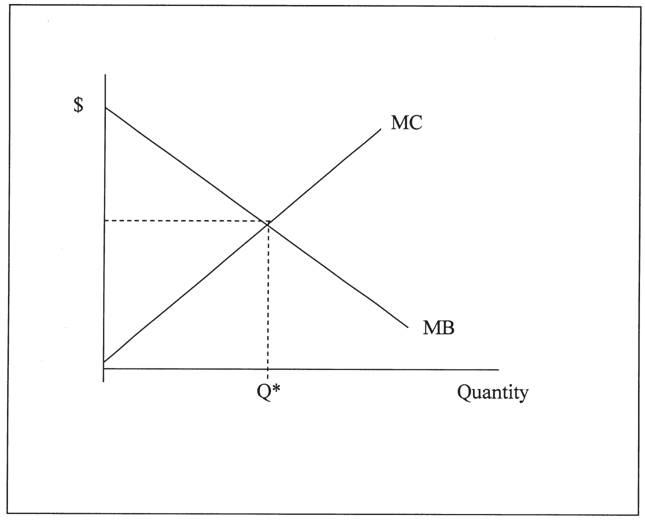


**Fig. 3. Marginal costs and marginal benefits.**

the resource will be decreasing in the quantity of the resource consumed. This relationship is illustrated in Fig. 1 by the marginal benefit curve (MB).[6]

The marginal benefit curve (MB) is synonymous with the demand curve used in economic principles, and provides several pieces of information regarding the preferences of individuals for a particular good or service.[7] Starting with a particular quantity on the horizontal axis, the MB curve reveals the marginal benefit or willingness-to-pay for that unit (Fig. 3). This value is the highest price the individual would pay for that additional unit. Starting with a dollar value on the vertical axis, the MB curve also reveals the greatest quantity that will be purchased at that dollar value or price. The area under the MB curve at any quantity is the total willingness to pay for that consumption level. This value, which is also referred to as the benefits, represents the amount a person would be willing to pay to attain that level of the good or service rather than go without it entirely (Field 1997).

## Costs

The cost of achieving a particular objective can be thought of as what is given up – i.e., inputs – to obtain the objective. Typically, we use the term opportunity costs – a term that measures the dollar value of the inputs in their next best alternative. As noted in Boardman et al. (1996), opportunity costs measure the value of what society must forego to achieve any particular objective. The reason we use the term opportu-

[6] While we present the MB function as a straight line that need not be the case. It is often represented by a negatively sloped curve.

[7] The MB curve for an individual reveals his or her preferences for the good or service and is a function of his or her willingness and ability to pay for that good. The summation of a number of individual demand curves (known as an aggregate demand curve) reveals the marginal benefit for a group of individuals.

nity costs is that by using the resources for 1 activity, we give up the opportunity of using these same resources in another activity. For example, in erecting fences along a highway to reduce the incidences of DVCs, the costs would include the capital and labor required to install and maintain the fencing. There may also be some administrative costs required to coordinate the installation and maintenance.

While benefits are modeled as decreasing at the margin, economists typically model costs as increasing at the margin. This relationship holds for both costs of production and costs in terms of foregone benefits from natural resources.[8] Increasing costs of production follows from the law of diminishing marginal productivity (e.g., Hyman 1996:199-203) which states that given some fixed factors of production the additional opportunity costs of production increase as more of that particular product is produced. The increasing foregone benefits suggest that, as more of a resource is used up today, the value of remaining units of the resource will increase. For example, as more and more deer are depleted via hunting thus leaving fewer for the remaining part of the season, the value of those remaining deer will likely increase (Fig. 1).

In the context of market goods and services, the marginal cost curve (MC) represents the supply relationship for the good. In this sense, this curve reveals the quantity of the good that firms would be willing and able to bring to market at various prices. This interpretation is not as appropriate for environmental goods and services. However, we can represent the cost of providing different levels of environmental goods with an upward sloping function. Again, the marginal cost of a particular quantity unit is found using the corresponding value on the vertical axis, and the area under the MC curve at that quantity represents the total cost of providing that quantity, barring any fixed costs.

### Net Benefits

The net benefits of a particular good or service are simply the difference between the value society places on that good or service, i.e., the most they would be willing to pay, less the resources they must forego to obtain it, i.e., the opportunity costs. For example, suppose a hunter values the opportunity to hunt deer during a particular season at US$100. That is, the most they would be willing to pay, i.e., the benefits, for hunting deer is US$100. Furthermore, let us suppose that in order to hunt deer, the hunter must purchase both a hunting license for US$20 and a deer permit for US$15, for a total cost of US$35. The net benefits of hunting deer to this particular hunter, then, is US$65.

