Valuation of Ecological Resources and Functions

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ABSTRACT / Ecological resources are natural resources that provide certain necessary but overlooked system maintenance functions within ecosystems. Environmental economics is in search of an appropriate analysis framework to determine economic values of such resources. This paper presents a framework that estimates and compiles the components of value for a natural ecosystem. The framework begins with the ecological processes involved, which provide functions within the ecosystem and services valued by humans. We discuss the additive or competive nature of these values, and estimate these values through conventional and unconventional techniques. We apply the framework to ecological resources in a shrub–steppe dryland habitat being displaced by development. We first determine which functions and services are mutually exclusive (e.g., farming vs soil stabilization) and which are complementary or products of joint production (e.g., soil stabilization and maintenance of species). We then apply benefit transfer principles with contingent valuation methodology (CVM), travel cost methodology (TCM), and hedonic damage pricing (HDP). Finally, we derive upper-limit values for more difficult-to-value functions through the use of human analogs, which we argue are the most appropriate method of valuation under some circumstances. The highest values of natural shrub–steppe habitat appear to be derived from soil stabilization.

Ecological resources are defined herein as those that provide necessary but unglamorous system maintenance functions within ecosystems as a result of their role in ecological processes. Ecological functions are frequently overlooked in terms of providing services that are valued by humans. Aside from selected wetland studies, (e.g., Barbier 1994, Constanza and others 1989, Batie and Mabbs-Zeno 1985, Gupta and Foster 1975), relatively little attention has been given to the valuation of common ecological resources and functions in comparison with their more glamorous cousins (e.g., endangered species or unique wilderness sites). This is particularly true of the nation's semiarid rangelands. Moreover, attention to date has focused mostly on values that are generated through human consumptive and nonconsumptive uses (as opposed to environmental functions), which we believe are only one aspect of the social costs and benefits of the functional values of ecological resources.

Ecological resources involve a number of natural processes that in turn provide a range of functions whose services are explicitly or implicitly valued by human beings. In attempting to provide a prototype empirical estimate of ecological resource values for shrub–steppe habitat, we follow the general approach of Thibodeau and Ostro (1981).

The key objectives of our study are to provide an ecological valuation framework that (1) aggregates the values of goods and services rendered by selected ecological functions, and (2) determines defensible upper and lower limits for the values of these resources. We believe that interpolating between these limits will provide more comprehensive and realistic estimates of the value of ecologically derived goods and services, thereby providing policy makers with a better understanding of the social costs and benefits of ecological preservation.

KEY WORDS: Natural resource economics; Ecological economics; Ecological resources; Shrub–steppe; Environmental valuation; Cost; Benefit; Value

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Approach

Test Case: Shrub–Steppe Habitat

Our study estimates the values of services rendered by shrub–steppe habitat of the intermountain West. In its natural state, this habitat maintains a vegetative cover that minimizes soil erosion while controlling

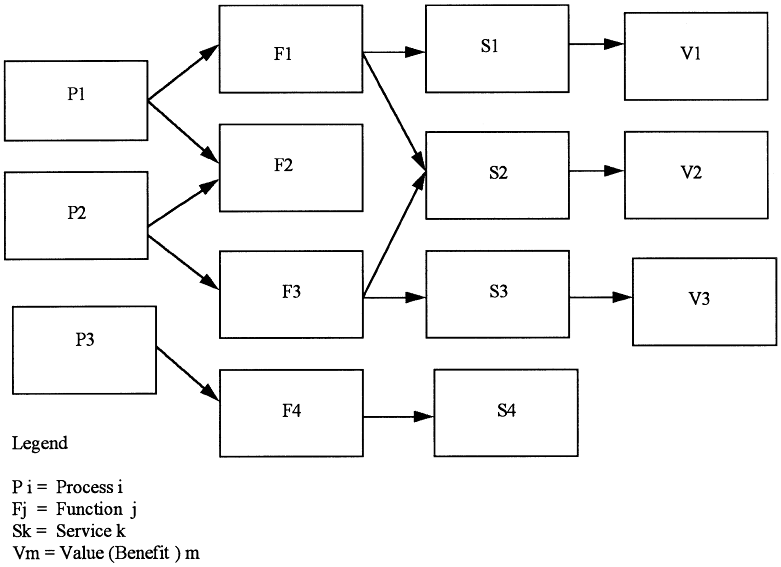

Figure 1. Relationship among ecological processes, functions, services, and economic values (benefits).

water filtration. It also serves as vegetative cover and food supply for numerous species of mammals and birds and helps moderate climate (Rogers and others 1988).

When valuing ecological resources, it is helpful to distinguish between ecological processes (interactions among the elements of the ecosystem), functions (aspects of the processes that affect humans or key aspects of ecosystem itself and can be thought of as the purposes of the processes), and services (attributes of ecological functions that are valued by humans) (see Figure 1). A process may have more than one function, or a function may rely on one or more processes. A function may or may not have a corresponding service, depending on whether humans value it. The measure of the value of the service is its economic benefit. For example, the shrub–steppe habitat of the intermountain West involves many complex chemical and biological processes and interdependent organisms. Many of these processes have functions that appear to matter only to the organisms themselves and appear to provide almost no services to people. This is where ecologists and economists wind up in sometimes acrimonious debate. Ecologists are concerned with study of the processes and functions of the ecosystem, functions that they assume would have value to humans if properly understood. Economists (and most other nonecologists) are more concerned with services and benefits understood by laymen, rarely consider the full range of processes, and thus may miss key functions whose services they should consider.

As a result, although shrub–steppe is the dominant land cover across vast areas of the intermountain West and northern Mexico, it is often considered unattractive, poor-quality grazing land that is usable for agricultural and/or urban development only when supplemental water is available for irrigation. When shrub–steppe is left in its natural state, the ecological functions and recreational uses it provides often go unnoticed by the general public and land-use planners. However, healthy operation of these processes gives rise to a number of functions that are valued implicitly or explicitly by human beings as services. When the underlying ecological components and processes are at risk, so are many of the functions and services derived from them. We make an attempt to bridge the gap in this paper by beginning with a reasonably complete set of processes, then tracing through the path of functions, services, and ultimately, economic value. Table 1 lists some of the processes, functions, and services of shrub–steppe lands. They are described in the next section.

The necessary perspective on how shrub–steppe habitats perform these functions is gained by referring to various studies conducted at the Fitzner/Eberhardt Arid Lands Ecology (FEALE) Reserve on the Hanford Site in Benton and Franklin counties of south-central Washington State (Figure 2). The 312-km^2 FEALE Reserve is situated on the northeast-facing flank of the Rattlesnake Hills, a long anticlinal ridge, varying in elevation from 150 to 1100 m. The area is semiarid with hot dry summers and cool wet winters. Average yearly precipitation ranges from 260 mm at the higher elevations to 160 mm in the lower areas, falling mostly in the fall and winter (Thorp and Hinds 1977).

Functions and Services of Shrub–Steppe Habitat

The soil stabilization, species maintenance, and biological diversity functions of shrub–steppe habitat are

Table 1. Processes, functions, and services of shrub–steppe habitat

Processes	Function	Service
Plant growth	Soil stabilization	Reduced PM_{10} count: improved aesthetics, reduced respiratory problems, reduced productivity loss through soil erosion, fewer traffic accidents and road closures, less household and vehicle cleaning
Ecosystem dynamics, self-maintenance	Maintenance of selected species, system components	Recreational hunting, horseback riding, nature hikes, birdwatching, education, and research
Nutrient cycling, water capture, photosynthetic capture	Biological diversity	Valued aspects of diversity
Water capture	Water retention in root zone, groundwater recharge, reduced runoff	Water supply from groundwater, reduced flooding, reduced erosion, improved water quality
Evapotranspiration, gas exchange, temperature modification	Niches for biota. Processes contribute to maintenance of local and regional climate	Climate moderation

