CHAPTER

9

A Social and Economic Assessment of Rangeland Conservation Practices

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INTRODUCTION

Rangelands provide a wide variety of ecosystem goods and services, and the conservation practices implemented on them produce a variety of direct and indirect economic and social effects. Basic ecological relationships and varying degrees of natural resource management determine the magnitude and quality of goods and services produced. Society determines what the relative values of these goods and services are at any particular location and time (Fox et al. 2009).

In this chapter, we examine the literature related to the economic and social aspects of ecosystem services impacted by the conservation practices of the Natural Resources Conservation Service (NRCS) of prescribed grazing, prescribed burning, brush management, upland wildlife habitat, riparian management, and range planting. In addition, we examine the social and economic aspects of invasive species management that cross different conservation practices. At the time of this synthesis, invasive species management was not a specific conservation practice, but the NRCS recently created a new conservation practice titled Herbaceous Weed Management that is evaluated in a separate chapter of this document. Valuation of ecosystem goods and services potentially impacted by the specified conservation practices, particularly those services for which markets do not exist, is also examined. Understanding valuation methodology is important in evaluating conservation practice implementation and funding decisions. In some cases, the nonmarket valued ecosystem goods and services are those most valued by society.

The reason for estimating some measure of value for ecosystem goods and services is that landowners and managers need to evaluate trade-offs for decision making (e.g., Maguire and Justus 2008; Nelson et al. 2009). One way to make this evaluation workable is to put all the resources in the same units, and the assignment of monetary value is one way to accomplish this. However, the concept that an ecosystem good (e.g., an endangered species) or service has an intrinsic value that is "priceless" or "infinite" does not serve decision makers well when choices have to be made. The failure to include social and economic nonmarket values in decision-making processes will likely lead to undervaluing the net benefits and lead to inefficient allocations of our nation's investments in conservation.

The NRCS has recognized that ecosystem goods and services are directly and indirectly affected by the various conservation practices that they implement on rangelands. In the description of each conservation practice, the purposes describe the expected benefits or outcomes of practice implementation. Additionally, for each conservation practice, a physical effects worksheet is published that more specifically describes the benefits and outcomes. Both of these are on the electronic Field Office Technical Guide (http://www.nrcs.usda.gov/technical/efotg) sections of the NRCS website. This is shown in the descriptions of conservation practices and in the economic analysis of benefits and costs. In examining the conservation practice descriptions, there are a variety of different ecosystem goods and services listed as being positively or negatively impacted by the different practices. Table 1 shows a list of potential goods and services that can come from rangelands as currently recognized by the NRCS. As shown, there are many facets of each general good or service, each of which can have its own effect on the quality

Western Juniper expansion, Dufur Wildlife Management Area, Oregon. (Photo: John Tanaka)

TABLE 1. NRCS estimated impacts of different conservation practices on different ecosystem goods and services. 0 = not applicable, 1 = neutral, 2 = slight impact, 3 = moderate impact, and 4 = substantial impact. Parentheses indicate a negative impact. Adapted from NRCS Physical Effects Worksheets for each conservation practice (available at http://nrcs.usda.gov/technical/efotg as of March 2008).

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
Soil—erosion						
Sheet and rill	2–4	2–4	3–4	3–4	3	2–3
Wind	2–4	2–3	3–4	3–4	3	2–3
Ephemeral gully	2–4	2–4	3–4	3–4	3	2
Classic gully	2–4	2	2–3	2–3	2–3	0
Stream bank	2	2	2–4	2–3	2	3–4
Shoreline	2	2	2–4	2–3	2	3–4
Irrigation induced	0	0	2–3	0	0	0
Mass movement	(2–3)	0	1	1	2	2
Road, roadsides, and construction sites	0	0	0	0	0	0
Soil—condition						
Organic matter depletion	2–4	2	3–4	3–4	1	3–4
Rangeland site stability	2–4	2–3	3–4	3–4	0	2–4
Compaction	(2)	0	2–4	3–4	0	3–4
Subsidence	0	0	0	2	0	0
Contaminants						
• Salts and other chemicals	2–3	(2)	2–3	2	0	2–3
• Animal waste and other organics—N	0	2	2–3	2–3	0	2–4
• Animal waste and other organics—P	0	2	2–3	2–3	0	2–4
• Animal waste and other organics—K	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—N	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—P	0	2	2–3	2–3	0	2–4
• Commercial fertilizer—K	0	2	2–3	2–3	0	2–4
• Residual pesticides	(2)	2	2–3	2–3	0	2–3
Damage from sediment deposition	2–3	2	3	2–4	0	2–3
Water—quantity						
Rangeland hydrologic cycle	2–4	2–4	2–4	3–4	0	2–4
Excessive seepage	(2–3)	0	1	1	0	2–3
Excessive runoff, flooding, or ponding	2–3	2	2–3	2–3	(3)	(3)
Excessive subsurface water	(2–3)	0	2	1	2–3	2–3
Drifted snow	0	0	0	0	0	0
Inadequate outlets	2	2	2	2	0	0
Inefficient water use on irrigated land	0	0	(2)	0	0	0
Inefficient water use on nonirrigated land	2–3	0	2–3	2–4	0	0
Reduced capacity of conveyances by sediment deposition	2–4	2	2–3	2–4	2	2–3
Reduced storage of water bodies by sediment accumulation	2–4	2	2	2–3	2	4
Aquifer overdraft	2	0	1	1	0	0
Insufficient flows in water courses	2–4	0	1	2–3	2	2–3
Water—quality						
In groundwater						
• Harmful levels of pesticides	(2)	0	2–3	2–3	0	2–3
• Excessive nutrients and organics	0	2	2	2–3	0	4
• Excessive salinity	(2)	0	2	2	0	4
• Harmful levels of heavy metals	1	0	2	2	0	2
• Harmful levels of pathogens	1	0	2	2	0	2–3
• Harmful levels of petroleum	1	0	0	0	0	0
In surface water						
• Harmful levels of pesticides	(2)	0	2–3	2–3	0	2–3
• Excessive nutrients and organics	1	2–3	2	2	0	4
• Excessive suspended sediment and turbidity	2–4	2	2–4	2–4	2–3	3–4

TABLE 1. continued.

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
• Excessive salinity	2	0	2–3	2	0	2
• Harmful levels of heavy metals	2	2	2	2–3	0	2–3
• Harmful temperatures	1	1	1	1	1	2–3
• Harmful levels of pathogens	0	0	2	2	0	3
• Harmful levels of petroleum	0	0	0	0	0	2
Air—quality						
Particulate matter less than 10 μm in diameter (PM 10)	0	2–3	2–3	2–3	2–3	2
Particulate matter less than 2.5 μm in diameter (PM 2.5)	0	2–3	2–3	2–3	2–3	2
Excessive ozone	Neutral	1	1	1	1	1
Excessive greenhouse gas						
• CO_2 (carbon dioxide)	0	3–4	2–3	2–3	2	2–3
• N_20 (nitrous oxide)	0	0	0	0	0	0
• CH_4 (methane)	0	(3)	0	0	0	0
Ammonia (NH_3)	0	0	0	0	0	0
Chemical drift	(2–3)	0	0	0	0	0
Objectionable odors	0	(2)	1	0	0	0
Reduced visibility	0	1	2–3	2–3	2	0
Undesirable air movement	0	0	0	0	2–3	0
Adverse air temperature	(2–3)	(2–3)	0	0	2–4	1
Plants—suitability						
Plants not adapted or suited	3–4	3–4	3–4	4	3–4	3–4
Plants—condition						
Productivity, health, and vigor	2–4	4	4	4	3–4	4
Threatened or endangered plant species						
• Plant species listed or proposed for listing under the Endangered Species Act	1	1	1	1	1	1
• Declining species, species of concern	1	1	1	1	1	1
Noxious and invasive plants	3–4	3–4	3–4	3–4	3–4	3–4
Forage quality and palatability	3–4	4	3–4	4	3–4	3–4
Wildfire hazard	3–4	4	2–4	0	0	0
Animals—fish and wildlife						
Inadequate food	2–4	2–4	2–4	2–4	4	3–4
Inadequate cover/shelter	2–4	2–4	2–4	2–4	4	3–4
Inadequate water	0	0	0	0	0	2–4
Inadequate space	2–4	3–4	3–4	3–4	4	2–4
Habitat fragmentation	2–4	3–4	3–4	3–4	3–4	2–4
Imbalance among and within populations	2–4	2–4	2–4	2–3	4	2–4
Threatened and endangered fish and wildlife species						
• Fish and wildlife species listed or proposed for listing under the Endangered Species Act	1	1	1	1	3–4	1
• Declining species, species of concern	1	1	1	1	3–4	1
Animals—domestic						
Inadequate quantities and quality of feed and forage	304	4	4	4	2–3	3–4
Inadequate shelter	(2–3)	(2)	2–4	0	0	0
Inadequate stock water	0	0	0	0	0	0
Stress and mortality	2–4	2–3	3–4	2–4	0	3–4
Human—economics						
Land—change in land use	0	0	0	2–4	0	2–4
Land—land in production	0	3	0	4	0	2–4
Capital—change in equipment	3	2	2	2	2	2
Capital—total investment cost	2–4	2	0	3	2	4

TABLE 1. continued.

	Brush management	Prescribed burning	Prescribed grazing	Range planting	Upland wildlife habitat	Riparian
Capital—annual cost	1–3	1	1	0	1	2
Capital—credit and farm program eligibility	Situational	Situational	Situational	Situational	Situational	Situational
Labor—labor	1–3	2–3	2–3	2	2–3	2–3
Labor—change in management level	1–3	2	2	2	Negligible	2
Risk—yield	(3–4)	(2)	(2–3)	0	(2–3)	0
Risk—flexibility	(3–4)	3	2–3	2	2–3	0
Risk—timing	4	4	4	4	0	0
Risk—cash flow	2–4	(2)	(2–3)	2	2	2–3
Profitability—change in profitability	2–3	2–3	2–3	2	(2)	Situational
Human—cultural						
Cultural resources and/or historic properties present or suspected to be present	2–4	2–4	0	2–4	0	(2–4)
Human—energy						
Depletion of fossil fuel resources	No effect	(2–3)	0	0	2	(2–3)
Underutilization of nonfossil energy resources	(2)	2	0	0	0	0

and quantity of the good and service and its relative value to society.

Table 1 shows the expected change in the ecological or societal parameters from six main conservation practices evaluated in this document (invasive plant species management is not included). An examination of Table 1 indicates the ecosystem good or service followed by the expected level of impact from each of the conservation practices. While some goods and services listed are considered "bads," such as soil erosion (services can be either positive or negative, and those with negative outputs are called "bads" as opposed to "goods"; they are the outputs on which humans either place positive or negative values), the numbers indicate whether the conservation practice will minimize (positive numbers) or accelerate (numbers in parentheses) soil erosion. For example, in their critique of the ecological impacts of ranching, Freilich et al. (2003) identified some of the potential benefits that may arise from proper livestock management and also factors that need to be mitigated as those practices are implemented.

To put the values from Table 1 in context, Table 2 shows the number of hectares treated by each of the major conservation practices from 2004 to 2008 by state and rangeland region. Table 3 shows the conservation practice expenditures from all NRCS programs by state from 2005 to 2009. From an economic point of view, the scale of the practices being implemented over this period determines the potential size of the impact. Different states use some practices more than others (Tables 2 and 3) with expected differences in goods and services produced. The differences in expenditures by state and conservation practice (Table 3) may be due to a variety of factors, including local preferences, acceptability, and needs. Expenditures by state will also vary based on the amount of private rangeland and the willingness of those landowners to participate in NRCS programs. Expenditures in conservation practices from 2005 to 2009 (Table 3) in aggregate provide some indication of how practices have been implemented in different states. Table 4 shows the total annual expenditures for the conservation practices from 2005 to 2009. Since NRCS conservation programs are cost-share programs, these expenditures indicate only the government's share of the total investment in conservation. These figures also illustrate that federal expenditures have generally been increasing for these conservation practices over the 5-yr period. The question being asked is, "Do the net societal benefits, including both market and nonmarket values, from these practices offset the known costs?"

In this chapter, we discuss the various ecosystem goods and services impacted by these seven

TABLE 2. Area of the main conservation practices implemented in rangeland states from 2004 to 2008.