Now the question arises as to what is lost if this hunter is not allowed to hunt deer. Obviously, the hunter will not garner the benefits of hunting the deer, which is measured at US$100. Yet, this is not what is lost, since the hunter would have had to forgo the US$35 to hunt the deer. That is, if he cannot hunt deer, he still has the US$35 to spend in some other manner. What are lost, then, are the net benefits that the hunter would have derived from engaging in the hunting experience, i.e., US$65. The loss to society of not allowing this hunter to hunt deer is the additional benefit above and beyond the costs required to hunt deer. This term, labeled above as net benefits, is commonly referred to as "consumer surplus," and is used to gauge the gain or loss to society from a particular action.

Focusing on Fig. 3 the net benefits (benefits less costs) of a given quantity is the area below the MB curve and above the MC curve. Economic efficiency in provision of the good or service is defined as the quantity of the environmental service that maximizes the total net benefits realized by society. Achieving this optimal quantity requires that both the marginal benefit and marginal cost relationships be measured and examined together. Economic efficiency is achieved by balancing the costs of an additional unit with the benefits of an additional unit. Because of the shapes of these 2 functions, by producing and consuming up to the point where the marginal benefit of another unit is equal to the marginal cost (MB = MC), an efficient solution is achieved and net benefits are maximized.

Q* shows the quantity that maximizes net benefits. At Q* the MB and MC curves intersect – i.e., MB = MC. To understand why net benefits are maximized at this point, consider producing either 1 more or 1 less unit. If society produces 1 more unit (Q* +1), the marginal cost of that additional unit is greater than the marginal benefit of that additional unit. Therefore, production and consumption of that unit causes net benefits to decrease. Alternatively, if production decreases by 1 unit (Q* -1), the forgone benefits are greater than the costs saved. Again, this would lead to a reduction in the overall net benefits. Hence, movements away from Q*, whether to the left or the right, necessarily lead to fewer net benefits and are therefore deemed suboptimal or inefficient.

### Net Benefits, Populations, and Time

While it may seem straightforward to calculate the benefits, the costs, and then the net benefits of a particular action, 2 factors arise in managing wildlife resources that can add further complexity to the issue. First, since wildlife populations are essentially renewable resources, actions to control populations today will have implications on the future availability of the population. Second, it is often the case that benefits and

[8] This principle follows from the scarcity of resources – the fewer units of a good that are available, the higher the marginal value of each unit. This cost is often referred to as the "user cost."

costs of wildlife management activities are incurred in different time periods (or by different individuals in the same time period).

A consideration of the full costs and benefits of a wildlife population requires that the dynamic nature of the resource be recognized, modeled, and directly linked to economic valuation. Empirical examples of so-called "bioeconomic models" are fairly uncommon in the literature because they require a synthesis of knowledge and methods from multiple disciplines.[9] To illustrate the importance of these different views on resource management, we present a well-known treatment of the problem of renewable resource management.

Assuming that a resource stock or population in question is characterized by a logistic growth pattern, growth of the population will be increasing in population up to some size, after which population growth is positive but decreasing until the stock reaches some maximum possible size (the carrying capacity of the environment). A given level of harvest (measured in the same units as growth) is said to be a "sustainable yield" when it is equal to the growth rate of the population. That is, when the rate of removal is equal to the rate of natural growth, a situation develops where the rate of removal is sustainable indefinitely, at least from a theoretical perspective. The population size that allows for the highest rate of growth is termed the maximum sustainable yield (MSY).

Managing a population at the maximum sustainable yield, though, may not be efficient from an economic perspective. Merely examining yield (the number of units of the resource that are harvested per time period) only reveals the benefits derived from a given population size. For efficiency, both costs and benefits must be considered to achieve an efficient sustainable yield (ESY). The ESY is defined by the harvest rate that, when maintained perpetually, produces the largest amount of net benefits. The 2 types of costs that are generally considered in such dynamic models include the costs of harvest efforts and the costs imposed on individuals outside the market (i.e., the "external costs" such as DVCs and crop damage). Because the costs of harvest efforts are likely decreasing in the size of the resource population while the external costs are likely increasing in population size, the shape of the cost function for a given resource is an empirical question. We can state, however, that the efficient sustainable yield is unlikely to occur at the same population size as the maximum sustainable yield.

Another potential complication associated with estimating the net benefits of a particular action is that some costs and benefits may be realized immediately while others may be realized in the future.[10] Indeed, most wildlife policy changes are likely to lead to costs and benefits that may be distributed over a lengthy time horizon. Furthermore, they may be distributed unevenly. For example, the costs of a given resource enhancement project may be borne by the present generation, but because natural stocks take time to regenerate, the benefits may not be realized until well into the future. Since a given dollar value realized in the future is worth less than a dollar in the present, future costs and benefits must be converted into present value terms in order to make a meaningful comparison.[11] The mathematical process of calculating the present value of future costs and benefits is called discounting, and relies on an assumption about the opportunity cost of funds, known as the discount rate.[12] Specifically, using a higher discount rate decreases the present value of future dollars relative to current dollars. A "zero" discount rate suggests that a dollar in the future is worth a dollar today. Given that the choice of discount rate can significantly influence the discounted present value of the costs and benefits of a policy change, it is a controversial topic (Mikesell 1977, Kahn 1998:111-113). While it seems there is no 1 right discount rate, there are certain situations that suggest the use of a rate similar to the risk-free market rate of interest, such as that earned on long-term U.S. Treasury bonds, and other situations where it makes intuitive sense to use a discount rate lower than the risk-free market rate.