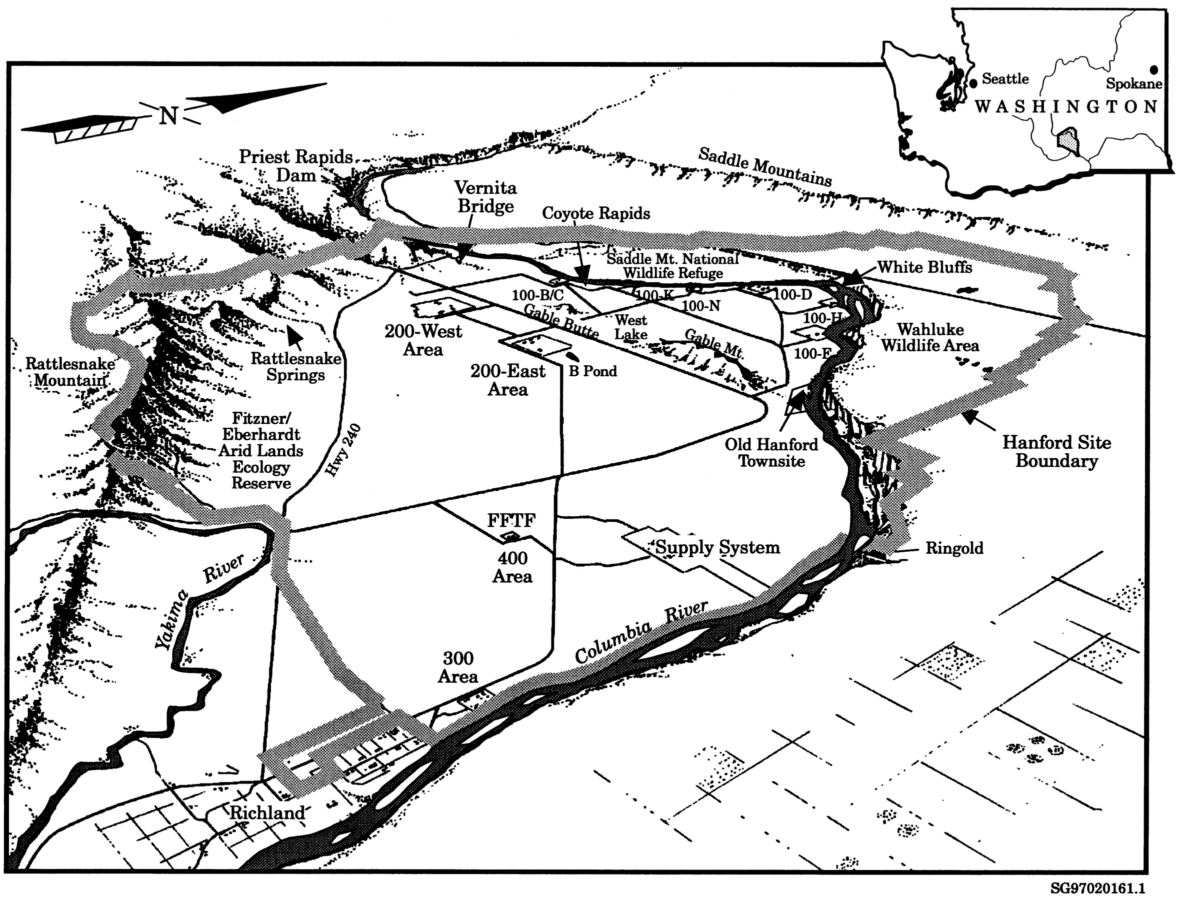

Figure 2. Location of the Fitzner / Eberhardt Arid Lands Ecology (FEALE) Reserve on the Hanford Reservation in south-central Washington State.

discussed below. Shrub–steppe habitats provide many other ecological functions. The vegetation in natural shrub–steppe habitats minimizes water erosion (Link and others 1995). In addition, shrub–steppe habitat provides carbon fixation functions (Rogers and others 1988). However, limited data are available to quantify these latter functions or several of the educational and recreational activities ordinarily conducted on shrub–

steppe habitats. Therefore, this paper concentrates on the valuation of ecological functions associated with soil stabilization, biodiversity, and game habitat. Water retention and climate functions are not considered because they were assumed to be relatively minor and data available on the study area were insufficient to evaluate their associated services.

Soil stabilization. Land cover is critical for reducing wind erosion in much of the arid West, and the impact of fugitive dust on human populations is significant in many regions. Benton and Franklin counties in Washington State have a long history of seasonal dust storms and accompanying air-quality problems. These events often occur during spring and fall and are believed to be in part the by-product of catastrophic wind erosion of fallow farmland. Wind erosion is dependent on a variety of factors, most notably soil moisture, soil type, ground cover, and wind velocity. Pristine shrub–steppe lands in the region have an annual erosion rate between one half and one ton per acre, while cropland in the two-county area experiences a much higher annual wind erosion rate of 10–15 tons/acre–equal to a depth of approximately 1 mm (Holmes 1994). Some bare land may experience loss rates of between 67 (Holmes 1994) and 700 tons per acre (Wohld 1991).

Fugitive dust is a nonpoint source of pollution based on the Environmental Protection Agency's (EPA's) National Ambient Air Quality Standards (NAAQS). Under these standards, particulate matter (PM) measurements in a given area must not exceed 150 μg/m^3 over a 24-h period and/or an average of 50 μg/m^3 over a one-year period. Most of the problems with fugitive dust emissions in Benton and Franklin counties occur on days when these standards are violated, with relatively little impact at other times. Relative to the NAAQS, the Benton–Franklin county region experienced 11 days of noncompliance over a two-year period, from 16 October 1991 through 13 May 1993. The three highest PM_{10} levels recorded in the two-county area are among the worst on record in the United States (Wohld 1991) and have been attributed to airborne dust. The most notable exceedence events are identified in the first column of Table 2.

Several impacts on human activities, including increases in traffic accidents, adverse impacts on health, additional household and commercial cleaning costs, and generally reduced aesthetic value of the environment are among the effects identified during these events. Data in Table 2 on excess automobile collisions (those above the expected number under normal driving conditions) attributed to impaired visibility suggest the possibility of estimating a damage value for automobile collisions and injuries sustained during severe dust storms. Acute respiratory effects are also associated with severe dust storms. Hefflin and others (1994) considered the impacts of PM_{10} levels above 1000 μg/m^3 on the number of emergency room (ER) visits for respiratory disorders in Benton and Franklin counties. Their study calculated the annual, monthly, and daily number of ER visits for respiratory disorders and made correlations between ER visits, PM_{10} levels, and meteorological conditions. Generally speaking, however, no significant effects were found. Finally, while effects on household and commercial cleaning costs have not been studied directly in the region, inferences can be made from studies elsewhere using benefit transfer techniques. These estimates are given in a later section of the paper. These elements of value can be added together as an estimate of the value of the services provided by the soil stabilization function.

Table 2. Dust storms and traffic safety in Benton–Franklin counties

Exceedence dates	PM_{10} (μg/m^3)	Excess collisions[a]	Injuries
10/16/91	1689	8	4
11/03/93	1166	7	3
10/21/91	1035	3	1
9/08/92	596	5	2
4/05/91	350	4	4
4/02/91	281	4	2
12/12/91	212	3	2
9/26/92	183	4	5
8/18/91	174	2	4
1/29/91	165	4	0
5/13/93	155	3	4

[a]Excess collisions are the number above the expected number under normal driving conditions.

Source: Personal communication with Charlie Fable, Washington State Traffic Commission, 25 August 1994.

Species maintenance (game habitat). Shrub–steppe land in the study region also maintains open space and species that provide for a variety of recreational services, most significantly upland game hunting. Based on survey data, state game officials have estimated that 8062 pheasant/quail hunters used the Benton–Franklin county area in 1991 (Wildlife Management Division 1993). Upland bird hunting is particularly notable near cultivated lands and surface waterbodies where cover is adequate.

Agricultural fallow and intensive grazing tend to reduce native food sources and cover for upland game birds (Washington State Department of Fish and Wildlife 1994). However, California quail (*Callipepla californica*) are abundant in Washington and in the Benton–Franklin county area, where an average of 12,871 quail are harvested each year (Washington State Department of Fish and Wildlife 1994). Ring-necked pheasant (*Phasianus colchicus*) also inhabit agricultural and riparian areas in southeastern Washington and are the state's most popular game bird. While agriculture provides some of the food for pheasants, for cover these birds also require the woody and thorny plants that are found on most shrub–steppe lands. Average harvest of pheasants in the Benton–Franklin county area is nearly 20,000 per year (Washington State Department of Fish and Wildlife 1994).

Other popular game birds include chukar (*Alectoris chukar*) and gray partridge (*Perdix perdix*). Chukar reside almost exclusively within shrub–steppe habitat and feed primarily on cheatgrass, seeds, shrub fruits, and insects. Partridge require a combination of open shrub–steppe land and cropland as habitat and feed mainly on cultivated grains, native plant seeds, and insects. In the Benton–Franklin county area, harvests average 1937 and 659 for chukar and partridge, respectively (Washington State Department of Fish and Wildlife 1994). We estimated recreational hunting values by applying traditional travel cost techniques (Clawson and Knetsch 1966) to data on expenditures, travel, and harvest by Washington state hunters (US Fish and Wildlife Service and US Bureau of the Census 1993).

Biological diversity. The shrub–steppe habitat maintains a unique character that includes species and ecosystem dynamics not found elsewhere. Some of this diversity is affected when the surface of the ground or the plant communities are disturbed. The effect can be determined by comparing disturbed and undisturbed lands. Throughout much of the intermountain West, including the FEALE Reserve, the shrub–steppe habitat has been disturbed by farming and grazing, but because access to the FEALE Reserve has been restricted for national defense reasons, the effects of disturbances have been much less and have been documented over the last 40 years (Rogers and others 1988). Thus, the FEALE Reserve lands can be used to calibrate our valuation process. Specifically, we have investigated the effects of plowing and grazing.

Disturbances have adversely affected water, air, soil, and biotic quality of parts of the FEALE Reserve. FEALE Reserve scientists rank soil and plant biotic quality using a rank of zero (for low quality) to 1 (for high quality) (Table 3). Soil quality is closely tied to biotic quality, which is a measure of the biotic complexity and mass of an ecosystem. A low-quality soil is one that is greatly disturbed and has lost much of its organic and nutrient content near the surface. Low complexity is associated with disturbances where biological diversity has been reduced. An ecosystem that exhibits high biological complexity is likely to have high air, water, and soil quality, and can be defined as having high integrity (King 1993).