Location	Brush management (ha)	Prescribed grazing (ha)	Range planting (ha)	Prescribed burning (ha)	Riparian herbaceous cover (ha)	Upland wildlife habitat management (ha)
National	1 590 489	32 682 716	519 881	619 786	15 781	21 860 411
Rangeland region						
Central	1 046 181	16 780 643	257 038	336 161	8 630	12 185 583
West	410 656	13 815 577	260 263	34 660	3 722	6 980 085
West state						
Alaska	384	1 416 984	0	3	0	184 083
Arizona	34 413	2 607 496	5 228	3 148	59	332 011
California	28 491	588 229	6 924	6 197	511	152 638
Colorado	25 196	2 245 726	88 363	4 119	154	668 710
Hawaii	4 136	23 822	123	1 330	0	57 358
Idaho	8 996	239 791	3 359	1 790	193	180 574
Montana	37	1 046 954	94 561	9 406	31	735 365
Nevada	7 196	59 813	1 733	0	29	7 934
New Mexico	239 599	2 630 730	15 684	6 014	12	3 202 863
Oregon	12 061	290 328	5 006	987	2 619	222 809
Utah	27 561	423 301	37 417	140	3	217 394
Washington	5	65 912	433	103	74	124 209
Wyoming	22 581	2 176 491	1 432	1 421	38	894 138
Total	410 656	13 815 577	260 263	34 660	3 722	6 980 085
Central state						
Kansas	96 700	673 583	85 938	149 027	2	455 409
Nebraska	13 832	1 370 129	58 034	16 558	1 252	186 720
North Dakota	3 306	541 545	4 366	113	3 885	158 965
Oklahoma	170 900	1 247 742	32 143	84 726	645	303 599
South Dakota	1 709	343 747	2 894	302	2 140	158 251
Texas	759 734	12 603 897	73 663	85 436	706	10 922 639
Total	1 046 181	16 780 643	257 038	336 161	8 630	12 185 583

major conservation practices and determine if the peer reviewed literature provides measures of the quantity of change in the ecosystem good or service or merely considers those changes without quantifying them. We then evaluate what the literature indicates about the social effects associated with implementation of the conservation practices, the economic consequences of the conservation practices, and the economic valuation of the ecosystem goods and services. Our intent is not to specify what these values are today because they will change over time. Rather, we seek to define the ecosystem goods and services and their social and economic benefits as well as how those values may be used in decision making.

ECOSYSTEM SERVICES

Ecosystem goods and services are defined as those things or experiences produced by natural systems on which humans place value (Alcamo et al. 2005; Fisher et al. 2009; Fox et al. 2009; Kremen and Ostfeld 2005; Millennium Ecosystem Assessment [MEA] 2005). In this section, we examine the types of ecosystem services that may be increased or decreased from the implementation of conservation practices, identify major relationships among ecosystem goods and services, and describe the primary issues associated with measurement of those relationships.

TABLE 3. All NRCS program funds expended on conservation practices by state, 2005–2009.

Location	Brush management	Prescribed burning	Prescribed grazing	Range planting	Riparian herbaceous buffer	Wildlife upland habitat management	Total
West rangeland state							
Alaska	4 768	—	405 184	—	—	33 208	443 160
Arizona	2 896 930	17 256	4 573 259	326 474	—	65 166	7 879 085
California	3 311 387	15 270	1 045 000	928 945	13 104	92 985	5 406 692
Colorado	1 645 065	4 176	2 405 289	575 649	31	140 504	4 770 714
Hawaii	2 249 800	—	546 932	89 765	—	1 490	2 887 986
Idaho	270 053	4 372	929 023	349 916	2 013	7 199	1 562 577
Montana	83 108	—	3 266 740	722 150	588	4 196	4 076 782
Nevada	758 417	—	176 834	171 984	—	307	1 107 542
New Mexico	18 185 311	68 398	1 333 336	1 084 828	338	351 235	21 023 445
Oregon	1 735 163	97 104	3 195 510	524 326	—	150 912	5 703 015
Utah	1 087 580	2 152	773 954	860 948	525	20 790	2 745 949
Washington	219	—	149 788	145 726	2 264	191 632	489 629
Wyoming	717 663	21 132	2 950 802	48 960	—	67 070	3 805 627
West total	32 945 463	229 860	21 751 651	5 829 671	18 862	1 126 694	61 902 202
Central rangeland state							
Kansas	4 141 308	417 368	11 378 028	299 756	—	1 058	16 237 518
Nebraska	2 281 417	238 001	1 081 426	681 287	311	626 510	4 908 952
North Dakota	70 628	—	1 350 708	189 920	266	341 624	1 953 145
Oklahoma	10 301 158	766 992	1 067 643	691 442	—	64 273	12 891 508
South Dakota	6 664	—	609 149	351 212	—	71 877	1 038 902
Texas	77 507 317	436 684	9 585 612	5 720 478	4 024	44 434	93 298 549
Central total	94 308 492	1 859 045	25 072 567	7 934 094	4 601	1 149 775	130 328 573
Grand total	127 253 955	2 088 905	46 824 218	13 763 765	23 463	2 276 469	192 230 775

There are a variety of conceptual models used to organize and classify various ecosystem goods and services for purposes of informing management decisions (Ruhl 2008; Swinton 2008). The Millennium Ecosystem Assessment (MEA) is one such model (Carpenter et al. 2006) that sorted ecosystem services into provisioning (e.g., food, freshwater, fuel wood, and genetic resources), regulating (e.g., climate regulation, disease regulation, flood regulation, and erosion regulation), cultural (e.g., spiritual/inspirational, recreational, aesthetic, and educational), and supporting (e.g., soil formation, nutrient cycling, and primary production) categories. The conceptual model by Fox et al. (2009) provided the framework by which the ecological systems interact with the social and economic systems and defined the ecosystem goods and services as extractable goods and tangible and intangible services. While the MEA model has gained acceptance in defining these ecosystem goods and services, there seems to be a large amount of double counting that could occur if it were implemented on the ground. From a valuation viewpoint, double counting creates problems in summing up the total effects. The Fox et al. (2009) conceptual model did not seek to define the different ecosystem goods and services but instead made explicit that ecosystem goods and services provide the connection between human systems and their environment and hence their source of value.

Part of the issue with defining ecosystem services associated with implementation of

TABLE 4. Total annual expenditures through all NRCS programs for selected conservation practices in 2005–2009.

Conservation practice	Year				
	2005	2006	2007	2008	2009
West rangeland states					
Brush management	3 747 652	5 634 082	6 052 764	9 013 795	8 497 171
Prescribed burning	3 165	29 918	61 380	83 048	52 350
Prescribed grazing	3 205 190	3 731 971	5 842 649	5 116 415	3 855 425
Range planting	788 901	774 455	1 439 449	1 548 191	1 278 675
Riparian herbaceous buffer	7 882	3 989	2 635	3 430	927
Wildlife upland habitat management	30 415	95 562	210 273	328 152	462 292
West total	7 783 205	10 269 975	13 609 150	16 093 031	14 146 840
Central rangeland states					
Brush management	14 198 925	17 012 119	17 297 158	24 859 881	20 940 409
Prescribed burning	232 389	186 525	258 648	643 463	538 020
Prescribed grazing	1 802 474	3 400 594	8 407 342	5 071 747	6 390 410
Range planting	1 267 242	1 497 869	1 521 917	2 195 160	1 451 906
Riparian herbaceous buffer	64	694	—	3 688	155
Wildlife upland habitat management	123 393	173 081	244 566	365 460	243 274
Central total	17 624 488	22 270 883	27 729 631	33 139 399	29 564 174
Grand total	25 407 693	32 540 858	41 338 781	49 232 430	43 711 014

conservation practices is that there is little research on their production functions (Kremen and Ostfeld 2005). Production functions describe the relationship between the quantities and qualities of inputs used to produce various quantities and qualities of outputs. In addition, there is a need to understand the form of the relationship between different outputs described as the production possibility frontier by economists. For example, a given amount of input (e.g., land) can produce a variety of outputs (e.g., cattle vs. wildlife). Herrick (2000) concluded that we need to demonstrate causal relationships between soil quality and ecosystem functions such as biodiversity and biomass as well as the ecosystem's response to disturbance. Lal (2007) similarly presented arguments that soil science is the key if we are to pursue going to a carbon-based economy and meet the diversity of needs and wants from ecosystems. In any case, the characteristics of many ecosystem goods and services that will make them difficult to assess in decision making include their public good aspects, spatial and temporal dynamics, joint production, complexity of ecosystems, interdependence benefits, and the interactions among these characteristics (Fisher et al. 2009).

There are many pressures on the production of ecosystem services originating from agricultural and natural resource management, including conservation practices, and societal issues, such as urbanization and land fragmentation. Converting land use from the production of typical agricultural products to the production of biomass for energy can also drastically alter the quantity and type of ecosystem services produced on a given land tract. Cook et al. (1991) estimated that the potential exists for 20 million ha of US rangeland to be converted to energy-producing biomass with impacts on wildlife habitat, soil erosion, salinization, groundwater depletion, and subsidence. The study did not, however, quantify the expected changes in those ecosystem services. Higgins et al. (2002) looked at the potential impacts of agricultural practices and development as threats to future waterfowl habitat conservation over time. They concluded that changing economic and policy pressures on farmers and ranchers have the potential to modify management practices to bring marginal land currently in the Conservation Reserve Program (CRP) back into crop production.

In a study to evaluate the effects of western juniper (*Juniperus occidentalis* Hook) control on an eastern Oregon ranch, environmental services were included in the ranch model to evaluate their response to juniper control (Aldrich et al. 2005). While this study did not estimate the production function for environmental services, it did show how erosion and wildlife populations may change in response to the implementation of different management alternatives. Wildlife species (quail, deer, and elk) responded differently because of unique habitat needs as the percent canopy cover of juniper changed from the alternative juniper control practices and the implementation timing. Erosion potential decreased as the trees were removed and cover of grasses and shrubs increased. Other models have been used in similar situations to evaluate tree control to increase forage production (Engle et al. 1996). Unfortunately, the manner in which various responses of ecosystem services might impact landowner decisions was not considered in either study.

A comparison of cattle production with quail and deer habitat in Oklahoma indicated that ranch returns varied based on the amount of wildlife present and the various brush management treatments that were used (e.g., prescribed burning, herbicide applications, and mechanical treatments; Bernardo et al. 1994). Lease hunting showed higher net returns per hectare compared to cattle production when wildlife populations were abundant but with no lease hunting income.

The development of a production possibilities frontier (PPF) requires knowing how each ecosystem good or service responds to common inputs such as vegetation (McCoy 2003). A PPF was developed to compare cattle and antelope in Wyoming (Bastian et al. 1991); however, the PPF did not show a great deal of substitutability because dietary overlap between the two species was minimal. It was concluded that it would take extreme value differences between cattle and antelope for the economically optimal combination of species to be only cattle or only antelope.

A PPF describing the effect of cattle grazing on carbon and nitrogen balance of mixed-grass rangelands was developed comparing light, heavy, and ungrazed pastures (Schuman et al. 1999). Aboveground biomass, carbon, and nitrogen showed a curvilinear (decreasing at an increasing rate) decline with increasing grazing intensity. When considering both above- and belowground biomass, a U-shaped curve was observed where total carbon decreased at an increasing rate and total nitrogen was about linear because of grazing intensity. In these kinds of studies, three treatment levels can begin the process of identifying the PPF curve in order to assess the trade-offs among unique combinations of the two products. More treatment levels will lead to better decision-making capabilities, allowing for more finely defined points to ascertain continuous trade-offs.

The quantity and quality of forage produced is one of the major ecosystem goods that is valued in the market, which makes it relatively easy to value compared to other ecosystem goods and services (Bartlett et al. 2002; Council for Agricultural Science and Technology 1996). However, forage production can have other values beyond that which is placed on it in the marketplace. The market value of forage tends to be heavily weighted toward domestic livestock production, but it can also have other values, such as wildlife feed and habitat, erosion control, and quality of life. These additional values are likely to be captured only through nonmarket valuation methods. It would be necessary to estimate a livestock production value, a wildlife feed and habitat value, an erosion control value, a quality-of-life value, and social benefits and costs if the total economic value for forage is to be estimated (Bartlett et al. 2002).

Implementing conservation practices can have the unintended consequences of reducing some ecosystem goods and services. For example, it is generally viewed that controlling salt cedar (*Tamarix* spp.) is a desirable practice when the objective is to improve riparian habitat for numerous wildlife species. However, as Dudley and DeLoach (2004) pointed out, when an endangered species such as the southwestern willow flycatcher (*Empidonax traillii extimus*) uses salt cedar for habitat, controlling salt cedar presents the problem of an incidental taking of an endangered species even though the native vegetation might eventually provide better habitat. In addition, salt cedar or its control can have impacts on water availability, other

Small acreage owners in Utah County, Utah, maintain livestock that can affect surrounding agricultural land. (Photo: Mark Brunson)

wildlife, aesthetics, and forage availability, resulting in multiple trade-off decisions. Beyond these trade-offs, one of the main reasons to control invasive plant species such as salt cedar is to prevent its eventual spread to other areas. The bottom line for such decisions are both spatial (e.g., water for downstream use vs. wildlife habitat at the point of control) and temporal (spread over time).

In a Texas study of CRP lands, a comparison of targeting only high potential programs based on costs, benefits, or the benefit-to-cost ratio found mixed results for environmental benefits (Babcock et al. 1996). They found that using the proper criterion (benefit to cost) can result in greater environmental benefits, including reduced wind and water erosion, increased surface water quality, or better wildlife habitat, compared to other selection criteria for enrolled lands. They noted that heterogeneity of environmental quality and productivity affects the magnitude of changes in the environmental effects. They were not able to estimate the impact on the production quantity for many ecosystem services because of the lack of ability to quantify physical trade-offs among the ecosystem services and the absence of a social value function to evaluate societal trade-offs.