Finally, it is important to acknowledge the potential uncertainty surrounding the outcomes of particular policies and thus the difficulties of choosing 1 policy over another using BCA. Because there is likely some uncertainty in the parameter estimation of in any cause-and-effect relationships involving wildlife, it is useful if not necessary to perform some type of sensitivity analysis. For example, consider a policy designed to reduce the growth of urban deer populations through the implementation of controlled public hunts or birth control. In either case, natural parameters such as deer population size, fecundity and mortality rates will be unknown but will have an important bearing on the efficacy of the policy change in accomplishing its objectives. It is therefore critical that the assumed parameters in the study are varied over a reasonable range of values to examine how sensitive the study's results are to changes in the assumptions. At a minimum, upper and

[9] One of the first empirical applications linking wildlife stocks and consumer surplus in a dynamic setting was Brown and Hammack's (1974) study on managing waterfowl. Other studies that use a dynamic approach include Keith and Lyon (1985), Cooper (1993), and Schuhmann and Easley (2000).

[10] Distribution of costs and benefits across different types of individuals presents a similar, and perhaps more complicated issue, as monetary costs incurred by 1 party must in some way be compared to monetary benefits accruing to another. Such equity considerations of wildlife policy changes, while typically not the focus of economic analysis, indeed warrant attention, as these matters are likely to be controversial.

[11] For example, for a 5% discount rate, US$100 today is valued at US$105 in 1 year.

[12] The present value, *PV*, of a dollar value, *v*, realized *t* years in the future is calculated as: PV = v , where r is the discount rate.

lower bounds on the results should be presented (Kahn 1998).

### Empirical Estimation of the Benefits and Costs

For most goods and services, a starting point for estimating costs and benefits is the market price that facilitates a transaction. The price paid by consumers is at least a lower bound on the benefits derived by the consumer and an upper bound on the costs of production borne by the producer.[13] Yet when it comes to environmental and natural resources such as wildlife, the market price captures only the market value of the resource. For many wildlife resources, the nonmarket value may likely comprise a larger share, if not all, of the total value of a resource to society than the market value. This, of course, presents a challenging problem in measuring the value of the resources in our BCA objectives or even in "pricing" the resources appropriately so that they are not overused and exploited inefficiently. Indeed, since price is a rationing device that allocates resources to their highest valued use, those resources that are not priced or under-priced are likely to be inefficiently allocated, consumed, and/or produced.

The remaining part of this section will discuss a number of techniques environmental and natural resource economists use to quantify the value society places on these resources. The 2 general categories include revealed preference methods and stated preference methods. Before we discuss these methods of valuing environmental resources, we briefly discuss the 2 types of values that comprise the full value of a resource, use and nonuse values (Fig. 2). The section concludes with a discussion of empirical cost estimation.

### Use and Nonuse Values

When considering the benefits derived from wildlife and other environmental goods and services, it is important to recognize that the value of a particular resource at any point in time can consist of both a use value and a nonuse value. Use value, as suggested in Boyle and Bishop (1987), can be divided into both consumptive use and nonconsumptive use, the common denominator being direct contact with the resource. Consumptive use would entail such uses as bagging a deer, catching a fish, or trapping a raccoon. Essentially, consumptive use means extracting the resource from its habitat (Boyle and Bishop 1987). Nonconsumptive use, alternatively, relates to the uses of resources that do not involve extraction. Bird watching and photographing wildlife, for instance, are considered nonconsumptive uses.

[13] Price will only represent the actual market benefits and market costs for the marginal consumer and producer. For the inframarginal consumers and producers, prices underestimate benefits and overestimate costs.

Nonuse values, alternatively, account for the intangible uses of a resource and represent what be referred to as the intrinsic value of the resource. Nonuse values themselves capture differ concepts of value. Initially, nonuse values were categorized as either an option value, which was introduced by Weisbrod (1964), or existence value, which was introduced by Krutilla (1967). Option value is the value people place on the future availability of a resource even though there is uncertainty surrounding its future use. Existence value is the value an individual places on a resource simply for its' preservation. As Brookshire et al. (1983) suggest, "Some individuals may derive satisfaction from knowing that a certain species and natural environments exist and therefore may be willing to pay for the preservation of such natural resources." For specificity's sake, 2 additional categories were defined since Weisbrod's (1964) and Krutilla's (1967) original classification, including bequest value and altruistic value. Bequest value captures the nonuse value for the resource today so that future generations will have the opportunity to use it, while altruistic value captures the nonuse value for a resource today by an individual so that others have an opportunity to use it today. It should be emphasized that while most environmental economists agree that resources can have both existence and option values, there is some disagreement about whether our valuation methodologies can measure these values accurately (Brookshire et al. 1983, Boyle and Bishop 1987, Brookshire and Smith 1987, Madariaga and McConnell 1987).[14]