Table 3. Effects of disturbances on quality scores for environmental dimensions of shrub–steppe habitat[a]

Disturbance	Soil quality	Plant biotic quality	
		Biomass	Species number
Undisturbed shrub–steppe	1.0	1.0	1.0
Short-term grazing	<1.0	1.0	<1.0
Old-field conditions (long-term recovery)	1.0	0.7	0.6

[a]Measured at Fitzner-Eberhardt Arid Lands Ecology Reserve in South-central Washington State.

Effects of Disturbing Shrub–Steppe Ecological Processes on Functions and Services

A variety of human activities disturb ecological processes, reducing some of the functions and services of these habitats. The most common ones are plowing and grazing.

Effects of plowing on functions and services. In semiarid regions, breaking soil by plowing temporarily reduces plant growth, disrupts ecosystem maintenance and dynamics, and greatly contributes to wind erosion of topsoil and reduction of air quality. Some natural recovery is possible. For example, in formerly cultivated old-field areas, dense cover of annual plants prevents wind erosion, while in virgin shrub–steppe habitats, soil cryptogams provide this function.

The short- and long-term effects of plowing have been studied extensively on the FEALE Reserve. The short-term effects of plowing on soil and biotic characteristics were investigated by plowing two old-field communities (Rickard and Vaughan 1988). In that study, litter mass (whose decomposition returns carbon, nitrogen, and other minerals to the soil) was measured and was found to be nine times less on the plowed fields than on the unplowed cheatgrass (*Bromus tectorum*) fields two years after plowing, indicating that plowing degrades soil quality by reducing litter and thus reducing return of organic matter and nutrients to the soil. The value of soil quality was reduced to 0 relative to the unplowed *B. tectorum* fields and remained at 0 the following year. After two years, significant effects on shoot biomass, species composition, and vascular plant diversity persisted in both fields. Short-term effects were mixed. Biomass was significantly higher in the plowed areas of the upper field, compared with unplowed areas (302 g/m^2 vs 85 g/m^2), while it was significantly lower in plowed areas at the lower field (160 g / m^2 vs 290 g / m^2 in unplowed areas). Plowing reduced the number of species from six to five in the upper field, while there was no effect on species number in the lower field. In both the upper and lower fields, two species appeared in the plowed ground of the upper field and two disappeared.

Long-term comparisons of wheat fields plowed prior to 1943 and undisturbed shrub–steppe areas revealed that the formerly plowed fields are now composed of alien annual weeds, with *B. tectorum* the dominant species (Rickard and others 1988). Alien plant species have dominated the farmed area since 1943 with only minor indications of the reestablishment of native perennial species (Link and others 1995).

Plowing adversely affects the biological diversity of shrub–steppe systems, even when they are allowed to recover. In the FEALE Reserve experiments, the native ecosystem exhibited higher biomass production (1640 g / m^2) than the alien-dominated old-field area (1205 g / m^2), so the old-field area received a relative biomass-production score of 0.7 (refer to Table 3). A greater number of vascular plant species occurred in the native system (at least 20) than in the old-field area (at least 12) (Rickard and Vaughan 1988), yielding a relative species-diversity score for the old-field area of 0.6 (refer to Table 3). The native habitat also had a higher population of small mammals (283 / 8000 trap nights vs 88 / 8000 trap nights). Five insect species occurred in the shrub–steppe system and six occurred in the alien old-field system (Rogers and others 1988). Finally, bird populations were higher in the native habitat (41 / 500 m transect vs 23 / 500 m transect) and were represented by more species (6 vs 3) (Rogers and others 1988). In summary, bare plowed land rapidly loses mass and soil nutrients, and water quality declines. Recovery of ground cover stabilizes the soil in the short and long run, but overall biotic quality apparently does not recover, even 50 years after disturbance.

Effects of grazing on functions and services. Cattle grazing is the most important economic use of the shrub–steppe habitat throughout the West. It is also the most widespread source of human-induced stress on the ecosystem (Rickard and Vaughan 1988). Rickard and Vaughan (1988) described the effects of three years of light cattle grazing by measuring yearly shoot biomass. Their experiments showed that when cattle grazed on shrub–steppe habitat, short-term biotic quality was strongly affected. Herbaceous shoot biomass was reduced every year in grazed plots, compared with ungrazed plots. However, recovery was also rapid. A year after grazing ceased, no effect of cattle grazing was observed on either shoot biomass (Rickard and Vaughan 1988) or small mammal populations (Rogers and others 1988). Grazing did appear to reduce seed production, below-ground biomass, and standing dead biomass. At places where the animals congregated, the soil was broken, greatly disturbed, and invaded by alien plant species—tumble mustard and cheatgrass (*Sisymbrium altissimum* and *B. tectorum*) (Rickard and Vaughan 1988). Population densities of insects and arthropods were similar in grazed and ungrazed habitats, with the exception of bluebunch wheatgrass (*Pseudoregneria spicata*) clumps. In this microhabitat in the month of April, mite density was found to be 3.3 times greater than in ungrazed plots (Rogers and others 1988). Soil quality was degraded by grazing at the FEALE Reserve (Wildung and Garland 1988). Soil respiration decreased over a two-year period on grazed land and was attributed to below-ground biological activity associated with above-ground photosynthetic activity.

In summary, the degree of degradation from grazing depends on the type and intensity of grazing. While some recovery from periodic and low-intensity grazing seems possible, intensive grazing degrades the biota, which contributes to increased wind and water erosion. Continuous grazing appears to lead to diminished shrub–steppe plant community and reduced ecosystem, air, water, soil, and biotic quality.

Elements of Ecological Resource Valuation

Our valuation framework identifies a set of environmental functions and services that we value in subsequent sections of the paper. Mathematically, we denote the economic benefits of ecological services of shrub–steppe by the value function $V(A)$, where A represents shrub–steppe acreage. The costs of deriving these benefits reflect the economic trade-offs (i.e., opportunity costs) in forgoing urban and agricultural development. These opportunity costs are denoted by $C(A|U)$, and are conditional upon alternative land use (U). To some degree, these opportunity costs are accounted for by fair market values of shrub steppe in the relevant real estate market. For example, real estate appraisers frequently assign fair market values to shrub–steppe sites based on their capacity to support livestock and other amenities that facilitate converting the land to urban or agricultural use. Deducting these opportunity costs from the value function yields the net

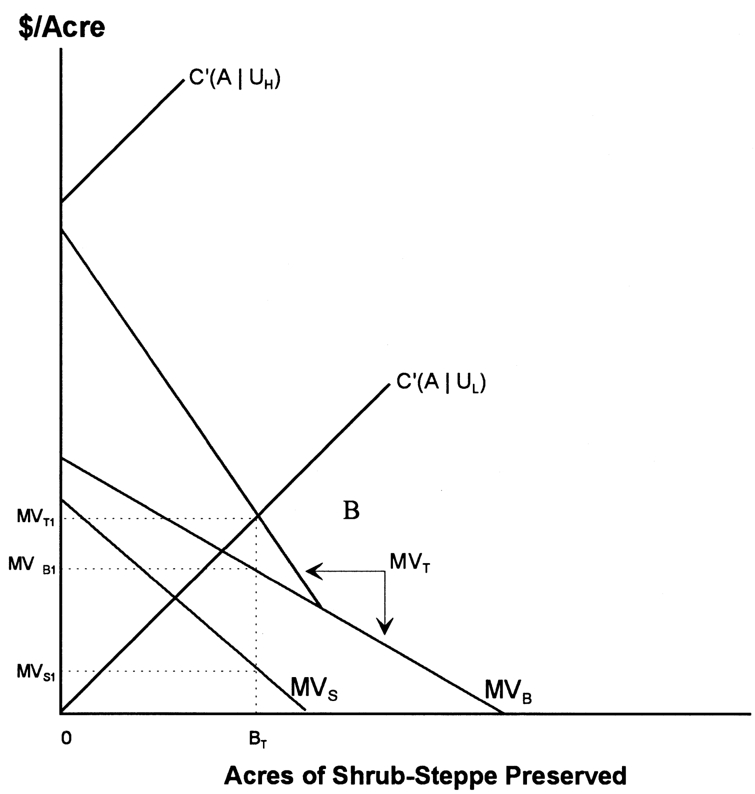

Figure 3. Marginal values of natural landscapes versus development.

preservation value through the following maximizing condition:

$$V'(A) - C'(A|U) = 0, \qquad \text{at } A = A^* \qquad (1)$$

where A^* is the economically preferred amount of shrub-steppe acreage to preserve. Under this maximizing condition, the additional value from preserving an acre is just sufficient to compensate society for the additional opportunity costs.