It may be possible to increase ecosystem services through changes in management options. Integrating crop and livestock systems in Texas was shown to improve nutrient cycling, reduce soil erosion, improve water management, interrupt pest cycles, and spread

Salt Cedar (tamarisk) control, Bosque del Apache National Wildlife Refuge, New Mexico. (Photo: John Tanaka)

economic risk through diversification (Allen et al. 2008). Grazing was shown to increase soil organic carbon and nitrogen contents with light grazing compared to no grazing or heavy grazing (Ganjegunte et al. 2005). Brush management may be a way to increase water yield as well as bird habitat for species that require grasslands (Olenick et al. 2004b).

In Arizona, an estimate was made of the value to home owners from riparian habitat restoration, and it was concluded that the benefits exceeded the costs in this case (Bark-Hodgins and Colby 2006). Although the value is not solely attributable to riparian restoration, it does identify hedonic pricing models (essentially a regression model that uses characteristics to explain differences in price) that relate numerous attributes to the value of the land as one method of estimating the value to property owners of some ecosystem goods and services. A similar hedonic model was used to estimate amenity values for agricultural land in Wyoming. Land that could offer several economic services, including scenic views, elk habitat, sport fishing, and distance to a nearby town, was shown to command a higher selling price (30%) than similar land that did not provide these ecosystem services (Bastian et al. 2002).

The relationship between the type of ecosystem being improved and the management uses of the riparian zone affect how a conservation practice can be implemented to improve riparian vegetation. Quinn et al. (2001) developed a relationship between a riparian zone classification and the potential for riparian zone improvement in ecological health. Implementing practices such as off-stream water development to draw cattle out of riparian areas can have beneficial effects on both livestock and the riparian area (Stillings et al. 2003). Changing the status of riparian habitat was shown to have differential effects on amphibians, reptiles, birds, and mammals (Ekness and Randhir 2007). They found that the higher the degree of disturbance to the riparian area, the greater the negative impact on these four wildlife groups. They concluded that spatial targeting of conservation practices will have the greatest positive effect when targeting headwaters and lower-order watersheds. They also developed a conceptual model to evaluate the role of conservation practices that affect watershed characteristics important to the wildlife groups that they studied. Freemark (1995) developed a spatial–temporal hierarchy to illustrate the scales at which conservation practices and other stressors can affect wildlife in agricultural landscapes. The implication is that at these different scales, conservation practices may have differential impacts on wildlife habitat and populations.

In this section, we have sought to define and identify the kinds of ecosystem services that can be expected to arise from the implementation of conservation practices. The basic premise is that these ecosystem services arise, either intentionally or unintentionally, from the conservation practice and can have either a positive or a negative value. Understanding the relationships and relative values of the ecosystem services is crucial for making investment allocation decisions and to determine whether an investment in a conservation practice is going to be profitable.

ECONOMICS

Prescribed Grazing

The NRCS conservation practice standard for prescribed grazing (US Department of Agriculture [USDA]-NRCS 2007) defines "prescribed grazing" to be the controlled harvest of vegetation with grazing animals, managed with the intent of achieving a specific objective. Sustainability of forage and livestock production are central concerns when designing a grazing strategy, but there is a body of literature dealing specifically with the economics of grazing. Choosing an optimal stocking rate is considered one of the most important grazing management decisions because the stocking rate decision affects vegetation, livestock production, wildlife and economic returns (Holechek et al. 2004). We first describe the economic model typically used to define economically optimal stocking rates and then review the literature dealing with the prescribed grazing conservation practice.

Optimal Stocking Rates. The stocking rate decision is a classic example of the well-known production economic model of profit maximization when defined from the input perspective (Debertin 1986; Workman 1986). The traditional myopic single-year economic model ignores potential interyear grazing impacts and equates the added economic value of an additional grazing animal to the added cost of that animal, a principle commonly known as equating value of marginal product to marginal factor costs (VMP = MFC). With diminishing rates of gain as more animals are added to the pasture, each animal added contributes less to profit than did the previous one, and at the economically optimal stocking rate, the last animal adds nothing to profit.

The conceptual economic model was described over 45 yr ago by Hildreth and Riewe (1963) and has been applied primarily to yearling stocker cattle because of the added complexities of cow–calf production (but see Hart et al. 1988b). Regardless of the animal class, the expected production rate for grazing animals is related to the number of animals grazing a given land area. Based on declining per head performance, a production function is defined that relates the gain per hectare that would be realized at alternative stocking rates. The principle of diminishing returns applies and animal gains will eventually decrease as stocking rate is increased. Total gain per hectare will also eventually fall, but this may occur at a stocking rate that is well beyond what would be detrimental to rangeland condition and future forage production. In this case, rangeland condition and sustainability over time become of key importance, and a dynamic economic model is needed. However, given no major year-to-year interactions, the single-year economically optimal stocking rate will lie somewhere between the relatively low stocking rate that would yield the biggest calf and the relatively high stocking rate that would give the most gain per hectare (Torell et al. 1991).

A search for literature in AGRICOLA with screening on rangelands and the search term "stocking rate" in the title or subject field, with "economics" and "rangelands" in the key words, identified 156 papers, of which no more than about 40 actually did an economic assessment of stocking rate alternatives. Numerous studies applied some variation of the single-period model of optimal stocking rates as described by Workman (1986) with notable examples including research conducted in Wyoming by Hart and various coauthors (Hart et al. 1988a, 1988b; Hart 1991; Manley et al. 1997). Hart's model applications improved the economic assessment of optimal stocking rates by modifying the input to be the number of animals grazing per unit of forage produced (grazing pressure [GP]) and not animals per hectare (stocking rate [SR]). The Hart studies were also unique in that long-term grazing studies were used to define key production relationships. Most economic studies about stocking rates and rangeland investment analysis have typically used biophysically simulated data (Huffaker and Cooper 1995; Aguilar et al. 2006; Teague et al. 2008).

Hart's revised definition of grazing input showed that a given cost–price situation results in an economically optimal GP for the current grazing period, but the optimal number of animals stocked per unit area (SR) is also determined for the given forage condition. The economically optimal stocking rate will depend on sale prices and production costs and will vary annually. Further, the production function captures key input–output relations crucial to the economic assessment. A favorable rainfall year means more forage, and the production function shifts upward. Rates of gain will be different for different grazing seasons, and the economic assessment is defined for a particular grazing season and grazing system. An altered grazing season or rotational scheme potentially

Grazing in Northeast New Mexico. (Photo: Victor Espinoza)

Low sagebrush grassland, Indian paintbrush, Big Horn Canyon National Recreation Area, Wyoming. (Photo: John Tanaka)

shifts the production function up or down. A limitation is that the traditional economic model does not include the economic value of other ecosystem services, and it considers production only during the current period.

Because of the complexity and lack of response data, few economic studies have moved beyond the simple single-period economic model of stocking rates. Yet, for the cow–calf producers most commonly using western rangelands, interyear interactions always occur, and the stocking rate decision becomes much more complex with the additional uncertainty about available forage now and in the future. The cow herd must be maintained across years, and forage availability is highly variable between years. Overgrazing during dry years becomes problematic and controversial when conflicts arise between livestock producers and land agency personnel about reducing stocking rates during drought periods.

Early dynamic economic studies of optimal stocking rates included Burt (1971), Karp and Pope (1984), Pope and McBryde (1984), Rodriquez and Taylor (1988), Garoian and Mjelde (1990), Torell et al. (1991), and Huffaker and Cooper (1995). There is a widely held belief that individual short-term optimization is at odds with long-term sustainability of an ecological–economic system, suggesting that a dynamic approach is needed. Several studies have not found this to be the case, however. Torell et al. (1991) found the *intertemporal* grazing impacts on forage production were not that important. If profit-maximizing livestock producers would maximize profit during the current period, then nearly identical stocking decisions would be made as those obtained from a dynamic decision model, and optimal stocking rates would be at sustainable levels. Falling animal performance was the critical driver for stocking rate decisions. The number of stocker animals in the pasture would optimally fluctuate annually with forage conditions, beef prices, and production costs. Only occasionally would the cost–price situation be such that major interyear forage impacts would occur (Torell et al. 1991). Similarly, Quaas et al. (2004) concluded that for typical semiarid rangelands, under plausible and standard assumptions, short-term optimization leads to sustainable outcomes. Contrary to this finding, Teague et al. (2009), using a simulation model of semiarid savanna rangeland, found that sustainable stocking rates were 67–75% those that would maximize profit to livestock producers. They note that earning potential was four times higher for range in excellent condition and suggest a need to stock rangeland more lightly so as to prevent rangeland degradation and improve range conditions. The risk of potential herd liquidation and the need for feed purchase along with other negative impacts to the rangeland resource increase as stocking rates increase.

Matching stocking rates with dynamic forage conditions has emerged as the most consistent management variable influencing both plant and animal responses to grazing (Briske et al. 2008). It follows that it is also the most important factor influencing ranch profitability and economic responses to grazing. Typical drought management strategies include increased supplemental feeding, maintaining a conservative stocking rate so that destocking is rarely necessary, maintaining grazing flexibility by having yearlings as one of multiple enterprises on the ranch, and leaving a significant amount of herbaceous production at the end of the grazing season (Stafford Smith 1992; Hart and Carpenter 2005). Torell et al. (2010) found that interyear forage variability decreased net ranch returns by 46% relative to what could be obtained without variable forage conditions.

Economics of Grazing Systems. A deferred, rotational or other type of grazing system must result in one of two production responses for the practice to be economically beneficial to a livestock producer. First, animal performance must be improved with the alternative grazing system (shifting the production function up), or, second, forage production must improve over time (shifting the production function up gradually over time). It is also possible that these grazing systems may support more effective management decisions by some managers to induce these production responses (see the section "Social Aspects of Conservation Practices"). The prescribed grazing system could result in lower cost, and that would potentially justify the practice. From society's perspective, grazing practices may also reduce fire danger, provide other habitat improvements that are valued, or allow integration and adoption of other management practices that add value.

A recent synthesis paper (Briske et al. 2008) summarized key findings from many different studies about the benefits of rotational grazing systems as compared to a continuous, season-long grazing strategy. The main conclusion drawn from the review was that "rotational grazing as a means to increase vegetation and animal production has been subjected to as rigorous a testing regime as any hypothesis in the rangeland profession, and it has been found to convey few if any, consistent [ecological] benefits over continuous grazing" (Briske et al. 2008, p. 11). As noted in the review, there has generally not been an economically measurable difference in plant production/standing crop or animal production between rotational and continuous grazing with similar stocking rates. The production function does not appear to shift up within a given year from improved animal performance or over time because of increasing forage production. Economically, this means that if rotational grazing requires more labor, capital, and management inputs, the lower-cost continuous grazing alternative would be preferred based solely on net ranch returns. This preference may be altered by other goals, such as nonvalued ecosystem services or other societal benefits.

Considering specifically the economics of implementing grazing systems, the CAB abstracts populated with 60 citations using the key words of "grazing systems" in the title and "economics" and "rangelands" in any other field. Further screening indicated only 23 of the articles were relevant. Nearly half the economic studies were conducted in Switzerland, Africa, and Australia. Some studies compared primarily different stocking rates or grazing intensities (Behnke 2000; Rook et al. 2004; Trapnell et al. 2006). Two African studies evaluated the

"…economic evaluations of grazing systems have consistently found season-long continuous grazing to be the most economical or not different in production and rate of economic return."

“The most valuable forage is not necessarily of the highest quality; rather, it is available when few alternatives exist.”

economics of multipaddock systems and found that few paddocks rather than many paddocks were most economical (Beukes et al. 2002; Mentis 1991). Hart et al. (1993) noted that cross-fencing and water development were important to achieve uniform utilization of forage and for minimizing grazing energy costs, but these goals can be achieved independently of the grazing system.

Not surprising, given the finding of the Briske et al. (2008) literature review that few forage and livestock benefits accrue from rotational grazing, economic evaluations of grazing systems have consistently found season-long continuous grazing to be the most economical or not different in production and rate of economic return (Heitschmidt and Kothmann 1980; Quigley et al. 1984; Van Tassell and Conner 1986b; Hart et al. 1988a; Heitschmidt et al. 1990). A somewhat different conclusion was reached by Owensby et al. (2008) where season-long stocking of a tallgrass prairie site in Kansas was found to have the lowest economic risk (i.e., variability), but returns per hectare were higher for an intensive early stocking system. Given added capital and labor requirements for more intensive grazing systems, not only must there be measurable production responses from the practice, but those responses must be substantial enough to justify the added expense. The literature does not show this to be the case, and, as noted in the prescribed grazing chapter, there is minimal information documenting the influence of intensive grazing systems on the effectiveness of adaptive management.

Briske and coauthors (this volume) extensively explored published literature dealing with grazing/wildlife interactions. They note that many wildlife species, including birds and wild ungulates, demonstrate a relative neutral response to the type of grazing system in place. Both positive and negative responses were noted. Given the general lack of response, it is not surprising that we found no studies that explored the economics of wildlife interactions and grazing systems.