### Revealed Preference Methods

Revealed preference methods examine decisions that individuals make regarding market goods that are used together with nonmarket goods to reveal the value of the nonmarket good (Kahn 1998). These methods require that a link be established between changes in the environmental good or resource and changes in the observed behavior of people. For instance, decreases in water quality along a particular stretch of river may result in fewer fish. Anglers, whose objectives are to catch fish, may move to another part of the river or to a different river altogether. Thus, a link can be established between the environmental resource, in this case water quality (or fish), and observed behavior – angler fishing location. In establishing this link, it is important to account for any other potential factors that may be causing behavior to change and, as mentioned in Bockstael and McConnell (1999), requires that we observe "paired observations" between different levels of the environmental good and the associated levels of observed

[14] These values are also referred to as "passive use" values. That people readily contribute to wildlife or other environmental organizations is often cited as evidence that these values exist. However, the topic of nonuse values in economics is not without controversy. Again, see the above references for a more complete discussion.

behavior. With this information, we can estimate a function, such as a demand or marginal willingness to pay function, which will allow us to estimate the value of particular changes in an environmental resource. The 4 most commonly used revealed preference methods we will discuss are the travel cost method (TCM), the random utility model approach, the hedonic travel cost method (HM), and the household production function (Bockstael et al. 1987, 1989).

The TCM, 1 of the most widely used valuation techniques, uses information on the travel costs and number of trips to a particular site to estimate the value of that site or the value of resources that comprise the site. As Mendelsohn and Brown note, the TCM uses the fact that people travel different distances to a particular site and therefore can be expected to participate at different levels. Using the distance traveled as a proxy for the price of a trip and the number of trips as the quantity, individual or group demand curves can be estimated for a site.[15] The net benefits of a particular site or the value of the resources within each site can then be estimated. That is, with data on the number of visits a group of hunters, anglers, or recreationists take in a given period of time, a measure of the quality of the resource realized on each trip,[16] and the travel costs incurred, we can estimate a demand or marginal benefit function for trips. This function can then be used to calculate the change in benefits from a change in the resource quality measure.

There are a number of wildlife valuation studies that use TCM. One of the first TCM studies valuing wildlife was by Miller and Hay (1981), who estimated the degree to which hunting site characteristics and travel distances affect hunter participation and calculated the annual losses to duck hunters from a 10% loss of habitat at waterfowl hunting sites. Sandry et al. (1983) used the TCM to evaluate Rocky Mountain elk tag pricing strategies in Oregon. They found that increases in the current price for elk hunting tags would increase revenues to the Department of Fish and Wildlife and would more closely equilibrate the demand for elk hunting permits with the departments exogenously determined supply. Brooks (1988), in using the TCM to estimate the net economic value for deer hunting in Montana, found that hunters would be willing to pay an additional US$108 more per trip, on average, than they currently pay.[17]

The Random Utility Model is a variation of the travel cost method where the choice of recreation site is viewed as a function of the satisfaction, or utility hereafter, derived from the alternative sites. A trip utility function is specified to be a function of site characteristics and can be estimated using data on individual trips and site characteristics. The estimated utility function can then be used to approximate the consumer surplus from a change in 1 of the site characteristics. For recreational hunting trips, a characteristic likely to influence site choice is the expected success rate.[18] This technique can therefore be employed to value hypothetical changes in hunting success rates. While there are many applications and studies using RUMs, including valuing changes in water quality and characteristics associated with fishing (e.g., Bockstael et al.1987, Morey et al. 1991, Kaoru et al. 1995, and Schuhmann 1998), analyses of other wildlife-related resources are sparse.[19]

The hedonic travel cost method is used to measure the value of separate characteristics of a resource (Brown and Mendelsohn 1983, Mendelsohn 1984, Palmquist 1991). In essence, by observing purchases of market goods – travel, in this case – which must be made to gain access to the resource, one can impute the implicit prices for characteristics of the resource. This method relies on differences in the travel costs and expenditures across agents in their gaining access to the resource to value the associated environmental amenities. For example, using data collected on the travel costs across individuals to various sites they visit, as well as the physical and environmental attributes of each particular site, we can estimate a function describing how travel costs (i.e., price) are related to those attributes. From here, the contribution of the environmental good to the "price" of travel can be approximated. Furthermore, the value of a change in the environmental good can be estimated.