Much of the preservation value of shrub–steppe sites depends on inherent ecological functions and processes. Because shrub–steppe habitats perform many ecological functions simultaneously, they embody elements of joint production. Moreover, when the services associated with ecological functions are available to many people at the same time, without their having to pay for them, shrub–steppe sites have the characteristics of public goods. Private markets perform poorly at producing public goods. Kneese (1984) notes that "there usually is no private incentive to produce them at all, because while many people could benefit from them, no single individual usually has a sufficient incentive to pay for them." The public-goods nature of shrub–steppe ecosystems is illustrated in Figure 3. In Figure 3 we consider the marginal value curves for services associated with two functions provided by shrub–steppe habitat: soil stabilization (MV_s) and biological diversity (MV_B). Both are shown conventionally sloping downward to the right (i.e., as more acres are preserved in a given geographic area, air pollution declines, biological diversity increases, and the marginal value of these services associated with additional acres declines).[1] In the natural state, shrub–steppe provides soil stabilization and biological diversity simultaneously without one impinging on the other. Assuming that their corresponding services are valued by society, their joint marginal value would be represented by the aggregate marginal value curve for acres of preserved land, MV_T, shown as the vertical summation of the marginal value curves for the individual functions MV_S and MV_B, sloping downward to the right. Meanwhile, the opportunity cost curves of foregone land development having low and high values (e.g., farmland when crop prices are low and high) are shown by upward sloping marginal cost curves $C'(A|U_L)$ and $C'(A|U_H)$. If both ecological values of land exist and are correctly accounted for, the socially efficient amount of

[1]In this discussion, we are implicitly assuming that location and shape of preserved habitat are held constant. In reality, the shape and location of preserved habitat relative to other patches of habitat may be very important to organisms that depend on it.

preservation would occur at point B, with B_T acres preserved at the total marginal value of MV_{T1}, with corresponding marginal values of the two functions at MV_{S1} and MV_{B1}. If landowners were compensated for these ecological values, this sum of marginal values would be just sufficient to compensate a land owner who forgoes land development.

In cases where the opportunity cost of forgoing land development increases dramatically, as represented along the upper marginal cost curve, no land would be set aside for preservation due to the relatively lower total marginal value of preservation when compared to development. A similar result would obtain if both ecological values were overlooked. A variety of indirect valuation methods are available and conceivably could be applied to valuing changes in sagebrush–steppe habitat. Cropper and Oates (1992) list three fundamental approaches: (1) the averting behavior approach, (2) the weak complementarity approach, and (3) the hedonic market approach.

In the averting behavior approach, the cost of purchased inputs can be used to offset the effect of an environmental change such as pollution. The willingness to pay for a marginal change in the level of air pollution, for example, is the price of the averting good, multiplied by the marginal rate of substitution between the averting good (effort and cost associated with household cleaning, for example) and pollution. This is the basis for many of the cost-based assessments of environmental change. In the weak complementarity approach, a good or service, e.g., improvement in water quality or a patch of habitat, is positively associated with changes in purchased goods, e.g., recreational trips. For example, in the case of the recreation value of game habitat, the marginal value of the hunting experience is assumed to just equal the marginal cost of the hunting experience, as measured by travel costs (see, however, Randall 1994). All recreation sites are considered to be perfect substitutes. Weak complementarity may not produce an accurate measure of the total value of a resource because there is implicitly no variation in the valuation of site quality among persons visiting the site (usually overcome with two-stage varying parameters techniques) or because substitutes for the site are not considered (sites are not perfect substitutes), in which case a discrete-choice model may be used. In the hedonic market approach, variations in the value of goods tied to a location (e.g., land values or wage rates) are assumed to be composed of prices for attributes of the location, including environmental values. The problem is that environmental values may be swamped by other factors affecting relative property and wage values in an area or there may not be sufficient variation in environmental quality to distinguish these values from other factors.

Some researchers have engaged in direct valuation of some hard-to-measure functions of the natural environment, such as visual aesthetics or preserving certain species. It is the only known method for estimating nonuse values, such as existence value, that have no markets indirectly associated with them (Freeman 1993). It has been used in at least one case to value arid environments (Richer 1995) and in at least two cases to evaluate soil stabilization in the context of blowing dust (Huszar 1989, van Kooten and Thiessen 1995). The contingent valuation method makes use of a survey instrument to directly question respondents concerning hypothetical values rather than relying on overt market behavior, however indirect. For this reason, contingent valuation has proved controversial (recent literature includes Neill and others 1994, Bishop and Welsh 1992, Carson and Mitchell 1993, Larson 1992, Lazo and others, 1992, Portney 1994, Hanemann 1994, McFadden 1994, Cambridge Economics, Inc. 1992, Freeman 1992, Kahneman and Knetsch 1992, Phillips and Zeckhauser 1989). Richer (1995) has applied CVM to protection of desert areas, but the ecological services at issue in that paper are different from the ones discussed here. Several practitioners have developed and continue to work on methods for reducing the known problems associated with CVM surveys (e.g., Kealy and Turner 1993). After considerable debate by a blue ribbon commission, NOAA (1993) has provided a set of procedural guidelines to practitioners of contingent valuation in an effort to improve the quality of responses. Unfortunately, many CVM surveys (even recent ones) violate these guidelines, leaving policy analysts in these instances to decide whether ‘‘some number is better than no number’’ (Diamond and Hausman 1994).

As noted in the first section of the paper, the various services of shrub–steppe habitat have certain joint-production or public goods aspects to them. When can these services be added together and when must they be substitutes? In particular, can we compare measures for the value of some of the services obtained in various ways by the techniques described above, and under what conditions will we have overlap?

We suggest the following three guidelines:

1. The analyst must decide the nature of the services provided by the ecological resources. Are they separable and independent sources of value, or does the increase

in one service affect the marginal value of the other? For example, does a less dusty environment enhance the recreation experience of hunting, or are the two independent (See Moschini and others 1994)? For the estimates of the values of the various functions to be additive, the functions must be separable, otherwise the values will overlap to some extent and in extreme cases could preclude one another. The services also cannot be substitutes for each other in production. Preservation of shrub–steppe habitat for its soil stabilization function also produces the function of maintaining hunting habitat but is a substitute for its preservation as an off-road vehicle recreation area or for its use in agriculture. In performing this evaluation, it is also important to identify those components of value that cannot be estimated due to theoretical or data problems. It may be possible to conduct a later sensitivity analysis or thought experiment to estimate the potential size of these unestimated values.

2. The analyst should examine the critical assumptions underlying the valuation of each function and determine what kind of estimate results by the method being used. In particular it is important to determine whether the value is marginal, average, or total and whether it is an upper-bound or lower-bound estimate. For example, indirect methods based on aversion behavior or weak complementarity typically yield estimates that are averages of the marginal values to individuals of the environmental service or function at the observed level of consumption of the service. However, such estimates do not include consumer surplus. Thus, for example, an estimate of the value of reducing blowing dust that is based on the averting behavior of increased household maintenance after dust storms would miss the consumer surplus associated with a more agreeable environment. Not having to perform extra dust-related maintenance likely is only a portion of the value to households, since health, recreation, aesthetics, and other elements of value also would be affected by blowing dust and would contribute to the value of reducing it (Huszar 1989, van Kooten and Thiessen 1995).

On the other hand, contingent valuation estimates typically are estimates of total consumer surplus associated with some environmental change [e.g., the level of blowing dust as in van Kooten and Thiessen (1995)], but frequently do not show whether averting behavior such as household maintenance could be affected if the level of the environmental service (e.g., damages due to blowing dust) changed.

The marginal cost of restoring an environmental resource is sometimes used as a proxy for the marginal value of the resource. Because human systems are often inefficient in comparison with natural systems at maintaining ecological processes and functions, such restoration costs can be expected to be upper-bound values. Combining a lower-bound value for one function with a higher-bound value for another function produces a hard-to-interpret total, unless the analyst can determine the likely relative biases of the estimates.

3. Finally, when values are being transferred from other studies, the analyst should observe the protocols that make such transfer of values possible.

In evaluating shrub–steppe habitat, our most difficult-conceptual problem was defining some of the more obscure services conferred on people by ecological processes. Unlike consumptive uses (like hunting) and some nonconsumptive uses of resources (like bird-watching) with which laymen are familiar, valuing some ecological functions requires considerable understanding of the potential outcomes when the integrity of an ecological resource is compromised, an understanding that is remote from the experience of most people. This makes it unlikely that these functions would be incorporated in, say, travel costs or hedonic housing values and also makes direct inquiry techniques problematic. A successful application of the contingent valuation methodology (CVM) in this case would require that the trained ecologist's understanding of ecological resource processes and functions of shrub–steppe habitat be very carefully conveyed to survey respondents (Samples and others 1986), and even so risks underestimating the values of associated ecological services. On the other hand, the one other promising technique, that of using costs of human analogs as a proxy for value, probably suffers from the frequent inherent inefficiency of human systems in supplying functions of nature.

We measure economic values of ecological services (and, ultimately, the resources themselves) by applying several alternative valuation techniques according to their own protocols and then comparing the results where possible. In this way we compensate to some degree for the limitations of the individual techniques. Some of the consumptive and nonconsumptive use values are estimated using the benefit transfer method (Brookshire and Neil 1992, Boyle and Bergstrom 1992), wherein researchers adapt existing environmental valuation estimates to new study sites. We adapt CVM estimates of the aesthetic value of air quality (Rowe and others 1980), travel cost methodology (TCM) estimates of the recreational value of upland game hunting sites (Adams and others 1989), and hedonic damage price

Table 4. Summary statistics of land value per acre, 1993–1994, Benton County, Washington

Class	Observations (N)	Mean value ($)	Standard deviation	Coefficient of variation
Urban	17	9208	5907	0.64
Irrigated agriculture	11	1484	505	0.34
Dryland agriculture	9	248	77	0.31
Pasture	11	67	38	0.57

Source: Benton County Assessor's Office.