Another area where grazing systems and deferred grazing has been shown to be beneficial is as an adjustment mechanism to drought and seasonal forage shortages. As noted by Tanaka et al. (2007), the seasonality of forage use is an important consideration in ranch planning because the number of forage alternatives is limited during certain months of the year, and some forages and harvested feeds are considerably more expensive. The most valuable forage is not necessarily of the highest quality; rather, it is available when few other alternatives are. A grazing scheme that leaves residual forage for carryover and use during a future short-supply period, allows riparian areas to be rested, extends the grazing season, and/or replaces an expensive feed alternative has substantial economic value (Stillings et al. 2003; Tanaka et al. 2007).

Greater reliance on livestock grazing compared to harvested forages is an effective way to reduce feed costs that requires a planned grazing strategy. Adams et al. (1994) estimated that the weaning weights of calves were increased 5 kg by grazing meadows during May instead of feeding hay, and feed costs were substantially reduced. Extending the grazing season in winter and spring increases ranch returns over traditional systems that used a greater amount of harvested forage. Winter feeding costs are the largest expense for many livestock operations (Prevatt et al. 2001), and innovative grazing schemes have the potential to reduce those costs.

Briske et al. (2008) noted that even if evidence for production benefits from rotational grazing is inconclusive or nonexistent, many livestock producers believe that such benefits do exist. The website by Holistic Management International (http://www.holisticmanagement.com/n7/results_07.html; last accessed March 27, 2009) documented the stepped-up level of management and perceived benefits that ranch managers practicing holistic management have received. The survey of 43 ranch managers in the northern Rockies indicated that a high percentage of participants now do annual ranch planning, set goals, and have annual and formally documented land monitoring programs in place. It is from these activities that the majority of benefits from added management likely occur rather than a specific grazing practice. Further, they largely believed that production benefits do in fact exist, in contrast to the experimental evidence summarized by Briske et al. (2008).

Management and financial skills have typically been taught at holistic management schools, and these skills may be the biggest benefit that livestock producers have received from intensive grazing system training. These benefits are discussed in greater detail in the section "Social Aspects of Conservation Practices."

Brush Management

"The economics of brush control must be determined by the amount of forage and meat products gained; however, the principal objective in brush control should be an upgrade in range condition" (Hyder and Sneva 1956, p. 34). This statement, made over 50 yr ago, clearly articulated what was then and continues to be the main reason and economic rationale for brush control practices. The NRCS now recognizes six broad reasons for brush management (USDA-NRCS 2003):

- Added forage for livestock
- Restoration of natural plant community balance
- Creating the desired plant community
- Controlling erosion, reducing sediment, improving water quality, and enhancing stream flows
- Maintaining and enhancing wildlife habitat including protection of endangered species
- Protection of life and property from wildfire hazards

A Texas landowner survey describing incentive for brush control found that increased forage production and water conservation were most important (Kreuter et al. 2005). Secondary incentives were to improve aesthetic values, benefit the next generation, improve wildlife habitat, and improve real estate values.

Forage Production Benefits. The traditional brush control economic analysis that Hyder and Sneva (1956) described uses standard net present value (NPV) tools with the cost of the brush control treatment compared to discounted net future forage production benefits expected to be realized over some finite treatment life. The key tasks and elements of the economic assessment are to define the expected forage response (an assessment of forage productivity with and without the treatment) and estimate the added livestock carrying capacity possible over time with brush control, select an appropriate discount rate to properly account for timing difference between benefits and costs, and price and value the added grazing capacity (Workman and Tanaka 1991). A positive NPV or a benefit-to-cost ratio greater than one implies an economically feasible rangeland management practice (Workman 1986).

The economics of controlling brush for enhanced livestock production is variable depending on the economic value assigned to the forage, the assumed rate of forage response and treatment longevity, the assumed proper use rate or allowance for how much of the additional forage will be harvested to generate additional livestock income, and the discount rate used. The economic value of the added forage is influenced by the quality of the forage for grazing, by the forage and feed alternatives available, and by livestock prices. Most important is whether the added forage would be available during periods when other forages are scarce and costly (Evans and Workman 1994).

The expected longevity of brush control treatments varies widely by brush species and range site, as does the forage response. Yet a consistent overstory–understory relationship has been noted for many shrubland communities and species. These relationships generally show a downward-sloping sigmoid or exponential curves when herbaceous yield ($kg \cdot ha^{-1}$) is plotted against brush canopy (%; Ffolliott and Clary 1972). This suggests that increased brush cover diminishes understory forage production but at a decreasing rate.

Herbaceous production has been shown to increase an average of three to five times following effective control of many brush species located on productive range sites, including sites infested with Wyoming big sagebrush (*Artemisia tridentata wyomingensis*; Hyder and Sneva 1956; McDaniel et al. 2005), broom snakeweed (*Gutierrezia sarothrae*; McDaniel et al. 1993), pinyon-juniper (Clary et al. 1974; Pieper 1990), and redberry juniper (*Juniperus pinchotii*; Johnson et al. 1999). Successful control of other species like mesquite (*Prosopis glandulosa),* salt cedar (*Tamarix*

“…forage production, not water yield, is the primary benefit of the brush control practice, especially on more mesic sites.”

spp.), and creosotebush (*Larrea tridentata)* resulted in an increase in grass cover, but with minimal changes in harvestable forage except on productive sites with adequate rainfall (Ethridge et al. 1984; Harms and Hiebert 2006; Perkins et al. 2006; Combs 2007).

The economics of brush management practices continues to be evaluated on the basis of the amount of forage and meat products gained by implementing the practice. The economic component of PESTMAN, a holistic decision support system currently in development at Texas A&M University (PESTMAN 2009), is driven by the anticipated forage response to a selected brush control treatment. Yet, as noted over 30 yr ago by Smith and Martin (1972), based on livestock production value, most rangeland management practices showed a negative benefit-to-cost ratio (costs exceed benefits) based only on the value of the added forage. This is a consistent and continuing conclusion from studies dealing with the economics of brush control practices. Increased returns from improved animal performance and production are usually too low for brush control to be economically justified (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a). Landowners recognize this, and many brush control projects are implemented under cost-share arrangements with state and federal land management agencies.

Torell et al. (2005a) found that a cost-share payment of about 30% of the treatment cost was required to justify control of big sagebrush in northwestern New Mexico when the added forage from the brush control practice was valued at an intermediate level of \$7 · animal unit month^{-1} (AUM; in 2003 dollars). The NPV of the investment was positive except at two relatively unproductive sites, when forage was valued at \$10 · AUM^{-1}. A rangeland management practice that adds forage during a critical and limiting season makes forage valuable, and many times added forage production alone justifies the improvement in these cases (Evans and Workman 1994).

If a brush control project successfully increases available forage, increasing livestock numbers is not always justified. In some cases, a justification for brush control is that the stocking rate on the area can be maintained nearer its actual capacity by recognizing that current stocking rates are not sustainable. Sagebrush control near Farmington, New Mexico, helped the Bureau of Land Management (BLM) avoid potential conflict and lawsuits with grazing permittees and other parties because positive steps were taken to reduce grazing pressure but without forcing major herd reductions (Torell et al. 2005a). Similarly, forestalling the need for controversial grazing reductions was a primary benefit of the 11-yr (1962–1972) Vale Rangeland Rehabilitation Program initiated on BLM lands in eastern Oregon (Bartlett et al. 1988). Therefore, the maintenance of sustainable livestock carrying capacity should be given explicit consideration when evaluating the effectiveness of brush management programs.

Watershed Benefits. Watershed benefits are increasingly used as justification for public expenditures for brush control. Large tracts have been treated at great expense to control brush, especially salt cedar, to realize perceived watershed benefits, including added stream flow, water yield, and aquifer recharge. For various reasons, trees and brush are perceived to have a higher evapotranspiration (ET) rate than herbaceous species within the understory (Wilcox and Thurow 2006). The argument is made that if ET loss can be reduced by managing rangelands for a greater grass component and a lesser tree and shrub component, more water will be available for runoff and/or deep drainage. As noted by Wilcox and Thurow (2006), this argument has been shown to be true in a variety of humid, montane and Mediterranean climates, where studies have shown increases in water yields tied to removal of trees and shrubs. In semiarid rangelands, however, water yield benefits have not been demonstrated on scales that would greatly alter regional water supplies (Wilcox 2002). Sturges (1983) noted that the response of the soil water regime to a brush control treatment is inversely related to the response in herbaceous production. This suggests that much of the added water from brush control in arid areas is used to produce greater herbaceous production. Thus, added forage production, not water yield, is the primary benefit of the brush control practice, especially on more mesic sites. It appears that there is no real potential for increasing stream flows in addition

Prescribed burning mosaic, near Lakeview, Oregon. (Photo: John Tanaka)

to the forage benefit unless annual precipitation exceeds 450–500 mm (Wilcox 2002).

Research addressing the economics of brush control for enhanced water yield has been conducted only on Texas watersheds. As noted by Wilcox (2002), the perception is widespread that the water supply in Texas can be substantially increased through aggressive control of mesquite and juniper. Several studies have explored the cost implications of Texas brush control practices from a rancher perspective (Lee et al. 2001; Olenick et al. 2004a), and surveys have been conducted to evaluate ranchers' willingness to participate in brush control projects designed to enhance water yields (Thurow et al. 2000; Kreuter et al. 2004, 2005). These landowner surveys indicated that a subsidized public cost-share program would be necessary for widespread participation in brush control projects by Texas ranchers if public watershed benefits were the goal.

Economic studies have generally evaluated the economic value of watershed benefits indirectly using the procedure described by Lee et al. (2001) and Olenick et al. (2004a). The discounted forage production benefits are assumed to go to private ranchers, and they have an assumed willingness to participate at a maximum cost up to this level. Beyond this point, the NPV of the investment would be negative (benefits < costs) for the private landowner. Watershed benefits or the public's benefit is estimated to be the present value of the brush treatment cost minus landowner forage benefits. This residual value is a surrogate measure of the value that society must place on watershed benefits if the investment is to

Prescribed fire in pinyon-juniper, Arizona. (Photo: John Tanaka)

be economically efficient and have a positive overall NPV. The analysis makes no attempt to estimate what watershed benefits actually were but instead estimated the level of required public benefits required to justify the total cost of brush control. The obvious justification and assumption was that the cost-share program accurately reflected social priorities. Skeptics can counter, however, that land management agencies' budgets and spending priorities most often reflect political and bureaucratic objectives and are not a reflection of social value (Skaggs 2008).

Additional Texas studies have used plant growth, hydrologic, and economic models to determine costs and benefits (added water) resulting from brush control (Bach and Conner 1988; Conner and Bach 2000; Lemburg et al. 2002; Olenick et al. 2004a). Simulated estimates indicated that the public cost of additional water ranged from $26 to $129 per 1 000 m^3 depending on area and type of treatment. This compared with an estimated $65 per 1 000 m^3 for leasing water pumped from the Edwards Aquifer (Olenick et al. 2004a).

Tamarix, or salt cedar, is often controlled on the basis of an economic justification associated primarily with water conservation. It is an expensive species to control ($4 000–$12 000 · ha^{-1}), requiring repeated mechanical, fire, chemical, and revegetation treatments as summarized at http://saltcedar.nmsu.edu. Economic feasibility studies assessing the costs and benefits of salt cedar control have been conducted for some western US waterways (Great Western Research Inc. 1989). Horton and Campbell (1974) estimated that water savings, based on the difference in water use between salt cedar and native vegetation, were as high as 3 000 acre-feet · yr^{-1} following salt cedar control on the Colorado River. It has been estimated that 568 000 acre-feet · yr^{-1} of water are lost to salt cedar from the Bonneville Unit of the Central Utah Water Project on the Colorado River at an estimated cost of $27 million annually (Brotherson and Field 1987). Zavaleta (2000) estimated that marginal water losses to salt cedar are comparable to annual precipitation totals for the arid western states where salt cedar has been a problem. She estimated that *Tamarix* stands consume 3 000–

4 500 $m^3 \cdot ha^{-1} \cdot yr^{-1}$ (8 219–12 329 $L \cdot d^{-1}$) of water, more than the native vegetation it replaced. Lost economic value in 1998 dollars was estimated to be \$284–\$447 $\cdot ha^{-1}$ of land infested by the invasive species. Using a multiyear average treatment cost of \$5 000 $\cdot ha^{-1}$ and with a 6% discount rate, Zavaleta (2000) estimated that it would take 16 to 50 yr to break even on the initial and follow-up treatment costs, depending on the value assigned to the water saved.

The economic assessment by Zavaleta (2000) likely used the popular *Tamarix* water consumption value of about 757 $L \cdot d^{-1} \cdot plant^{-1}$. Owens and Moore (2007) reviewed the literature on water use by *Tamarix* and concluded that a more realistic estimate of the maximum daily water use was less than 122 $L \cdot d^{-1} \cdot plant^{-1}$. As Owens and Moore (2007) noted, at this reduced estimate of water savings, the economics of spending \$5 000–\$8 000 $\cdot ha^{-1}$ to control salt cedar would be dismal, based only on water conservation benefits.