Traditionally, hedonic models have been used to value characteristics of market goods but there are numerous applications to natural resources. Brown and Mendelsohn (1983) used a hedonic travel cost model to estimate the value of congestion, scenery, and fish density to Washington steelhead anglers. Livengood (1983), in his analysis of white-tailed deer hunting in Texas, used a hedonic travel cost model to value the deer harvested on leased land. Deer were again the focus of Mendelsohn's (1984) study that used the hedonic travel cost method to value increases in deer density in Pennsylvania. Finally, Englin and Mendelsohn (1985) used the hedonic travel cost method to estimate the impact of forestry on recreation in the Cascade Mountains.

The household production function (HPF) approach estimates wildlife values by linking changes

[15] It should be emphasized the TCM is generally used to estimate the demand at 1 particular site and not a number of sites.

[16] For example, the number of fish caught on a fishing trip or a measure of the quality of game hunted.

[17] Also targeting deer hunting, Balkan and Kahn (1988) estimated the value of increasing deer populations in New York. Finally, Adamowicz et al. (1990) developed a variant of the traditional TCM, essentially a sequential TCM, to value sites that allow big horn sheep hunting in Alberta.

[18] Note that the change in success need not be an improvement. The loss in benefits from a decrease in hunting quality can also be estimated with this method.

[19] Schwabe et al. (2000) used a RUM to estimate the value of increasing deer season length as a means of increasing hunter welfare and decreasing the impacts of deer populations on crop damage and deer-vehicle collisions.

in observable household outputs to changes in environmental resources. In effect, we assume that the household, in its efforts to maximize utility with a limited budget of time and money, produces outputs such as recreation time, expenditures on certain wildlife-related goods, number of trips to engage in a specific type of resource-related activity, harvested deer, and other goods. The relationship we have to establish, then, is between changes in the resource in question and 1 or more of the outputs that go into the household production function. This method, unfortunately, is subject to a number of difficulties in the estimation process thus limiting its use in evaluating wildlife resources (Pollack and Wachter 1975, Bockstael and McConnell 1981, Brown and Mendelsohn 1983). Empirical applications of this method include Keith and Lyon (1985) who estimated the value to hunters of an increase in a Utah deer herd.

### Stated Preference Methods

Stated preference methods employ survey techniques to solicit value measures directly from individuals by asking hypothetical questions. That is, rather than drawing inferences about value from observed behavior, stated preference methods ask people about the values they place on nonmarket goods such as wildlife resources. Obviously, the principal advantage of revealed preference methods is that they use actual market behavior as a starting point for the estimation of resource values. However, this also means that these methods are insufficient to capture the value of resources that are not associated with direct use. Stated preference techniques, which rely on the use of surveys, are the only methods able to capture the nonuse values associated with a resource. Two of the more popular stated preference techniques are the contingent valuation method and conjoint analysis.

The contingent valuation method (CVM) is the most commonly employed of the stated preference techniques for valuing environmental and natural resources. It has been used to value wildlife resources for both consumptive and nonconsumptive purposes, including wildlife such as bald eagles and striped shiners (Boyle and Bishop 1987), grizzly bears and bighorn sheep (Brookshire et al. 1983), deer (Loomis et al. 1989a), and waterfowl (Brown and Hammack 1973, Bishop and Heberlein 1979). Survey respondents are presented with detailed descriptions of resource quality and are asked to either directly state their willingness to pay for hypothetical changes in that quality or to indicate whether or not they would be willing to pay a specific value. While the contingent valuation method is the subject of a great deal of controversy,[20] it also holds great promise for the study of nonmarket values. In 1990, as a result of the Exxon Valdez oil spill, the National Oceanic and Atmospheric Association (NOAA) drafted the Oil Pollution Act, which specifically advocated the use of the contingent valuation method for measuring nonuse values associated with damages to natural resources.

Within the environmental and natural resource valuation literature, conjoint analysis (CA) is the less used stated preference technique to date. Essentially, it differs from CVM in that conjoint analysis asks people to make hypothetical choices across pair-wise bundles of goods or by having people rank a number of alternatives with a "price" being 1 alternative or characteristic in the bundle. While this technique is just starting to gain momentum in the environmental and natural resource valuation literature (e.g., Louviere 1988, Adamowicz et al. 1999), it has been widely used by researchers in other disciplines (e.g., transportation, marketing, and psychology) and for other problems.

The above discussion provided a brief description of some of the most widely used environmental resource valuation techniques and some examples of how they have been used. Some of these methods are accepted by federal and state agencies. For example, the National Marine Fisheries Service has designated random utility models as its chosen valuation technique. CVM and TCM have been supported and used by the U.S. Water Resources Council, and by state fish and game agencies in such states as Oregon, Nevada, California, Idaho, and Maine (Loomis 1993). While these methods are widely used, it is important to stress that none of the approaches mentioned is without its flaws. Indeed, there is continual debate on the validity and tractability of each method discussed above.