(HDP) estimates of the value of reduced offsite wind erosion (Huszar 1989).

Valuation of Shrub–Steppe Ecosystem Functions

As discussed above, shrub–steppe habitat embodies ecological functions that provide services valued by humans (e.g., maintenance of water and air quality, recreation opportunities, and biological diversity). We now attempt to value these services, using the valuation framework and methods discussed above. Because of the difficulty of estimating the value of environmental services provided by shrub–steppe habitats, we use information from a variety of sources.

Opportunity Costs of Preservation

Location and prospects for development appear to be among the most important variables that determine the fair market value of land in its natural state. These variables depend on the physical, legal, and economic characteristics of the land. Different types of development have different effects on the ability of the land to perform some of its natural functions. For example, dryland farming as currently practiced in the region reduces vegetative cover and the ability of the land to maintain clean air and water supplies. However, dryland farming may be compatible with some recreational uses (e.g., hunting).

Data used for estimating opportunity costs of shrub–steppe preservation were obtained from the Benton County Assessor's Office (Table 4). They represent all 48 sales transactions recorded during the year 1993–1994 in Benton County, involving parcels of rural undeveloped land, ranging from 1.2 to 640 acres in size, and totalling 7700 acres. The sample contains 17 property transactions for residential and/or commercial development and 31 transactions involving property destined for agricultural development. Sales of land destined for agriculture were further categorized based on whether the land was irrigated or whether it would be used as dry pasture land or dry farmland.

Estimating Benefits

Below, we attempt to estimate benefits or the value of services from native shrub–steppe habitat in Benton and Franklin counties. In some instances, values are measured by the costs of replacing or maintaining these services using human analogs, such as various soil stabilization measures or the acquisition of hunting sites. In other instances, other valuation approaches are used to provide information on the willingness to pay for environmental amenities, such as the aesthetic value of cleaner air, or the recreational value of a hunting site.

Value of soil stabilization. Economic damages due to blowing dust can be reduced by preserving natural shrub–steppe ecosystems. This suggests an in-situ economic value associated with the soil stabilization function of shrub-steppe. This value is illustrated in Figure 4.

In Figure 4, the marginal willingness to pay for preserving shrub–steppe habitat for soil stabilization purposes (S_1 in panel A) is related to the sum of the damages done by blowing dust ($M_i + R_i$ in panel B). While there are other damages (discussed briefly below), the two principal damages in Benton and Franklin counties are the impacts of blowing dust on public health and safety and on household maintenance. Preserving acreage (movement from B to A in either panel) reduces the marginal and total damage due to high dust levels. The marginal value declines from S_1 to S_0, as does total damage. The zero damage point is shown as 0 in both panels.

Pricing of public health and safety. Damage values were estimated for cost and inconveniences to the traveling public due to road closures and additional road maintenance required after acute dust storms. Some information is available on these costs. Between 1 July 1990, and 30 June 1991, various dust storms impaired driver visibility such that 26 roads spanning 308 miles were closed for a total of 161 h in Benton and Franklin counties. In addition, the Washington State Traffic Safety Commission calculated that traffic accidents associated with blowing dust cost an average of $2413 for property damage and $8495 per injury (Table 2). The Washington State Department of Transportation (WDOT) estimated that the administration of these road closures cost taxpayers $45,000 and necessitated the clean-up of approximately 11,000 tons of windblown soil at about $200/ton from ditches and roads in 1990. To complete the estimate, we assumed an instantaneous average vehicle loading on the rural highways involved in the closure of two vehicles per mile and an

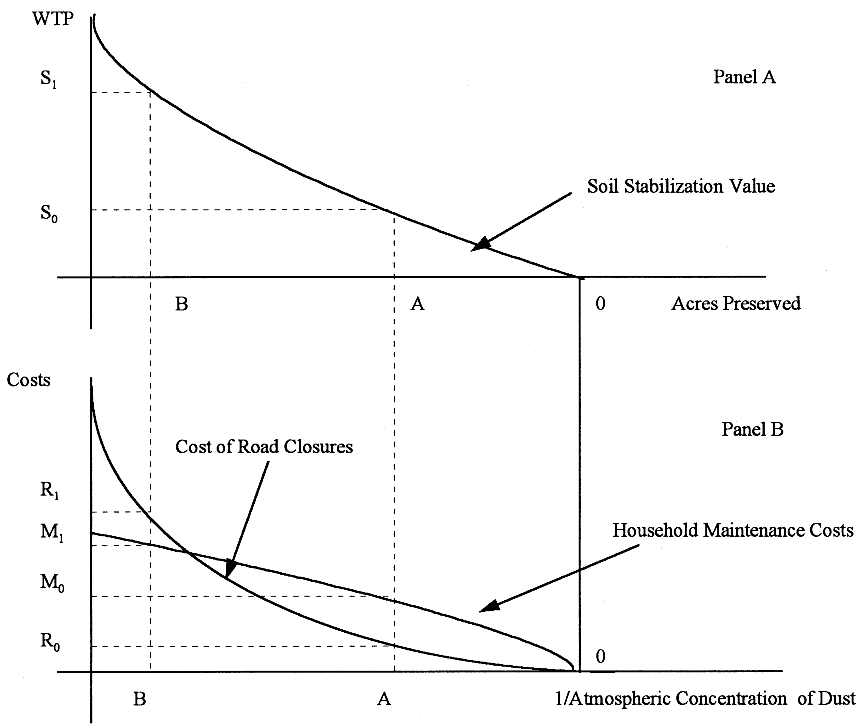

Figure 4. Relationship of damages by windblown dust to acreage preservation value for soil stabilization.

average delay cost of \$23/vehicle/h during road closures.[2] We also assumed that the primary acreages contributing dust were the dryland farms in Benton County (which are upwind from the main affected areas of the two counties), a total of about 91,000 acres (Washington State Department of Agriculture 1993).

We estimate the cost of wind-blown dust to the motoring public on a per-acre basis as:

$$C = \frac{\{\Sigma P_{PM} \cdot [\text{Coll} \cdot \text{Cost1} + \text{Inj} \cdot \text{Cost2}]\} + \text{Cost3} + \text{Miles} * \text{VEH} * \text{Hours} * \text{Cost4}}{\text{Acres}}$$

where C is the cost of windblown dust to motoring public per acre of farm land, P_{PM} is the probability of an incident with PM_{10} count exceeding 150 μg/m^3, Coll is the number of excess collisions per incident, Cost1 is the average property cost per collision, Inj is the number of injuries per incident, Cost2 is the average medical cost per injury, Cost3 is the average annual administrative costs for road closures (\$45,000), Hours is the average road closure hours per year (161), Miles is the average miles of road closed per closure (308), VEH is the average instantaneous traffic loading per mile of closed road (estimated 2 vehicles/mile), Cost4 is the average road closure cost per vehicle hour (estimated \$23/h), and Acres is the acres of land contributing to dust loading.

The probability of an incident where PM_{10} exceeds 150 μg/m^3 (our measure of heavy dust loading) was assumed to be the average incidence over the three-year period reported in Table 2 for which we have records. The 11 incidents shown in Table 2 occurred over three years. Based on this experience, the probability of PM_{10} exceeding 150 μg/m^3 is about 0.0133, for an expected number of 3.66 days/yr. Using the cost equation above, the average annual cost to traffic from occasional blowing dust incidents was about \$50/acre/yr.

Health effects. Analysis of acute respiratory effects of blowing dust in Benton–Franklin counties was undertaken by Hefflin and others (1994). Based on their findings that the cost impact of excess emergency room (ER) visits resulting from blowing dust was low (about 56 excess ER cases per event costing about \$200 per case) and that the probability of blowing-dust events was an average of only 3.66 events per year, we calculated

[2]Informal checks with WDOT gave us average vehicle loadings on the rural state highways of up to 11 vehicles per mile, with other rural roads much lower (about 2 vehicles per mile on rural state highways and 0.3 per mile on county roads). The \$23 figure comes from an average salary of \$10.00/hr/adult occupant of a passenger vehicle, 1.3 adults delayed per passenger vehicle delayed, and a \$50.00/hr cost per vehicle for large commercial vehicles, consistent with WDOT practice. Area rural vehicle counts are about 75%–80% passenger vehicles and small commercial vehicles and the rest are large commercial vehicles over 10,000 lb.

acute respiratory effects to be worth only \$0.45 / acre / yr. Other health costs, if any, were not available.

Household cleaning costs. A number of studies have been done over the years that have attempted to relate cleaning and other household costs to particulate pollution as a special case of ''defensive'' or ''averting'' expenditures that prevent or counteract the adverse effect of pollution (Cummings and others 1981, Courant and Porter 1981, Shibata and Winrich 1983, Harford 1984, Harrington and Portney 1987, Bartik 1988). A few of these have directly involved offsite costs of soil eroded from agricultural lands and rangelands. For example, Piper (1989) has calculated offsite wind erosion costs for the western United States, while van Kooten and Thiessen (1995) have estimated direct costs on households from blowing dust in Revelstoke, British Columbia.