Wildlife Benefits. Wildlife habitat and, therefore, wildlife populations are variously influenced by the relative cover of brush and tree species, and many wildlife species prefer a much denser overstory then would be optimal for livestock forage production and water yield alone (< 5% brush canopy). Surveys of Texas landowners indicated an average brush canopy cover of 41% on their ranches as compared to a preference of 27%, a level that would maximize the value of lease hunting for white-tailed deer (Thurow et al. 2000). Many Texas ranchers are more interested in brush thinning than brush eradication because of the high revenues derived from lease hunting (Kreuter et al. 2004).

Few studies have evaluated the economics of brush management for enhancing wildlife values and especially with consideration of the trade-offs with other resource values. Aldrich et al. (2005) developed a multiperiod linear programming model to evaluate the economics of western juniper control in central Oregon. Profit from livestock income was maximized by choosing economically optimal juniper management strategies so as to manipulate available forage resources for livestock production. Wildlife income was not considered a source of ranch revenue, but equations were included to evaluate how optimal juniper control strategies for livestock production would impact quail, deer, and elk numbers. Given the conflicting level of desired juniper for cattle versus wildlife production and with maximization of income from cattle only, wildlife numbers were projected to decline following livestock-revenue profit-maximizing strategies.

Standiford and Howitt (1993) developed an optimal control model that considered multiple resource values from management of California's hardwood rangelands, including forage production, oak wood sales, and hunting revenue. Key relationships were identified, including expected oak tree growth rates, the interaction of tree overstory versus livestock forage production, and hunting revenue potential under different tree canopies. Revenue from hunting was defined to increase with oak crown cover, whereas revenue from cattle decreased with an increasing canopy of oak trees. Economically, optimum oak canopy was found to vary depending on the resource values considered, but the optimum was consistent with expectations. When only livestock production value was considered, the optimal control model indicated that oak trees would be gradually cleared because of the resulting additional forage for livestock. Over the assumed 13-yr planning horizon, oak canopy would be reduced from 55% to less than 5%, but immediate tree clearing was not feasible given imposed realistic budget constraints. Adding firewood harvest to livestock income resulted in a light tree harvest for firewood (2–3% of oak canopy). The NPV addition from firewood revenue was 3%. Managing for cattle and wildlife increased NPV by 40% and greatly changed management of the oak canopy. The oak canopy would be maintained at 55%, no firewood would be harvested, and reduced cattle numbers would be optimal. The marginal economic value of the oak canopy for wildlife habitat exceeded the marginal value of greater livestock forage and firewood volume.

Bernardo et al. (1994) highlighted that wildlife is an increasingly important source of ranch income, and hedonic ranchland valuation

Spraying salt cedar along Pecos River. (Photo: Kirk McDaniel)

models show market preferences for ranches with wildlife revenue and hunting potential. Ranches with quality wildlife habitat and scenic appeal bring premium prices in the ranch real estate market. Torell et al. (2005b) estimated that 25% of New Mexico ranches have wildlife income-earning potential and that these ranches sell for premium prices, especially scenic mountain ranches with elk herds present.

As discussed above, Standiford and Howitt (1993) have shown optimal brush management strategies for wildlife to be different from optimal production strategies for livestock on California's hardwood rangelands. Bernardo et al. (1994) similarly concluded that some reduction in cattle grazing was necessary to maintain deer and quail habitat at desired levels. As noted by the Texas Parks and Wildlife Management Department (2008), most wildlife species are selective foragers, preferring to feed on a wide variety of plants rather than a few specific ones. Therefore, habitat improvement recommendations should emphasize the need for an even distribution and high availability of potential forage plants from season to season. Brush management can allow solid stands of woody vegetation to be interspersed with cleared areas over the landscape. Cleared strips or blocks can produce desirable forb and browse production while retaining an adequate mosaic of woody cover for escape, nesting, or protection from the elements. Properly utilized brush management practices can improve the availability of escape cover and food plants for both wildlife and livestock, though livestock production profitability will not be maximized.

Prescribed Burning

Various search engines, including AGRICOLA (Agriculture abstracts), CAB Abstracts, and wildlife and ecology studies worldwide, were used to evaluate the peer-reviewed literature that is available on the economics of prescribed fire as a conservation practice. "Fire" was required in the title and key words, or the title had to include "economics." After some screening and elimination of irrelevant papers, 46 citations dealt with fire and economics. Seven of these papers also used the term "social" in either the title or the key words. Six of the papers dealt primarily with fuel reduction on forested lands using prescribed fire.

Fire Hazard Reduction, Liability, and Risk Concerns. Of the papers dealing with human and economic aspects related to rangeland fires, nearly half the papers dealt with wildfires and using prescribed burning as a way to reduce the risk and economic damage from wildfires (Kaval et al. 2007; Mercer et al. 2007; Yoder and Blatner 2004). As noted by Kaval et al. (2007), residents within wildland–urban interface zones recognized the positive role that prescribed burning can have in reducing the dangers of wildfire and are willing to pay for positive fire risk mitigation measures. In this Colorado study, residents responding to the willingness-to-pay survey were willing to pay an average of nearly $\$800$ year^{-1} for fuel management treatments to reduce fire risk. At a low $\$5 \cdot$ household^{-1} annual bid price, all survey respondents were willing to pay for prescribed fire so as to reduce fire danger, while at a relatively high level of $\$1\,500 \cdot$ household^{-1} annually, only 14% were willing to pay for prescribed fire treatments.

Cost studies have shown prescribed fire to be a cost-effective fuel reduction method. Yet Hartsough et al. (2008) found that using prescribed fire to reduce fire danger was relatively high cost in the western United States because of terrain and stand conditions, high fuel loads, and the need to ensure that prescribed fires do not escape. Evaluating data at seven sites in the western United States indicated that gross costs of mechanical fuel reduction treatments were more expensive than those of prescribed fire, but net costs were similar or less after the market value of the harvested wood products were deducted. The relative merit of using prescribed fire versus mechanical canopy removal was highly sensitive to the market value of forest products generated by the mechanical operation (Hartsough et al. 2008). As noted by Omi (2008), fire hazard reduction through fuels management is controversial, and the literature on fuel treatment effectiveness in reducing fire danger is nearly nonexistent, as are the data required to assess the trade-offs between alternative fuel reduction treatments.

Seven papers addressed fire escape and the liability risk associated with prescribed fires, all published since 2000. As noted by Yoder (2008), prescribed fire is considered a useful but risky method of reducing wildfire risk, increasing forage production, and improving wildlife habitat. This risk has resulted in new laws and reforms (Sun 2006), and Yoder found that these new liability laws and regulations have effectively reduced the incidence and severity of escaped fires.

Improve Forage Production. Some type of intervention may be needed to improve range condition and to redirect what may be a continued decline in rangeland productivity. However, Fuhlendorf and coauthors found a very weak argument for using fire to increase herbaceous vegetation production, particularly perennial grasses (this volume). They note that perennial grasses generally declined in the years immediately following prescribed fire but that most perennial species recovered within 2–3 yr. The delay in realized grazing benefit is problematic for realizing positive economic returns from the prescribed fire treatment given the time value of money. Further, conducting a prescribed burn may be limited or delayed by air temperature conditions, relative humidity, wind speeds, and the availability of fine fuels to carry the fire so as to minimize the risk of fire escape, to minimize damage to perennial grasses, and to carry the fire over the desired area. Fire is not as easy or as convenient to use as chemical treatments for brush control. McDaniel et al. (1997) noted that over a 6-yr study period on the shortgrass prairie of New Mexico, the desired fire conditions recommended by Wright and Bailey (1980) were rarely observed. For many arid rangelands, accumulating fine fuels under a dense brush canopy can be particularly problematic for implementing prescribed burning treatment (Bastian et al. 1995; McDaniel et al. 1997; Teague et al. 2001).

Eight papers studied the economics of prescribed fire as a strategy for reducing brush overstory and increasing the production of understory forage species. Two of these studies were outside the United States (Trollope 1978; Henkin et al. 1998). Most of the US studies evaluated the economics of prescribed burning for control of honey mesquite and cactus in the Texas Rolling Plains (Teague et al. 2001, 2008). Other economic studies addressed oak–hickory forests (Bernardo et al. 1992), eastern red cedar (*Juniperus virginiana*; Bernardo et al. 1988), Macartney rose (*Rosa bracterata* Wendl.; Garoian et al. 1984), and big sagebrush (Bastian et al. 1995).

Of the limited economic studies about prescribed fire to enhance forage production, a common prescription was an initial chemical treatment to reduce the brush overstory followed by prescribed fire treatments at 5–7-yr intervals as a maintenance treatment (Van Tassell and Conner 1986a; Teague et al. 2001). The economics of prescribed fire treatments were estimated to be better than chemical treatments even if the burn treatment was considered to be considerably less effective in overstory reduction and longevity (Teague et al. 2001). This is because, ignoring fire risk, the cost of prescribed burning was assumed to be much cheaper than chemical treatments in all the studies. Bastian et al. (1995) estimated that the cost of prescribed fire was only half that of chemical treatment, and Teague et al. (2001) estimated the cost of follow-up burn treatments to be only 10% of the initial $\$56.81 \cdot ha^{-1}$ chemical treatment. Perhaps the biggest limitation of the economic studies of brush control, including both chemical and fire, was the estimation of the forage production response curves. Forage response can be expected to be highly variable depending on soils, pretreatment brush cover, climatic conditions, and forage and brush

“Perhaps the biggest limitation of the economic studies of brush control, including both chemical and fire, was the lack of data to estimate forage production response curves.”

Snow geese at Bosque del Apache National Wildlife Refuge, New Mexico. (Photo: John Tanaka)

species. The economic studies were largely based on relatively short response studies (< 10 yr) and various unmeasured assumptions about the rate of brush reinvasion. Given the general nonlinearity of overstory–understory relationship, the economic studies concluded that a relatively dense stand of brush must be present initially to be economically feasible for control by either fire or chemical methods. None of the studies mentioned or considered the reduction in perennial grass cover in the immediate years following prescribed burning treatments as compared to chemical treatments.

Wildlife Benefits. We found only two studies investigating the economics of prescribed fire to improve big game habitat and increase wildlife income. A study by Gonzalez-Caban et al. (2003) (also published as Loomis et al. 2002) developed production functions relating deer harvest response to prescribed burning. Diminishing marginal benefits were noted. An additional 445 ha of prescribed burn increased deer harvest by 33 head, whereas the next 1 502 ha of prescribed burn increased deer harvest by eight head. When compared to the estimated \$519–\$593 $\cdot$ ha^{-1} cost of conducting prescribed burns, the economic value of the added deer harvest was only 3.4% of the total cost for the first 445 ha burned.

Teague et al. (2001) studied the economic response of honey mesquite control in the Rolling Plains of Texas from both herbicide and prescribed fire treatments. They noted that burning at a 5–7-yr interval improved wildlife habitat. Their economic evaluation of herbicide and prescribed fire treatments was most sensitive to realizing a wildlife income response. If treatment on any part of the ranch increased wildlife income, then the NPV of the brush control treatment was substantially increased.

Rangeland Planting

Rehabilitation of rangeland by seeding and planting began in the western United States in the late 1800s, and, according to Heady and Child (1999), more literature exists on range seeding than any other practice in range management. They also note that the environmental movement after 1970 demanded less seeding of rangeland with monoculture species and more with native species. Rehabilitation and prevention of erosion have replaced increased forage production as the primary objective for seeding public lands. Rangeland seeding is also considered advantageous for managing weeds and cheatgrass (*Bromus tectorum*; Young and Clements 2009).

The Heady and Child (1999) textbook chapter (chapter 24) provides a wealth of information about ecological considerations about rangeland planting. Seeding guidelines they identify include the following:

- *Is the seeding needed?* Removal of the competitive brush overstory may be adequate. Seeding has the greatest potential for profitable returns when native vegetation does not exist.
- *Is the climate favorable?* Successful plantings are infrequent in areas receiving less than 250 mm of precipitation per year, but failure for areas with greater than 600 mm of rainfall are less frequent.
- *Is the habitat favorable?* Select seeding sites with the most herbaceous response potential.
- *What species should be planted?* Consider seasonal forage demands and the potential to replace expensive feeding alternatives. Recognize that a mixed diet is generally more desirable and often will produce greater livestock gains than a monoculture.
- *Manage the seeded area.* Provide grazing deferment after establishment and do not overgraze.

The steps required to analyze the economics of revegetation projects include incorporation of all these considerations as they relate to potential costs, benefits, and risks (Workman and Tanaka 1991).

Economic literature evaluating rangeland seeding are generally at least 30 yr old, and seeding success is variable and depends largely on selecting a desirable site in an adequate rainfall area. Much of the literature exists in the form of extension guides and bulletins. These bulletins generally provide guidelines for designing an economically successful project instead of evaluating the economics of specific improvement projects or case studies (Lloyd and Cook 1960; Wiens et al. 1969; Kearl and Cordingly 1975; Wambolt 1980; Kearl 1986; Workman and Tanaka 1991; Heady and Child 1999).