## Valuing a Deer – A Comparison Across Applications

A valid question at this point is, how would these different techniques compare in the values they assign a particular resource? Indeed, given there are a multitude of assumptions associated with each 1 of these methods that may likely influence the actual outcome, and given the often tentative links that are made between changes in the environmental or wildlife resource and how people respond or suggest they would respond, we would not expect the values to be equal across methods. Yet, we would hope that the values would not be an order of magnitude off without a good reason. One resource that has been estimated using 3 different techniques is the value of an additional deer. Estimates of the value of an additional deer from outside the environmental and natural resource economics literature have ranged from US$965 (Reed et al. 1982) to US$1,468 (Romin and Bissonette 1996b) in 1996 dollars. Reed et al. (1982) based their estimates on a value determined in a district court case in Golden, Colorado, whereas

[20] Much of the controversy centers on the various types of biases that may result when individuals are asked to state true values for hypothetical changes in resource quality. See Kahn (1998:102-109) for details on these biases.

Roman and Bissonette (1996a) used estimates of total hunting expenditures on licenses and permits in Utah.

Within the environmental and natural resource literature, we can compare the results put forth by Keith and Lyon (1985), Loomis et al. (1989a), and Schwabe et al. (2000). As mentioned above, Keith and Lyon (1985) use a household production function approach to estimate the value to recreational hunters of a 1-unit increase in mule deer herd size. They combine an estimate of the value of increasing hunter success with an estimate of the responsiveness of hunter success to an increase in deer herd resulting in an estimate for the value of an additional deer of US$64.26 in 1996 dollars.

Loomis et al. (1989a) used a contingent valuation survey to estimate the willingness to pay for an additional deer while considering both consumptive use (i.e., the chance to bag an additional deer) and nonconsumptive use (i.e., the chance to view a deer). They surveyed California households and deer hunters to obtain willingness to pay estimates for deer. Their results suggested that the consumptive value of an average buck was approximately of US$165, and the nonconsumptive value of an average deer was US$15.50. In 1996 dollars these amount to US$208.78 and US$19.61, respectively.

Finally, Schwabe et al. (2000) used a random utility modeling method and estimated the value of a deer at approximately US$182 in 1996 dollars. This value estimate represents what Loomis et al. (1989a) termed the consumptive value. These results are surprisingly similar across studies in that they are less than an order of magnitude different.

## Costs

Of course, there are a variety of costs associated with wildlife resources. There are direct costs the wildlife resources impose on humans, opportunity costs associated with their habitat, external costs they impose on other wildlife resources that humans value, and finally the costs associated with maintaining a particular resource level or population. To perform a proper BCA, all of these costs should be acknowledged and balanced against the benefits of maintaining the resource over time. The example of maintaining a particular deer population provides a more illustrative description of each of these costs.

The direct costs deer impose on humans have been widely discussed and enumerated. One of the direct costs of maintaining a particular deer population is the impact of deer-vehicle collisions on human life, morbidity, and property damage. These costs are typically estimated by using medical and/or insurance costs, or value of life statistics. For instance, Reed et al. (1982) surveyed vehicle repair costs from Colorado State Patrol accident reports and benchmarked those costs against accident values as reported in insurance claims. Hansen (1983) surveyed drivers that had submitted accident reports to the Michigan State Police and obtained values associated with property damage, injury and/or death. Conover et al. (1995) surveyed the literature to come up with an average vehicle repair bill. With the exception of Hansen (1983), though, costs associated with human injury and fatalities were neglected.

Crop damage is another major direct cost imposed on humans by deer. Typically, the costs of agricultural damages from deer can be estimated as the market value of the lost commodity, and/or the cost of implementing preventative measures. For example, McNew and Curtis (1997) reported the extent of deer damage to select grain crops in Maryland by multiplying farmer-reported acreage losses by grain prices at harvest time. Furthermore, these authors used regression analysis of farmer-reported estimates of damages and on-site deer populations to calculate a deer population elasticity of crop damage. This elasticity measure allows for the estimation of the additional damages that would likely be incurred from a given increase in the deer population.[21] In addition to crop damage and vehicle accidents, wildlife also impose direct costs through the transmission of diseases (e.g., lyme disease), wildlife attacks, and damages to households and the timber industry (Conover et al. 1995).

In addition to direct costs, opportunity costs associated with wildlife habitat (the foregone value that their habitat could garner) should be considered. That is, any wildlife resource requires a habitat that could, in most cases, be put to use in other activities. Wenders (1995) asserted that landowners, whether public or private, often receive little compensation or generate little revenue by providing or maintaining an adequate habitat for elk. As such, he suggested that competing interests for the use of the land, particularly by timber or grazing interests, lead to a less-than-optimal supply of habitat for elk from a social perspective. Similarly, Brower and Slangen (1998) discussed how the variety rich vegetation alongside banks and ditches alongside peat meadowland provide excellent habitat for many plant and bird species. Yet, much of this habitat is losing ground to agricultural interests. Indeed, in this instance the opportunity cost of maintaining the habitat is the foregone income that could be generated by cropping it.