In the current study, since no local cost estimates were available, we have estimated these cleaning costs through benefit-transfer methods (Brookshire and Neil 1992). The literature has identified specific protocols to be followed in the process of applying benefit-transfer methods (Boyle and Bergstrom 1992, Kask and Shogren 1994) for transfer of study results (especially willingness-to-pay values) from sites at which studies were conducted (study sites) to sites where values were needed for policy analysis (policy sites). Two important requirements are that the nonmarket commodity under consideration at the original study site must be closely similar to the nonmarket commodity at the policy site and that the site and sample characteristics of the two sites must be comparable. The first consideration leads us to set aside the body of literature that applies to air pollution other than soil erosion. For example, we did not consider studies such as those of Manuel and others (1982) or RER (1991), which focused on other sources and types of increase in suspended particulates in the atmosphere and which had other associated chemical properties such as high levels of acidity from oxides of sulfur. The second consideration led us to prefer studies from the western United States, focused on small towns and rural areas. Finally, the assignment of property rights at both sites must lead to the same theoretically appropriate welfare measure (i.e., willingness to pay or willingness to accept).

The most relevant study that we could locate was the 1986 Huszar and Piper survey-based study in New Mexico that determined annual offsite household cleaning costs associated with wind erosion of agricultural land and rangelands (Huszar 1989, Huszar and Piper 1986). The survey provided data on total annual household cleaning expenses to the wind-erosion rate (tons per acre) in the eight major land resource areas of the state, household income level, and variables affecting attitudes toward cleanliness such as ownership of the property, and the number of years the family has lived at the present residence. From these, Huszar (1989) estimated a series of household cleaning cost functions. Huszar considered the following specification to represent the best formulation, having the highest significance values for the variables of interest and best overall explanatory power (Student *t* values in parentheses).

$$\ln TC = 2.98$$
$$\underset{(7.514)}{-\,0.14X^{-2}} \; \underset{(2.011)}{-\,0.09Y^{2}} \; \underset{(2.251)}{+\,0.01Y^{3}} \; \underset{(5.912)}{+\,2.40Z} \; \underset{(2.578)}{-\,0.02W} \quad (4)$$
$$R^2 = 0.310 \qquad df = 236$$

where TC is household costs from blowing dust, X is wind-erosion rate (tons per acre), Y is household income level (1–8), Z is own (2) or rent (1), and W is years at present residence.

Using Huszar's cost function, we conducted a sensitivity analysis of household cleaning costs to changes in the wind-erosion rate. Wind erosion is dependent on a variety of factors, most notably soil moisture, soil type, ground cover, and wind velocity and direction.[3] For example, in the New Mexico study average annual wind erosion rates varied from 0.2 tons / acre to 6.4 tons / acre / yr (Huszar and Piper 1986). Pristine shrub–steppe lands in the Benton–Franklin county area of Washington State have an annual erosion rate between only 0.5 and 1 ton / acre (Holmes 1994), which was used as a lower bound of household cleaning costs. In contrast, although higher values have been found, cropland in the two-county area experiences an average annual wind-erosion rate of 10–15 tons / acre (Holmes 1994). Fifteen tons per acre was used as the upper bound estimate in our comparison of willingness to pay. These values bracket the averages used in the New Mexico work.

Fitting Benton–Franklin county demographic and ecological information into the Huszar damage function at various wind erosion rates yields annual offsite household costs (mainly protection costs) due to wind erosion ranging from \$423 / household (\$154 / person) at 0.5 tons / acre to \$740 / household (\$270 / person) at 15 tons / acre in 1990 dollars. Household cleaning and automotive maintenance costs probably represent 85% of this cost: \$131 / person at 0.5 tons / acre and \$230 /

[3]Huszar did not deal with wind direction. Here, since most of the population in the two counties is downwind from most of the bare ground in Benton County but upwind from most of the bare ground in Franklin County, we assume that Benton County fields are the primary source of windblown dust experienced by most of the population.

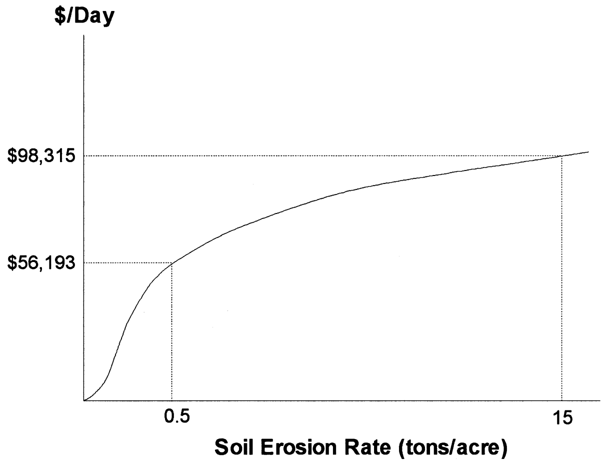

Figure 5. Estimated household cleaning costs associated with soil erosion in Benton and Franklin counties, Washington.

person at 15 tons/acre.[4] The difference between the two values represents the impact of land clearing on household maintenance costs in Benton and Franklin counties of about $99/person/yr. This is comparable to the mean extra annual exterior (painting, cleaning, and landscaping) and interior cleaning costs and automotive maintenance incurred by households in van Kooten and Thiessen (1995) of $473/household or $155/person (1990 dollars), while the upper value is less than the $517/person (1990 dollars) experienced in the dustiest locations (MLRA 77) in New Mexico (Huszar and Piper 1986). Figure 5 illustrates the cost per day for lower soil erosion in Benton and Franklin counties, where the inverse of the soil-erosion rate represents the good being valued. At the average soil-erosion rate of 15 tons/acre on cropland in Benton and Franklin counties, the total cost in the two counties is $94,059/day. Comparatively, the total cost is $53,761/day if the soil-erosion rate is 0.5 tons/acre, that which is found on natural shrub–steppe. The difference between the two ($40,298) is an estimate of the shrub–steppe's value in reducing the daily household cleaning costs associated with offsite windblown dust. Multiplying by days of the year and dividing by the 91,000+ dryland farming acres assumed to be the major source of soil erosion, yields approximately $162/acre/yr in 1990 dollars.

[4]Because the New Mexico household data included estimated costs for health care and recreation, this equation also includes these costs. In the New Mexico data, health and recreation costs vary from 0% to 13.7% of the total, with higher values associated with dustier locations (Huszar and Piper 1986). As a crude adjustment, we suggest that the stated costs in the current study are probably biased upward by about 15% and need to be deflated by that much to reflect only household cleaning and automotive maintenance costs. The equation itself is subject to some uncertainty.

Contingent value of visibility associated with windblown dust. Contingent market valuation can be used to estimate willingness to pay for decreased dust, or, alternatively, what people would be willing to accept as compensation for increased dust. This technique can be used in particular to estimate the aesthetic value of visibility, apart from effects on household maintenance, health effects, traffic safety, etc. Rowe and others (1980) used contingent valuation to estimate the value of visibility in the Four Corners area. Using their contingent valuation estimates, we developed a benefit-transfer model for Benton–Franklin counties. In applying the contingent value-based parameter estimates to the Benton–Franklin counties region, we accounted for a variety of socioeconomic and demographic site characteristics and obtained benefit-transfer estimates of households' willingness to pay for improved visibility [i.e., reduction in airborne dust (PM_{10} levels) and elimination of household cleaning expenditures].

Recognizing the benefit-transfer issues discussed above, we note that our commodity specification (quality of visibility) is similar to the one evaluated by Rowe and others (1980).[5] To capture the aesthetic realities of our study site, air quality was measured using daily observations of PM_{10} over the period 1990–1994 provided by the Benton–Franklin County Clean Air Authority. The site and sample characteristics of the population were adjusted using Benton–Franklin county census data on the urban/rural population, the age distribution, ethnicity, gender, and levels of household income.

Having made the above adjustments, we found the collective willingness-to-pay across 54,000 households in the Benton–Franklin area of approximately $364,400 per exceedence day (i.e., a day on which PM_{10} levels equal or exceed 150 $\mu g/m^3$, the safe minimum standards set by EPA under NAAQS and our measure of reduced visibility; see Figure 5). Assuming that the probability of an exceedence day results in an expected 3.66 incidents per year and that 91,000 acres are responsible, the annual aggregate willingness-to-pay is only $14.66/acre/yr or $24.70/household, far less than the traffic safety costs or cleaning costs calculated above. For comparison, this value is about half the value calculated by van Kooten and Thiessen (1995) for WTP to eliminate the blowing dust problem at Revelstoke, British Columbia of $4.78/month or $57.36/household/yr. In view of the fact that the $57.36 was to eliminate all blowing dust at Revelstoke (not just reduce dust loading), the relative values seem plausible.

[5]Visibility was the focus in the Four Corners study, while in Benton and Franklin counties we are concerned with several aspects of dustiness—difficulty in outdoor recreation and minor morbidity effects, for example.