The probability of successful seeding establishment was highlighted in the only study found concerning the economics of range reseeding in the desert Southwest (Ethridge et al. 1997). This 6-yr study of seeding trials on the Jornada Experimental Range near Las Cruces, New Mexico, considered 14 different plant varieties, including introduced and native species. The study indicated that reseeding was not an advisable financial investment for the Chihuahuan deserts of southern New Mexico because of the high probability of stand failure. Estimated NPV was negative for all species planted and seedbed preparation strategies.

Most of the economic studies on rangeland planting—or rangeland seeding or reseeding, as it is categorized in the literature—are about the economics of seeding crested wheatgrass (*Agropyron desertorum* and *A. cristatum*). From 1945 until 1965, several million hectares of sagebrush rangeland were seeded to crested wheatgrass in the Intermountain West (Young 1994). It was estimated that in Nevada, 0.4 million of the 11 million ha of sagebrush rangeland were seeded to wheatgrass. The seeded area constitutes only 2% of the total rangeland in Nevada but produces 10% of the harvestable rangeland forage (Young and Evans 1986).

Seeding sagebrush rangelands to crested wheatgrass was generally found to be a very economical practice because forage production was 3–20 times greater than that of the native plants it replaced, calf crop and average weaning weights increased, and early spring use replaced expensive hay as an alternative lower-cost feed. In some studies, rates of return were estimated to be in the range of 10–22% with an anticipated stand life of 25 yr or more (Kearl and Cordingly 1975; Shane et al. 1983). Not all economic studies estimated positive economic returns, however. Godfrey (1986) reviewed 24 economic studies conducted between 1943 and 1979 that dealt with seeding crested wheatgrass and found net economic returns to be positive in nine of the studies and variable or unknown in the other studies. He attributed the variability in NPV estimates to four main reasons: 1) some of the plantings were failure, 2) low-production areas were seeded instead of high-potential areas (they used a worst first selection criteria), 3)

Bison at the Wichita Mountains Wildlife Refuge, Oklahoma. (Photo: John Tanaka)

too much was spent for brush removal, and 4) other, less productive improvements were included along with the seeding project.

Widespread planting of crested wheatgrass is no longer common, as it is an exotic monoculture, and successful widespread planting altered sagebrush habitat required for sage-grouse (*Centrocercus urophasianus*; Connelly and Schroeder 2000). As noted by Young and Evans (1986), the golden age of planting crested wheatgrass lasted for barely a decade, from the mid-1950s until the mid-1960s. Its role is now considered to be in the reclamation of drastically disturbed lands (Depuit 1986), and it has potential when seeded with forage kochia (*Kochia prostrata* ssp. *virescens*) to outcompete cheatgrass (*Bromus tectorum*; Harrison et al. 2000). Widespread use and plantings indicates that this forage species was one of the most economically successful rangeland plantings in its day, when only livestock forage production was valued.

Riparian Herbaceous Cover

Literature dealing with the economics of improving riparian herbaceous cover focuses primarily on nonrangeland (e.g., forested wetlands) areas and on tree cover rather than herbaceous cover. An EBSCO search with the terms "contingent valuation" and "riparian" returned 25 citations, but there were no studies found that specifically dealt with the economic value of establishing riparian herbaceous cover in rangeland areas as a way to improve riparian areas.

One of the few economic studies, a contingent valuation to estimate the benefits and costs of riparian restoration projects along the Little Tennessee River in North Carolina (Holmes et al. 2004), found net benefits from riparian ecosystem restoration to be strongly positive but much larger for large-scale projects. Restoration benefits were described in terms of five indicators of ecosystem services: abundance of game fish, water clarity, wildlife habitat, allowable water uses, and ecosystem naturalness. Other economic studies explored the willingness to pay and willingness to accept payment for provision of a riparian strip or corridor for habitat preservation (Amigues et al. 2002; Qiu et al. 2006). The equivalent of a positive benefit-to-cost ratio was indicated with results consistent with housing price differentials (house prices with or without riparian strip or corridor for habitat preservation) for stream access in the area (Qiu et al. 2006). The willingness of recreational visitors to pay for riparian area preservation that maintained bird diversity in the San Pedro River basin of Arizona was explored by Colby and Orr (2005). They estimated that a one-time aggregate monetary willingness to pay by nonlocal visitors for riparian area preservation was \$2.77 million.

Upland Wildlife Habitat Management

As previously noted, wildlife income is an increasingly important part of total ranch income, and ranchland market values are greatly influenced by scenic views, recreation, and hunting opportunities. Torell et al. (2005b) found that adding wildlife income to a New Mexico ranch contributes 2.5 times more to ranchland market value then does a similar amount of livestock income. These ranchland real estate market effects have been noted for many years (Pope 1985; Torell et al. 2005b). The obvious implication is that significant opportunities exist to increase economic values through upland wildlife habitat management if increased hunting and wildlife viewing opportunities can be created.

A great deal of research has been conducted to estimate the demand and economic value of various wildlife species, with obvious value implications for wildlife habitat improvement. One notable author, John B. Loomis at Colorado State University, has contributed greatly to the development and application of nonmarket valuation procedures. However, much of his research and other related research, as noted by Daniels and Riggs (1988), has concentrated on estimating the value of the wildlife and not the habitat sustaining wildlife. As noted by Bernardo et al. (1994), the weak link is the lack of data required to translate physical effects of habitat improvement practices into altered wildlife numbers and economic benefits. The Bernardo et al. paper provides one of the limited cases where the production trade-offs between cattle grazing and wildlife habitat were estimated. A second study was a California study on oak rangeland described earlier by Standiford and Howitt (1993). A third study involved estimation of the production possibility frontier between stocker cattle and antelope by Bastian et al. (1991). Glover and Conner (1988) developed a linear programming model to evaluate the optimal (profit-maximizing) mix of cattle, sheep, goats, and deer on a representative ranch in the Edwards Plateau region of Texas and found that active management for wildlife added to net ranch income. Another study by Loomis et al. (1991) evaluated livestock grazing strategies that would potentially improve deer habitat in California and concluded that implementing a rest–rotation livestock grazing system with 1 yr or more of nonuse in a 3-yr cycle would increase hunting value far beyond the value lost from reduced livestock grazing. Others have estimated forage values for cattle versus wildlife (Martin et al. 1978; Cory and Martin 1985; Loomis et al. 1989), but they did not provide estimates within a multiple-enterprise context that estimated production possibilities and trade-offs.

Wildlife valuation procedures use various techniques to estimate a consumer's willingness to pay where no established market exists, relying on demand analysis and consumer surplus estimation (Sorg and Loomis 1985; Champ et al. 2003). Valuation of wildlife habitat uses these value estimates for wildlife and expands to a benefit-to-cost assessment where the economic value of increased wildlife numbers is compared to the cost of practices that improve habitat and ultimately wildlife numbers. Studies that have attempted to estimate the linkage between altered habitat and wildlife numbers include the prescribed burning assessment described above (Loomis et al. 2002; Gonzalez-Caban et al. 2003) where benefits from additional deer harvest was estimated to be no more than 3.4% of prescribed burning treatment cost, suggesting that deer hunting benefits represent only a small part of the multiple-use benefits of prescribed fire.

Lenarz (1987) concluded that treatments to increase forest openings were never cost

"Very little if any research exists showing the direct noneconomic effects of NRCS rangeland conservation practices on individuals, households, or social systems."

Planning in the tallgrass prairie, near Bowie, Texas. (Photo: John Tanaka)

effective based on the value of hunting licenses, but it was cost effective based on total gross hunting-related expenditures. Daniels and Riggs (1988) recognized and corrected the low economic value assigned by Lenarz (1987) by considering only the value of the hunting leases. They based deer values on standard willingness-to-pay measures and concluded that a positive NPV would be realized from investments to create forest openings whenever cleared areas were less than the 3% level and a reasonable discount rate was used.

Garrett et al. (1970) estimated the demand for deer hunting in Nevada and valued the habitat that supported hunting activity. They compared the habitat value to selected rehabilitation projects expected to alter deer numbers. The first improvement considered was crested wheatgrass (*Agropyron cristatum*) seeding with the estimate that the practice would be detrimental to deer numbers and wildlife value because of the decrease in forage species most desirable to deer. Chaining of pinyon-juniper at two sites was estimated to result in a positive benefit-to-cost ratio, ranging from 1.65 to 2.09 for the two sites, based only on increased economic value from deer utilization supported by a more open woodland canopy.

Similar to the hedonic modeling approach used by Torell et al. (2005b) to evaluate the contribution of wildlife to western ranchland values, Netusil (2006) used urban housing sales within the Fanno Creek Watershed within the city of Portland, Oregon, to evaluate how real estate prices varied with different amounts of upland wildlife habitat. Close proximity to a stream increased property values. A property's sale price was found to increase as the percentage of regionally significant habitat on the lot increased but at a decreasing rate. Property owners placed a premium price on lots with habitat providing the highest ecological values (large forest patches, wetland areas, and large contiguous patches) and discounted lots with lower-valued habitat. The maximum impact on house lot sale price was when upland wildlife habitat coverage on the property was about 38%.

SOCIAL ASPECTS OF CONSERVATION PRACTICES

Very little if any research exists showing the direct noneconomic effects of NRCS rangeland conservation practices on individuals, households, or social systems. It is likely that many producers do realize psychological benefits from conservation, as stewardship outcomes typically rank high among the management goals of livestock producers (Huntsinger and Fortmann 1990; Sayre 2004). Moreover, livestock producers who believe strongly in a responsibility to society are more likely to engage in environmentally desirable management practices, such as invasive weed control and riparian protection (Kreuter et al. 2006). Thus, non–peer-reviewed feature articles often refer to the psychological rewards ranchers enjoy because they employ conservation practices on land they hope to preserve for posterity (e.g., Little 2005; Smith 2008). Such rewards are often hard to document scientifically, however.

Indirect evidence of psychological benefit from implementing conservation practices comes from Holistic Management (HM), a program that typically advocates rotational grazing as part of an overall ranch management plan. Montagne and Orchard (2000) found that participating ranchers in the northern Rockies reported increased personal satisfaction after having adopted an HM approach. Stinner et al. (1997) found after interviewing HM participants nationwide that 91% reported improvements in quality of life after HM training. However, HM is a whole-ranch program that focuses on time management, goal setting, and monitoring as well as prescribed grazing, and neither study separated the effects of different aspects of the program. Moreover, such studies can speak only to the *perceived* effects of conservation; we have found no evidence that land managers, farm/ranch households, or group members that engage in conservation practices actually score higher on measurements of psychological or social well-being than producers who do not use such practices.

Much more is known about why people choose to adopt conservation practices than the relative effectiveness of their implementation. Studies of innovation adoption offer insight as to which outcomes are anticipated by landowners and managers and the circumstances under which those outcomes are likely to be sufficiently valued to produce a change in management practice. Several thorough reviews

New residential subdivisions in former ranchland, Eagle Mountain, Utah. (Photo: Mark Brunson) have been produced over the years (e.g., Nowak and Korsching 1983; Clearfield and Osgood 1986). Most recently, Prokopy et al. (2008) reviewed 55 separate studies over a 25-yr period that explored adoption of agricultural best management practices (BMPs) in the United States. Their goal was to identify general trends in how adoption of conservation practices is related to social-psychological, enterprise-based, and social and economic factors. Most of the reviewed studies focus on soil, nutrients, and pest management; very few focused on the water or livestock management practices pertinent to grazing lands. Nonetheless, their findings offer general guidance about the role of anticipated benefits in the implementation of practices.

The variables most strongly associated with adoption of BMPs were attributes of the decision maker or of the farm/ranch operation: demographic factors, such as income and education; access to information, capital, and social support; and farm size (Prokopy et al. 2008). Producers' awareness of environmental problems and their overall environmental attitudes were positively associated with BMP adoption, not surprisingly suggesting that farmers and ranchers are more likely to adopt conservation practices if they believe that conservation is important.

Rogers (2003), whose theories on innovation adoption and diffusion have been highly influential in many fields including agriculture, identified three categories of factors that affect adoption rates: attributes of the potential adopter, the adopter's social system, and the innovation itself. In this chapter, we are most interested in the latter category, as it encompasses the perceived personal, family, or social benefits that potential adopters ascribe to conservation activities on grazing lands. Wejnert (2002) further divides the pertinent innovation attributes into two decision metrics: the degree to which the benefits of a practice are thought to outweigh the costs and the degree to which the positive or negative consequences of adoption accrue to the private individual versus the public good. Both criteria relate to producers' beliefs about personal, social, and economic factors as well as environmental benefits of a practice under consideration for adoption. Perceived costs can be psychological or social, just as are benefits; for example, Grigsby (1980) argued that one of the most significant barriers to innovation among ranchers is a belief that the innovation somehow threatens their ranching lifestyle.