Another type of cost is the cost wildlife resources impose on other wildlife resources. While these costs are somewhat more complicated to calculate since they often require "costing" other nonmarket goods, they can have real impacts on valued resources. For instance, in maintaining a certain density of deer per square mile, potential indirect costs the deer impose on other

[21] For example, they estimate that a 10 % increase in the Maryland deer population would likely cause US$1.15 million in additional losses in revenues from corn, soybeans and wheat.

wildlife resources include loss of habitat for herb and seedling species, or loss of the songbird species habitat (Guynn 1999). Coyotes, by preying on sheep, may impose additional costs on humans via less wool. Unless these "other resources" have particular market-based values, imputing a cost from their potential decline would require some type of valuation study described above.

A final type of cost is the cost of interventions to maintain wildlife populations. As mentioned previously, the USDA Animal Damage Control Program spent over US$26 million in 1988 alone on efforts to control damages from wildlife, while the administrative costs reached US$11 million. In addition to administrative costs, there are often costs associated with installing and maintaining structures intended to either maintain existing wildlife populations or reduce the damage wildlife population impose on society. For example, efforts to reduce deer-vehicle collisions have included the use of a variety of technologies, including deer whistles on cars, deer signs, fences, underpasses and overpasses, and reflectors (Romin and Bissonette 1996b). With the exception of administrative costs, estimating the costs of installation and maintenance is relatively straightforward. For operating and maintenance costs, the market price on materials and labor provides a good estimate of their annual costs. For installation costs, again, the market prices on materials and labor (discounted over the life expectancy of the project) provide a suitable annual cost estimate.

Thus, in estimating the costs of maintaining wildlife resources or populations, market prices often provide a good starting point. Indeed, in calculating the costs associated with human and property damage, the land foregone for habitat, and mitigating or maintenance factors market prices provide a suitable approach. Yet, for the external costs on the environment and other wildlife resources, as well as for the enumeration of the associated administrative costs, other means will likely be necessary.

## POLICY IMPLEMENTATION ISSUES

Once the estimated optimal population size is found, the question of how to achieve that level arises. Or, if the current level and optimal level are the same, how can such a population size be maintained? Two general approaches to regulating wildlife that may be used to achieve population objectives include open-access regulations and limited-entry regulations (Kahn 1998). Open-access regulations are those regulations that modify user behavior yet not user participation directly. Limited-entry techniques try to encourage the desired behavioral outcome through the use of economic incentives.

### Open-Access Regulations

Open-access regulations, similar to what are generally termed "command and control" style strategies, call for the direct regulation of the users of the wildlife resource through rules or standards. As such, they mandate restrictions as to how, when, or where users can utilize the resource. For example, open-access regulations in hunting would include restrictions on the type of guns allowed, allowable hunting days per week or weeks within the year, or limits on the number of bucks or does a hunter can bag. These regulations are designed to maintain the wildlife resource population at a desired level, but because they do not directly target who participates, they do not necessarily serve to ration or allocate the resource toward the users who value it the highest. The restrictions imposed by open-access regulations increase the costs to the users of the resource. Hence, indirectly, open-access regulations can lead to relatively less participation by less efficient users of the resource if the added costs to the individual hunter of complying with the regulations exceed the value generated to the individual from hunting. For example, restricting allowable hunting days to Mondays through Saturdays imposes added time constraints on users relative to allowing hunters the option of hunting Monday through Sundays. Similarly, limiting the allowable weapons to shotguns rather than rifles limits the range at which hunters can bag deer, and again imposes an additional cost.

Interestingly, while restrictions on use can impose additional costs directly on the users of the resource, the intent of these restrictions is often to limit the costs that individual users indirectly impose on other users and nonusers. For instance, as Lueck (1995) pointed out, some hunters can impose harmful effects on others by using certain weapons that increase the probability of an injury to a nonuser (dynamite, high-powered rifles, etc), or by hunting in areas of high incidence of non-hunting human presence (roadsides, urban areas, etc.). As such, open-access regulations can directly address these potential problems.

Yet, it should be noted that while open-access regulations can indeed minimize the impact of user behavior on others, users are still likely to impose costs on each other given that access to the resource is still open to all who are willing to overcome the additional constraints. Since the regulation does not directly ration participation rates, the resource is open to uncontrolled access whereby decisions by individual users of the resource rarely consider the impact of their actions on other users.

### Limited-Entry Regulations

Alternatively, limited entry regulations, similar to incentive-based instruments, are a flexible approach that also target optimal population levels but do so by encouraging private interests to coincide with social interests. This works in 1 of 2 ways. The regulation may include either a tax or a fee to limit participation (e.g., a fee for the right to fish or hunt in a given area). Or, this regulation can require a system of permits or licenses that give the owner the right to use the resource. Preferably, the permits are salable or transferable and thus can be reallocated to the users that value the resource the most.