Annual value of soil stabilization function. For Benton and Franklin counties we estimate that the annual traffic costs, identifiable respiratory health costs, and extra cleaning costs associated with wind erosion of soil and exceedence of EPA's PM_{10} standards are \$50 / acre, \$0.45 / acre, and \$168 / acre, respectively. These benefits do not overlap and can be summed to yield \$218 / acre / yr. Aesthetic values are more problematic. Based on benefit transfer studies, households in the area might be willing to pay up to \$14–15 / acre if visibility were improved so that the EPA standards were never violated. Ideally, we would like to be able to add visibility values to the other three sources of value. However, because of the manner in which it is measured and uncertainties in transfer techniques, it is not clear to what extent this contingent value for visibility overlaps with traffic safety, health, and household maintenance values or is a separate component of value.[6]

Point of comparison: Cost of restoring an ecological function. Cost-based methods can be used as points of comparison to the values derived above. The first is the cost of restoring an ecological function, which may provide some insight into the collective willingness-to-pay for such functions. Recall the difference between soil erosion rates on farmed and undisturbed land. We used 15 tons / acre as a wind erosion rate for cropped land (although wind erosion may be as high as 67 tons / acre / yr on bare land in Benton–Franklin counties; Holmes 1994). In contrast, wind erosion loss on undisturbed shrub–steppe lands is estimated to be between 0.5 and 1 ton / acre / yr (Holmes 1994). The costs of reducing the differential between the soil-erosion levels of 15 and 0.5 tons / acre / yr may be estimated by examining the administration of the Conservation Reserve Program of the Soil Conservation Service of the US Department of Agriculture, and the use of various human-engineered soil stabilization technologies.

Under the Conservation Reserve Program, 123,720 acres of farmland were taken out of crop production in Benton and Franklin counties in 1994. Once these lands are out of production, various engineering systems have been used to stabilize loose soils, including planting vegetative strips, or erecting windbreaks. Advocates of the Conservation Reserve Program suggest that the enrollment of lands subject to wind erosion (i.e., nominal farmlands), provide social value by reducing the severity of dust storms and PM_{10} levels. At the same time, Conservation Reserve Program proponents maintain that nominal farmland is transformed into prime habitat for deer, pheasant, and other wildlife.

The Franklin County program presently enrolls 84,000 acres costing \$4 million per year in annual payments to farmers, while the Benton County program presently enrolls 39,720 acres at a cost of \$1.8 million per year. The Conservation Reserve Program payments can be interpreted as the marginal value to farmers who forgo crop production on the land. This amounts to an annual marginal value of approximately

$$\frac{(\$1.8 + \$4.0) \text{ million per year}}{(84{,}000 + 39{,}720) \text{ acres of land enrolled}} \tag{5}$$

or \$47 / acre / yr. This marginal value is approximately at the midpoint between the annualized value of dryland farms at \$12 / acre / yr and the annualized value of irrigated farmland at \$74 / acre / yr. In perpetuity, the marginal value of an acre of Conservation Reserve Program land is \$940 using a discount rate of 5%. Economically, \$47 / acre / yr (or \$940 / acre) represents the marginal value of land that is set aside to reduce soil-erosion losses (annually or in perpetuity).

A second point of comparison: Costs of soil stabilization techniques on farmed land. A second method of estimating the marginal value of soil stabilization is to consider the direct costs of employing soil stabilization technologies (e.g., windbreaks and vegetative strips on highly erodible soils) to reduce blowing dust while still farming the land. The direct costs of employing these technologies are sensitive both to crop type and rotation cycle; due in large measure to the wide differences in wind erosion for the various crops types and rotation cycles. For instance, in the case of continuous potato farming, vegetative strips cost \$21.65 / acre / yr / rotation and windbreaks cost \$6.15. In contrast, these same preventive measures cost only \$10.83 for vegetative strips and \$6.15 for windbreaks if winter wheat is planted following the potato harvest. However, none of the systems reduces emissions to the 0.5 tons / acre characteristic of native shrub–steppe because all reported systems leave some bare soil.

Retiring bare land from production through the Conservation Reserve Program (and reducing erosion rates to those approaching native shrub–steppe) costs an average of \$47 / acre / yr, while keeping the land in farms and planting vegetative strips and windbreaks, although less effective, costs between \$6 / acre / yr and \$22 / acre / yr. Thus, it appears that more land could be retired from farming and returned to continuous vegetation with a net economic benefit, even counting only off-farm soil erosion benefits. However, the benefit calculation compared with them is sensitive to the

[6]Assuming that the benefit-transfer value is accurate and represents only visibility, it should be looked at as a separate component of value. However, it may capture other components of value closely related to visibility.

estimate of the acreage responsible for wind-blown dust. For example, had we used the total acreage in both Benton and Franklin counties that is planted in row crops (318,000 acres) rather than just 91,000 acres as our denominator for the traffic, health, and cleaning costs, the per-acre value of these costs would have shrunk from $220/acre/yr to about $63/acre/yr—still a good deal, but a much closer call.

Recreational value of hunting on open space. Shrub–steppe land in the study region provides open space for a variety of recreational uses, including upland game hunting. Recreation is produced jointly with other preservation values, but is not precluded by some types of farming. Our estimate of this element of preservation value is based on the costs of replacing shrub–steppe hunting sites using human analogs (game ranches) and on an estimate derived from the application of travel cost methods to shrub–steppe sites.

Set-aside hunting sites. We estimate replacement costs of open shrub–steppe land for hunting from the 1994 sale of the undeveloped Barker Ranch site in the Benton–Franklin county area. The Barker Ranch encompasses 2000 acres in the Horn Rapids area along the Yakima River within the Benton County limits. Private hunting clubs have used the property for years, and the new owners state that, "the property will remain in its current state for some time as a wildlife area for sportsmen." The property is populated with waterfowl, upland game birds, and small animals. The sale price of the Barker Ranch may be viewed as the implicit pure WTP rental price that some hunters would be willing to pay for game habitat to use after all other costs of hunting are paid. Based on the $3 million sale value of the Barker Ranch, the replacement costs for shrub–steppe habitat used for upland game hunting are approximately $1500/acre. Annualized at a discount rate of 5% in perpetuity, this is approximately $75/acre/yr, or about the price of irrigated land in the area. Because of the exclusive nature of the arrangement, this probably can be viewed as an upper-bound value on willingness to pay for hunting. It appears to be unrelated to the value of the land for soil stabilization.

Travel cost-based estimate. We also derived a travel-cost-based estimate of willingness to pay for game hunting on shrub–steppe sites, based on US Department of Fish and Wildlife information about the number of upland bird hunters who use Benton–Franklin counties, and information on the number of pheasant and quail harvested in the area (Washington State Department of Fish and Wildlife). Statewide, 54% of the days spent hunting small game were used in pursuit of pheasant and quail. Multiplying this ratio by expenditures for food, lodging, transportation, and equipment when hunting small game yields estimated expenditures of over $11 million for hunting the two upland game birds in Washington. Approximately 12% of the annual statewide pheasant and quail harvests are taken in Benton and Franklin counties (US Fish and Wildlife Service and US Bureau of the Census 1993). Prorating statewide expenditures for hunting pheasant and quail by this proportion yields an estimate of over $1.3 million spent annually in the two-county region.

Using these data we derived an estimate of the value of shrub–steppe habitat for hunting upland birds in Benton–Franklin counties based on travel costs to these sites. The travel-cost method assumes that willingness to pay for a recreational experience at a site can be inferred from the number of site visitations and travel costs to the site, where the per capita number of visitations is inversely proportional to the travel cost (distance) (Clawson and Knetsch 1966). Travel-cost data were obtained from the Washington State Department of Fish and Wildlife (Upland Game Division) and the National 1991 Survey of Fishing, Hunting, and Wildlife Associated Recreation (US Fish and Wildlife Service and US Bureau of the Census, 1993). We apportioned travel by upland bird hunters to Benton–Franklin counties into five zones based on state averages: those hunters that traveled less than 25 miles to their hunting site, between 25 and 50 miles, between 50 and 100 miles, between 100 and 250 miles, and over 250 miles.

The average cost for hunting per small game hunter in Washington state was $193 in 1991. Assuming that this cost varies in proportion to distance traveled, we estimated cost of hunting in Benton–Franklin counties by distance traveled. We estimated an average cost per zone by multiplying the average cost for the state for small game hunting (AC = $193) by the ratio of median distance in each zone (d_i) for hunters using Benton and Franklin counties to the average distance travelled (d) for the state for small game hunting, i.e.,

$$AC_i = \left(\frac{d_i}{\overline{d}}\right) \cdot \overline{AC} \tag{6}$$

where i is 1–5 and represents zones.