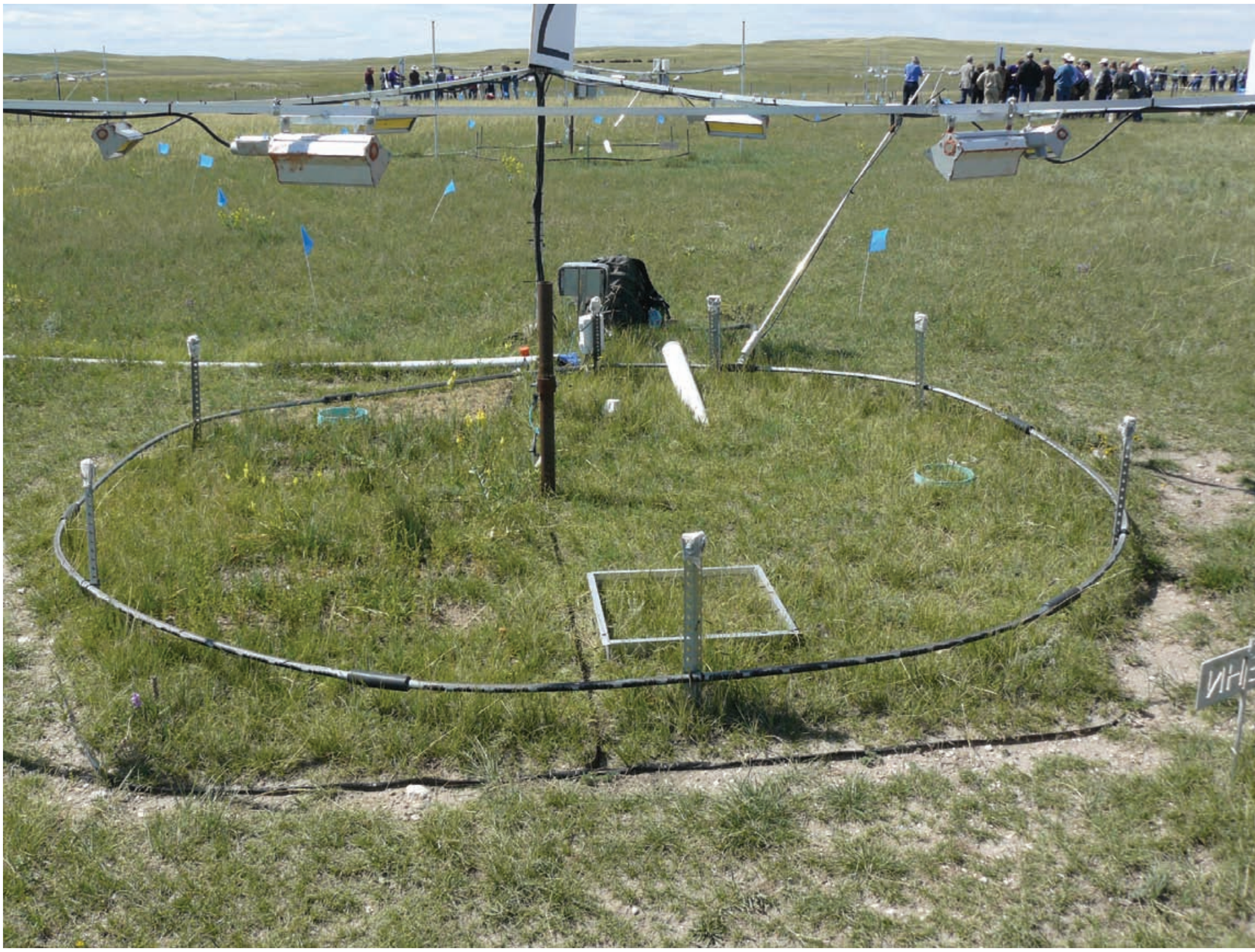
INNOVATION–ADOPTION STUDIES

Innovation–adoption studies abound in agriculture. Most of these focus on crop producers, but a number of researchers have explored practices recommended by the USDA-NRCS for grazing lands conservation. In this section, we describe more general studies that may include multiple practices; practice-specific research is described in subsequent sections.

In southwestern Oregon, Habron (2004) found that landowners implemented upland *conservation practices* such as off-stream *livestock* water developments and rotational grazing more often than fencing or tree planting in riparian areas. Key influences on whether producers adopted any practice at all were whether they used irrigation, shared management decisions with a spouse, believed in scientific experimentation, and discussed *conservation* with others. The key factors predicting adoption of specific BMPs depended on the kind of *practice* implemented.

A Utah study asked ranchers who have reputations as innovators what outcomes had led to their adoption decisions (Didier and Brunson 2004). One of the most influential outcomes was social: interviewees often reported that they were motivated by a desire to demonstrate stewardship to federal land managers and/or the public. The authors did find that innovation attributes were important to ranchers, especially in the negative. For example, interviewees reported that they or neighbors had rejected conservation practices because of perceived poor cost-to-benefit ratios or difficulty in pilot-testing a practice before full adoption. Brush management was cited as an example of the former barrier, while prescribed grazing—especially in the form of a short-duration rotational grazing system—was typically noted as an example of the latter.

Barao (1992) surveyed Maryland producers who had attended an extension field day



Measuring CO_2 and temperature effects, High Plains Research Station, Cheyenne, Wyoming. (Photo: John Tanaka)

to learn which demonstrated practices, if any, were subsequently adopted. Grazing management was the most common practice adopted (34%); livestock nutrition improvements, such as changing pasture species composition or analyzing forage/feed, were less commonly adopted. Perceived outcomes of the practices were not found to be as important to these decisions, however, as the ease with which the practice can be learned and the results of a change can be observed. In another assessment of an extension program, a Livestock Systems Environmental Assessment tool, Koelsch et al. (2000) found that producers who used the tool were most likely to cite a desire for improved environmental stewardship as the most important reason for doing so.

Nonadoption

A few studies have taken the opposite approach to understanding adoption decisions, and these also may prove useful—if people are not adopting conservation practices because they do not consider them worthwhile, that would suggest that such practices are not thought to provide personal benefits. However, Gillespie et al. (2007) found in a survey of 1 700 Louisiana beef producers that the most influential reasons for nonadoption of 16 BMPs were because they felt the practice would not work on their property or were unaware of the practice. Similarly, Prokopy et al. (2008) found that access to information influenced likelihood of adoption. Thus, knowledge of USDA conservation programs can also be an impediment to adoption. In

Kansas, Smith et al. (2007) found that 80% of survey respondents knew about EQIP; 31% participated in the program. Those figures are considerably higher than in an earlier study by Cable et al. (1999), who found that only 54% of survey respondents were aware of state or federal cost-share programs for private land conservation practices. For those who are aware of these NRCS cost-sharing programs but do not participate in them, lack of interest in conservation was not a significant influence on nonadoption; instead, ranchers tended to cite perceived regulatory impacts and paperwork as important reasons not to enroll (e.g., Didier and Brunson 2004; Smith et al. 2007).

Geographical Regions

Most of the studies described here pertain to a specific region, such as brush management in Texas (Taylor 2005; Kreuter et al. 2008) or prescribed burning in the desert Southwest (Sayre 2005). Some results of these studies are likely to be applicable to grazing lands nationwide, especially those relating to characteristics of land managers themselves, such as the importance placed on conservation as a management goal or sociodemographic and information access influences on adoption.

However, geographic factors are likely to influence landowners' beliefs about the perceived outcomes of conservation practices and thus the likelihood that those practices will be implemented. For example, Regen et al. (2008) found that protecting wildlife habitat and prairie restoration were important issues for a majority of landowners in a region straddling the Iowa–Missouri border. Most respondents reported using brush management practices to control eastern red cedar, but only 25% used prescribed burning, a practice frequently recommended for control of this species. In contrast, 38% of respondents to a survey by Kreuter et al. (2008) had used prescribed burning, mainly to control brush. Similarly, Liffman et al. (2000) found that 25% of landowners in Alameda and Contra Costa counties, California, had used prescribed burning in the previous 5 yr, while 34% of those in Tehama County had done so. In both the Midwest and California situations, a likely explanation for lower burning rates is likely to be the juxtaposition of grazing lands with other land uses (cropland in Iowa and Missouri and rapid exurban development in Alameda and Contra Costa counties), whereas burning is less likely to pose liability and permitting difficulties in areas where grazing lands dominate.

Elk in the Pequop Range in Elko County, Nevada. (Photo: Tim Torell)

Prescribed Grazing

As was noted previously, researchers studying HM have found that practitioners report improved quality of life as a result of participation in their decision-making program (Stinner et al. 1997; Montagne and Orchard 2000). A similar conclusion was derived from the Sustainable Grazing Systems program in southern Australia. This program was established in 1996 to address declining pasture productivity. Nearly 10 000 Australian livestock producers received training and new skills, participated in demonstrations, and integrated management and goal setting into their ranching operations, similar to those participating in the HM program.

It is not known whether perceived improvements from program participation was a result of having employed a prescribed grazing system or some other factor associated with the learning experience but in the Montagne and Orchard (2000) study, interest in alternative grazing systems was one of the most frequently cited reasons for change. Overall, ranchers reveal positive ecological changes on the land and increased economic as well as personal satisfaction. In Minnesota, a psychological benefit was reported in collaborative research project by farmers and scientists that found that rotational *grazing not only* improved soil, pasture, and stream quality but also boosted the confidence of the farmers in their ability to employ more sustainable *grazing* practices (Badgley 2003). Again, it is not clear whether it was the grazing system or the collaborative process that improved farmers' confidence levels.

Collective interests of groups can also benefit from conservation practices beyond the individual benefits. Armstrong and Warner (1992) reported that adoption of a rotational grazing strategy by the Walker River Tribe in Nevada promoted tribal interests by benefiting all natural resources. This assumes, however, that resource benefits truly exist. As noted above, Briske et al. (2008) states that even if evidence for benefits from rotational grazing is inconclusive, many livestock producers believe that such benefit exists. They suggest that "personal goals and values . . . are inextricably integrated within grazing systems, and they are likely to interact with the adoption and operation of grazing systems to an equal or greater extent than the underlying ecological processes" (p. 10). A basic theory of psychology holds that people are motivated to behave in ways that are consistent with their beliefs, and when evidence suggests that those behaviors are unhelpful, they may tend to reject the evidence rather than reject the belief (Festinger 1957). Consumer researchers (e.g., Mano and Oliver 1993) have applied this theory to explain postadoption satisfaction levels, suggesting that the act of having adopted a new product or behavior predisposes one to evaluate it positively. If we apply this to rotational grazing, one explanation for the continued perception of realized benefits is that there are psychological rewards associated with doing so.

Moving cattle in the Blue Mountains, Oregon. (Photo: John Tanaka)

Brush Management

While brush management is a recognized NRCS conservation practice, it may or may not be considered "conservation," depending on the purpose of the practice and the historic and current conditions where it is implemented. For example, removing encroaching junipers to improve wildlife habitat and water availability in central Texas might be considered conservation, whereas removing all sagebrush from a native shrub–steppe community and planting a nonnative forage grass would not. Even in the former case, brush removal might constitute "conservation" if intended to benefit black-capped vireo (*Vireo atricapilla*) yet detrimental if the site is occupied by golden-cheeked warblers (*Dendroica chrysoparia*). Unfortunately, brush management implementation studies do not necessarily distinguish between management for conservation and management for other purposes.

Kreuter et al. (2001) surveyed Texas county extension agents to assess landowner interest in and adoption of Brush Busters, a collaborative extension/research program. Respondents reported that the landowners with whom they work perceive the program to be an "inexpensive, convenient, safe, effective, and predictable method for controlling brush." Interestingly, the authors recommended that to increase adoption rates, technology transfer professionals should emphasize the short-term economic benefits of Brush Busters rather than the long-term environmental benefits. This suggests that ranchers who implement this practice may not obtain personal benefits from implementing brush management as a conservation practice; rather, the benefits may accrue to society. Similarly, Thurow et al. (2001) found that economic factors were associated with ranchers' willingness to enter into a brush control cost-share contract but that conservation motives were not. Further evidence that conservation alone cannot motivate brush management comes from Olenick et al. (2005), who found in a 2003 Texas survey that landowners generally held favorable views toward programs that would reduce brush cover to increase water yields or to improve wildlife habitat, but they disapproved of programs that would encourage the proliferation of woody plants in an attempt to increase atmospheric carbon sequestration. Landowner attitudes were also associated with the voluntariness and flexibility associated with any proposed program to enhance ecosystem services.

Prescribed Burning

Prescribed burning is a practice where the conservation benefits are offset by potential risks, including a loss of forage, regulatory difficulties associated with smoke and burning permits, weak legal protections against liability, and potential escape of the fire onto a neighboring property (Liffman et al. 2000; Brunson and Evans 2005). Ranchers may believe that prescribed fire would benefit their land but are reluctant to implement. For example, Sayre (2005) studied eight locations in southern Arizona and New Mexico where prescribed burning was part of wildlife conservation efforts on grazing lands. He found that while interest in restoring fire to the landscape was high, use of prescribed burning was limited by trade-offs between conservation goals and forage availability and by real or perceived regulatory scrutiny. Burning was most feasible where there were institutional structures that allowed for collaborative management across ownership boundaries.