### Evaluating Open-Access and Limited-Entry Regulations

In deciding upon the economic merits of a particular strategy within these 2 general alternative approaches, 3 criteria emerge. First, does the strategy allocate the resources to the users who value it the most? Second, are the users utilizing the resource in the most cost-effective manner? And third, are the full costs of using the resource borne by those users who are realizing the benefits from its use?

The incentive-based systems of transferable permits and fees do tend to allocate the resource to those who value it more, as those who are not willing to pay the fee or buy the permit will not have access. Even if the fee is initially set too low, it still has the effect of excluding users whose value of using the resource is lower than the fee. In the case of transferable or marketable permits, we are likely to observe a flow of permits towards users who value the resource more (buyers of permits) and away from users who value the resource less (sellers of permits). Another advantage of tax or permit systems is that they minimize the costs to users in engaging in the activity. That is, users have the flexibility to choose the least costly manner in which to use the resource. Yet, and this is necessarily a problem with this sort of regulation, users may still impose external costs on others depending on the methods they choose to utilize the resource. So, while the regulations minimize the costs to the users, they may impose undue external costs on other users or on society in general.

In principle, then, both types of policies could achieve the desired objective of reducing damage to the stock and therefore promote growth. However, the limited-entry strategy is likely to meet the objective at lower cost since open-access strategies treat all users of the resource the same, regardless of how much it costs them individually to comply with the standard.[22] We therefore see that the limited-entry regulations meet the requirements of the first 2 criteria – allocating the resource to the highest valued users and minimizing the costs of using the resource to the user, whereas open-access regulations do not. However, with its ability to target specific harmful actions, open-access strategies may be more effective at reducing some of the potential external costs that users may impose on others. It would seem, then, that the optimal strategy would include components of both types of regulation. The limited-entry component serves to limit the users to those that value the resource the most while the open-access regulations would minimize some of the negative effects that users impose on others. Indeed, this combination strategy is what we often observe in practice.

Finally, our discussion has not fully addressed the issue of public versus private land ownership. This distinction is extremely important for the optimal management of wildlife resources, including both optimal population size and efficient regulatory strategy. For a complete discussion of the public/private land use dimensions and their impact on policy, see Loomis (1993) and Lueck (1995).

## CONCLUSION

There is little uncertainty that wildlife-human conflicts impose significant costs on society. Yet, as most wildlife managers, hunters, and nature enthusiasts would agree, there is also enormous value associated with these same wildlife resources. In this review, we have attempted to provide a framework to help decision-makers manage these wildlife resources more efficiently. In essence, we have illustrated the potential use of benefit-cost analysis to improve the efficiency of wildlife resource management. Optimal wildlife populations can be estimated through a careful balancing of the benefits and costs of wildlife to society, both now and in the future.

In particular, we described a number of methods for placing a value on certain types of wildlife resources. These methods, including the revealed preference and stated preference techniques, can help to quantify the value of these resources to society, particularly those values which might not be assessed when using the traditional market-based pricing approach. Indeed, the nonmarket values from many of the wildlife resources may likely overshadow their market-based values. Furthermore, we introduced a variety of cost categories to describe the wide range of opportunity costs associated with maintaining wildlife populations, including (1) harmful wildlife-human interactions, (2) impacts on other wildlife resources, (3) foregone production or value of their habitats in other uses, and (4) the direct regulatory and mitigating costs of control. We showed that a stand-alone market-based approach to wildlife resource management will fail to balance the full range of costs and benefits from maintaining wildlife resources. Rather, these types of resources are better managed, with optimality in mind, by using a

[22] For economic efficiency, that is, to achieve the desired result at lowest possible cost, and given that users are likely to differ with respect to ability and effort, the cost-effective strategy allows the user the freedom to choose the most efficient means of using the resource.

combination of regulatory strategies that both maximize the value of the resource to the actual users while minimizing the private and social costs to other users and nonusers.

Finally, in reviewing the literature, 2 strands seem to have evolved. There is the literature associated with valuing wildlife resources, and includes the work by Hammack and Brown (1974), Loomis et al. (1989b), Keith and Lyon (1985), and Mendelsohn (1984). There is also the literature on the costs of wildlife management and efforts to reduce harmful human-wildlife interactions, including the work by Conover et al. (1995), Roman and Bissonette (1996a,b), and Reed et al. (1982). In essence, there is increasingly more literature and research on both the benefits associated with wildlife resources and the costs associated with wildlife resources. What seems to be lacking, then, is a merging of these 2 strands into useful and informative exercises investigating optimal wildlife management strategies. With due acknowledgement of the problems associated with estimating both the values derived from wildlife resources and the costs that are associated with their maintenance, the principle elements required for proper benefit-cost analyses do exist.

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