Willingness to pay for hunting shrub–steppe-dependent game birds was then calculated for each zone by multiplying the difference between the highest average cost (over 250 miles—the choke price) and the average cost of the zone being considered by the number of hunters in that zone. It appears likely that this is a lower

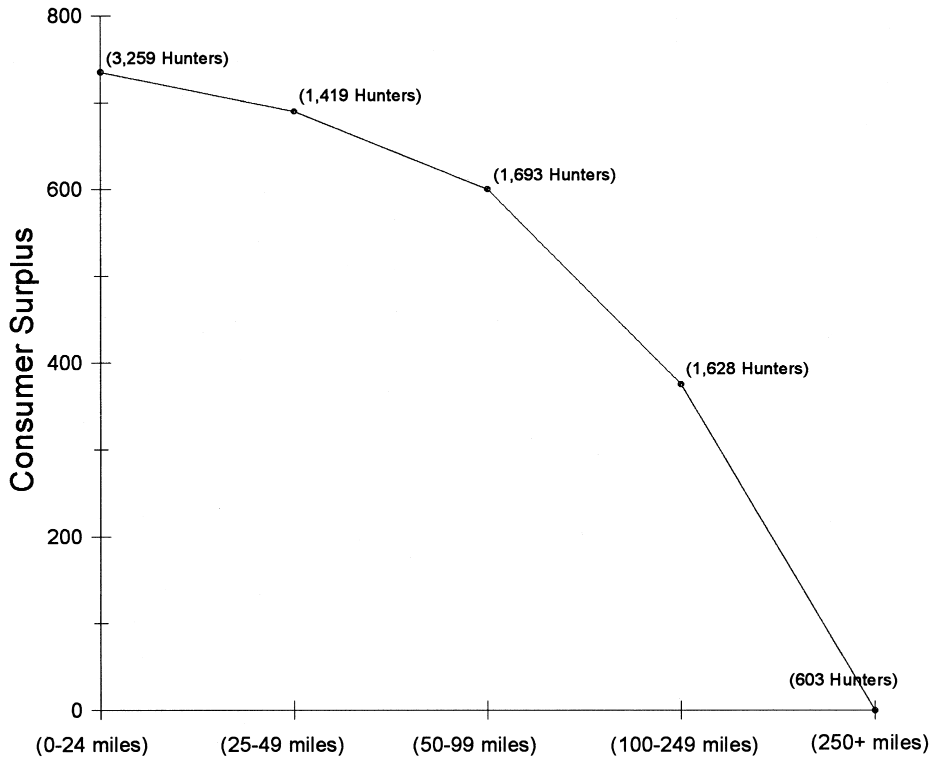

Figure 6. Aggregate willingness to pay for hunting upland game birds in Benton and Franklin counties, Washington (travel-cost method).

bound value of willingness to pay.[7] Willingness to pay was aggregated across zones, yielding $3.2 million in annual recreational benefits (Figure 6). Assuming a 5% discount rate, the capitalized value is $64.2 million.

Benton-Franklin counties contain an upper-bound estimate of 565,498 acres of land available for upland game bird hunting (Washington State Department of Fish and Wildlife 1994). Dividing the capitalized value by the estimate of upland bird habitat acreage yields a capitalized value of $113.56/acre or about $6/acre/yr on an annual basis. This value seems low in comparison with the $75/acre/yr paid for the Barker Ranch, until one realizes that the Barker Ranch is probably better-than-average habitat and the hunters involved in the purchase of the Barker Ranch have the options of improving the habitat, increasing the population of game birds, and restricting access, all of which increase the quality of the hunting experience.

[7]In effect, we assumed that cost per hunter for all small-game hunting destinations is the same for all hunters in each zone and varies only with the distance traveled and not with the number of trips per hunter, duration of trips, or number and type of substitute activities available. If nearby hunters take more frequent trips, then distance overestimates the relative willingness to pay. Furthermore, any fixed-cost component to hunting each year will reduce the premium on hunting in Benton and Franklin counties, since only the variable cost component varies with distance.

Biodiversity. Valuing the biodiversity function of shrub–steppe habitat was our greatest challenge. We base our value on a human analog, the costs of reestablishing ground cover with natural vegetation as an essential step to establishing habitat. Typical costs for revegetation of the shrub–steppe habitat are about $1500/acre in the Benton–Franklin county areas, for an annual value of about $75/acre/yr. A lower-bound value for productivity of restored areas is found in Table 3, because old-farm sites at the FEALE Reserve that were allowed to naturally recover are dominated by alien species (weeds) and are only about 0.7 times as biologically productive as native, undisturbed sites. Deliberate reseeding would likely be more effective at reestablishing some native species than was observed at the FEALE Reserve, but even if ground cover were reestablished by reseeding with native grasses, it appears that the resulting human analog acres could be less efficient than undisturbed systems at providing the biodiversity function that humans value, implying that marginal restoration costs are not necessarily a good proxy for marginal benefits of undisturbed (preserved) shrub–steppe habitat.

This difference in productivity between disturbed and undisturbed shrub–steppe habitat has two possible interpretations for valuation of the undisturbed habitat. First, one could assume that the value at the margin of an acre of undisturbed land is greater than the cost of restoration because the undisturbed land is more valuable in producing biodiversity. The cost of restoring natural habitat could then be adjusted upward to obtain the marginal value of undisturbed habitat. A simple linear adjustment based on the comparative old field results for shrub–steppe at the FEALE Reserve yields a relative marginal value of undisturbed land at \$75/0.7 (worst case) or \$75/1.0 (perfect restoration) yielding between \$75 and \$107/acre/yr. This would appear to be a reasonable interpretation if land managers were prohibiting any further development of shrub–steppe habitat under their control and at the same time were attempting to restore habitat by reseeding.

The second interpretation of human analog restoration costs is that the land management agency has a choice between preservation of undisturbed habitat and restoration of disturbed habitat as competing methods of obtaining biological diversity. Typically, some undisturbed habitat is being developed. It is not rational for a land management agency to restore biodiversity at \$75/acre/yr if it can obtain a better quality habitat less expensively by simply preserving undisturbed habitat for free (i.e., with no cost other than its opportunity cost).[8] The conclusion in this case is that if land is being newly developed and restored at the same time in the same general area, then the land coming under development must be worth less for its ecological services than the land being restored for this purpose. The value of the free ecological service on the newly developed land can be estimated by performing a downward adjustment of the restoration costs. In this case, with a linear adjustment, it is (\$75 $\times$ 0.7) to (\$75 $\times$ 1.0) = \$52 to \$75/acre/yr. Given a choice of methods, the land-use agency should equate the marginal values of restored and undisturbed habitat, which should both be \$52 to \$75/acre/yr. The implicit threshold value for the biodiversity function alone is \$52 to \$75/acre/yr at current habitat abundance, because restored land provides the other principal functions of the native habitat (e.g., soil stabilization and recreation) about as well as native habitat does. Thus, of all the functions discussed, only the biodiversity function is involved in the choice between native habitat and exotic grasses. If no land is being restored, then no inference appears to be possible from restoration costs concerning marginal habitat values of existing undisturbed lands in the immediate vicinity, although it does suggest a marginal habitat value for lands that could be developed.

[8]The acquisition cost is the same in both cases—with dryland farms as the alternative opportunity, the agency pays the \$12.40/acre/yr average price to acquire the land for restoration; with preservation, it forgoes the \$12.40/acre/yr that it could earn by selling the land.

Table 5. Summary of selected values estimated for Shrub-Steppe Habitat (dollars/acre/year)

	Measurement technique	Annual value per acre (\$)
Function		
Soil stabilization	Contingent valuation: benefits transfer to reduce PM_{10} count	4–14
	Cost of Conservation Reserve Program land acquisition program	47
	Cost of soil stabilization program with farming (analog)	6–21
	Expected cost of traffic accidents and road closures	15–50
	Extra cleaning and maintenance costs	48–169
Recreation	Hunting club annualized rental value (analog)	75
	Travel costs (WTP)	6
Species diversity	Annualized restoration costs, adjusted for productivity	52–75
Opportunity costs		
Grazing	Annualized value of grazing land	3.35
Farming	Annualized value of farmland (dry)	12.40
	Annualized value of farmland (irrigated)	74.20
Urban	Annualized value of building sites	460.40

Summary and Conclusions

Shrub–steppe ecosystems provide functions and services of considerable direct and indirect value to humans. Our estimated annual per-acre values of shrub–steppe habitat are summarized in Table 5 (for both 91,000 acres and 318,000 acres of farmland in the case of wind erosion because it is not clear how many acres contribute to the blowing dust problem). Most importantly, we found that the value of the soil stabilization function or biodiversity functions alone could outweigh dryland farming at the margin, if land markets expressed (for example) the preferences of people who bear the costs of traffic hazards and perform extra household cleaning as a result of blowing dust or of people who value species diversity.

The highest values shown in Table 5 are for the soil stabilization function, regardless of how we measure the benefits of shrub–steppe. Because they leave erodible bare earth, farmed land, overgrazed land, and urban building sites (temporarily), appear to cause the offsite damage that accounts for the bulk of the shrub–steppe value. Attention to mitigation and maintaining ground cover to prevent blowing dust in inhabited regions could reduce that source of offsite damage. Similarly, grazed and farmed areas could be managed to maintain some important recreation values. However, the remaining biodiversity value might be as much as $52 to $75/yr/acre under present circumstances. That value cannot be changed through mitigation.

We expect that many of the values in Table 5 would not show up in a conventional contingent valuation study because it would be extraordinarily difficult to provide respondents with enough technical ecological knowledge to understand the underlying processes, functions, and services discussed in this paper. Instead, a contingent valuation study would likely provide an estimate largely based on aesthetics (visibility), with only weak links to the underlying implications for land cover and with no consideration at all of biodiversity issues.

We have demonstrated that with careful and imaginative use, environmental and economic data can be combined to estimate the values of environmental functions and services of even the least appreciated ecological resources. The broader implication of the values in Table 5 is that undisturbed natural habitats have values worth preserving, whether or not conventional analysis can discover them. Furthermore, whenever development of undisturbed habitat is proposed, the processes and functions of the natural habitat should be investigated and attention should be given to low-cost ways to preserve those functions and services even when they have no easily recognized market value. Future work on valuation should therefore be focused on those aspects of value that cannot be preserved when the land is converted to other uses. For land that is already converted, future work should be focused on those aspects of value that can be mitigated through better management of vegetation and land cover.

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