In the Great Plains, an institution has arisen for just that purpose. Prescribed burning cooperatives offer landowners a chance to learn from peers how to apply fire safely and effectively and reduce liability concerns (Taylor 2005). In Texas, Kreuter et al. (2008) found that members of a large cooperative had more positive attitudes than nonmembers about the ecological role of fire and the use of prescribed fire. The authors suggest that the group not only offers opportunities to learn and reduce liability but also promotes cooperative behavior that can benefit a ranching community. Thus, in the context of a prescribed burning association, there may be a social benefit to implementing this conservation practice.

Rangeland Planting

Few authors have specifically addressed the social aspects of rangeland planting as a conservation practice, but these provide interesting insights as to the influence of changing societal norms. As noted above, seeding of crested wheatgrass was one of the most common vegetation management practices in the western United States during the 1950s and 1960s. By many estimates, it was also one of the most economical practices because of the species' competitiveness, early grazing use, and productivity. However, as the use and restoration of native plants has grown more popular, societal opinion has turned against the idea of replacing native rangeland with a monoculture of an exotic species that can persist for decades (Johnson 1986; Conner and Bach 2008). Negative characterizations of this highly adaptable and productive forage species have some critics who see planting crested wheatgrass as an ill-advised subsidy of ranching on public lands (Abbey 1988; Hess 1992) and those who complain that it has reduced sagebrush habitat for Greater sage-grouse (Connelly and Schroeder 2000). While it is still used on private ranches, crested wheatgrass seeding in public land grazing allotments has declined (Conner and Bach 2008), often restricted to areas vulnerable to invasion by exotic annual grasses where rapid revegetation is needed for site stabilization after wildfire. Even then, plans may call for the use of "assisted succession" (Cox and Anderson 2004) to replace crested wheatgrass with native perennial species as soon as is practical.

Young and Clements (2009, p. 179) described these largely social pressures affected rangeland management:

> The politics of bureaucratic survival called for saying and doing as little as possible. Public land managers learned never to propose a seeding to increase forage supplies because government agencies, environmentalists, and archaeologists would descend en masse demanding a full environmental impact statement. Even wildfire burns were not seeded. When public pressure dictated seeding some very important habitat, the seed mixture was composed of species that had no chance of establishment and was seeded by aerial broadcasting on unprepared seedbeds. Young managers who tried seeding and failed were careful never to try again.

Riparian Herbaceous Cover

Considerable research has explored attitudes toward riparian restoration and protection, but almost all has focused on tree cover rather than herbaceous cover. For example, Lucht (2007) analyzed interest in adoption of agroforestry and conservation practices, including riparian planting, among agricultural producers, resident nonfarm landowners, and absentee nonfarm landowners. She found comparatively high interest but low knowledge levels compared to other practices. Absentee nonfarm landowners had the highest level of interest.

Public information campaign in an area undergoing transition from farm/ranch land to small-acreage subdivisions, Arimo, Idaho. (Photo: Mark Brunson)

Ryan et al. (2003) found that Michigan farmers were motivated primarily to adopt conservation practices along riparian zones, not for the economic returns it would provide but because of their strong attachment to the land and their desire to convey the message that they are good stewards of the land. They noted that strategies for conservation must respect farmers' attachment to the land, the desire to practice good stewardship while deriving income from the land.

One study that did focus on nonwoodland planting was by Smith et al. (2007), and they reached conclusions similar to Ryan et al. (2003) about developing strategies for conservation. They found that Kansas farmers and ranchers were less likely to plant riparian filter (buffer) strips than to employ other forms of best management practices because filter strips must be enrolled into the Continuous Conservation Reserve Program, thereby incurring restrictions on haying and grazing use.

Upland Wildlife Habitat Management

Upland wildlife habitat management, more so than other practices described here, is likely to enhance the use value for landowners as well as the bequest or existence values. The benefits that can be realized from wildlife are many and well known (Manfredo 2008). Because habitat enhancement also enhances the likelihood of being able to successfully view or hunt wildlife, many landowners will improve their land for wildlife. For example, when Cable et al. (1999) surveyed 900 Kansas agricultural producers about wildlife and riparian areas, they found that more than a third of respondents reported that they had idled land or changed management practices specifically to help wildlife. The most common "extremely important" motivations for doing so were to preserve wildlife for future generations (55.6%) and because the landowner enjoyed watching wildlife (51.7%). A nationwide survey of farmers and ranchers by Conover (1998) found that about half of respondents manage their properties to enhance attractiveness to wildlife.

Both personal and monetary benefits influence decisions to implement wildlife habitat improvements, and it can be difficult to separate the two. Van Kooten and Schmitz (1992) found that agricultural producers participating in a waterfowl habitat enhancement project in western Canada held more positive attitudes toward wildlife than nonparticipants and therefore can be assumed to obtain personal benefits from participation, but positive attitudes alone were not sufficient to motivate habitat improvements in the absence of economic incentives. While this study was not completed in the United States, other studies conducted in this country (e.g., Troy et al. 2005) support the idea that the social or psychological benefits of wildlife habitat enhancement typically do not offset costs of doing so without some sort of economic incentive.

Incentive to entice landowners to adopt wildlife management programs may take several forms. Conover (1998) reported that nearly 80% of farmers and ranchers had encountered some sort of wildlife damage on their properties. This is one of several impediments to implementation of wildlife habitat improvements. Cable (2002) reported that while Kansas agricultural producers believe that it is important to protect wildlife habitat, fewer than half had set aside any of their property for wildlife. The primary reasons were fear of increased trespassing by hunters if their land became known as especially attractive to wildlife and the cost of idling any of their land. Costs are even greater when the wildlife may be protected under the federal Threatened and Endangered Species list (Brook et al. 2003; Elmore et al. 2007). Thus, any positive personal outcomes associated with implementing the upland wildlife habitat management practice must be great enough that a landowner is willing to take on the risk of other consequences, such as damage, trespass, or increased regulatory scrutiny and reduced management flexibility.

Incentives for wildlife habitat management on private land can be nonmonetary as well as monetary. Programs such as the Safe Harbor and Candidate Conservation Agreement with Assurances programs of the U.S. Fish and Wildlife Service as well as other collaborative land management efforts that exist across the West seek to protect landowners against regulatory risks in exchange for taking actions on behalf of wildlife (Belton 2008; Womack 2008). Belton (2008) surveyed members of local working groups that attempt to maintain habitat for the greater sage-grouse. Participation in these efforts can promote cohesion among landowner neighbors and enhance cooperation with government agencies, but the "peace of mind" that such efforts are intended to provide are dependent on trust levels in the agencies responsible for wildlife management and protection. As before, Belton's (2008) research suggested that people are more likely to participate if they receive monetary compensation for providing habitat for grouse.

VALUE OF ECOSYSTEM SERVICES

Types of Values

Ecosystem services benefit society in numerous and diverse ways. We can differentiate between those goods and services that are place bound (in situ) and those that can be derived from multiple locations (ex situ). There are a variety of classifications, including consumptive and nonconsumptive, market and nonmarket, primary and secondary, and in situ and ex situ (Brown et al. 2007; Cooper and Dobson 2007; Breckenridge et al. 2008). The basic issue is how to account for all the benefits and costs associated with the services derived from rangeland ecosystems. As noted earlier, each of the conservation practices can potentially produce different kinds, qualities, and amounts of these goods and services, depending on location, natural potentials, current states, and other factors.

Brown et al. (2007) used a traditional approach by dividing ecosystem goods into nonrenewable and renewable. Nonrenewable goods included rocks, minerals, and fossil fuels, while renewable goods include wildlife and fish, plants, water, air, soils, recreation, and aesthetics. Ecosystem services include purification of air and water, nutrient cycling, maintenance and renewal of soil and soil fertility, pollination of crops and natural vegetation, dispersal of seeds, maintenance of regional precipitation patterns, erosion control, biodiversity maintenance, control of pests affecting plants or animals, protection from the sun's harmful UV rays, partial stabilization of climate, moderation of temperature extremes and the force of winds and waves, and mitigation of floods and droughts. In a listing of primary and secondary benefits from using pesticides, Cooper and Dobson (2007) divided primary benefits into agricultural production, energy needs, and preventing problems and secondary benefits into farming communities, national issues, and global issues. The interaction among all these benefit categories is complex and makes such separation difficult (see the section "Ecosystem Services").

Economic Valuation

In valuation of nonmarket ecosystem goods and services, there are few acceptable methods used in the literature. It is important to note that all these seek to estimate what a person or household would willingly pay to have that good or service or to put a value on damages from losses or costs avoided (Olewiler 2004). The comparability of these values with goods and services that are actually paid for out of the individual's income remains a question. In other cases, for the experiments to have validity, the consumer being questioned needs to have a very clear idea of what goods or services are at stake.

Before delving into the valuation question, it is important to note that the nature of ecosystem goods and services that are not traded in private markets lead to what is known as market failure and results in goods and services being either undersupplied or overused (Lant et al. 2008). Goods and services that are provided through private markets involve feedbacks that can potentially provide efficient levels of production. Rules or incentives put in place to deal with market failures can lead to inefficient levels of production and essentially require the regulating agency to guess at market-clearing prices or quantities. It is important that economists work closely with ecologists and other specialists in a truly collaborative process in their efforts to estimate values (Heal and Barbier 2006).

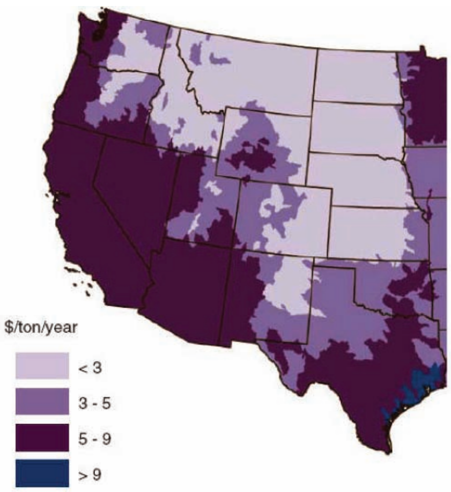
In a study that estimated the value of "diversity in biodiversity," Christie et al. (2006) found that the public did not generally understand different attributes of biodiversity even as they valued biodiversity itself. The public was also found to be relatively indifferent as to how biodiversity was achieved, but most attributes of biodiversity examined had positive values. Of course, biodiversity as an attribute is a multifaceted concept that occurs at many scales (West 1993), and one has to carefully define what is meant by the term before it can be valued.

It is also important to recognize who is likely to be the recipient of the benefits or who is setting the value (Burger et al. 2008). As they note, when valuing an individual species that does not have immediate or direct value, it is usually conservationists or regulators that set the value. On the other hand, when the individual species has a direct value to individuals, values are set through businesses, social scientists, or others with a direct connection to that species. The other types of resources they examined were the value of ecosystems to human communities and intact ecosystems with ecological, aesthetic, and existence values to people. Burger et al. (2008) also suggests specific economic value estimation methods that are appropriate for each type of environmental good or service (Table 5). The valuation methods that may be used in different situations are travel cost,

TABLE 5. Economic estimation methods suggested by Burger et al. (2008).

Type of ecosystem good	Methods
Resources themselves	Use survey of selected businesses
Specific resources for individuals	Use sample surveys to estimate direct values; estimate direct, indirect, and induced values using regional economic models
Resources for communities	Estimate replacement value, insurance costs, regional economics
Intact ecosystems	Use contingent valuation to estimate existence values; estimate regional economics

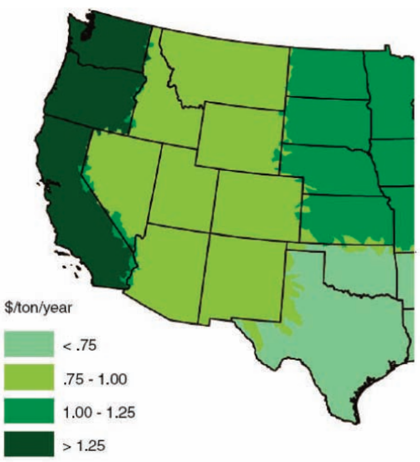
FIGURE 1. Water erosion benefit estimates (adapted from Hansen and Ribaudo 2008).



HUC-level[1] water-erosion benefit categories are: *reservoir services, navigation, water-based recreation, marine fisheries, freshwater fisheries, municipal industrial, steam electric, irrigation ditches, flood damages, soil productivity, road ditches*, and *municipal water treatment*. Only *reservoir services, navigation*, and *water-based recreation* are estimated at the HUC level. Other values are based on FPR-level[2] estimates.

[1]HUCs are watersheds defined by the U.S. Geological Survey's 8-digit hydrologic unit codes.
[2]FPRs are USDA's multi-state Farm Production Regions.

FIGURE 2. Wind erosion benefit estimates (adapted from Hansen and Ribaudo 2008).



HUC-level wind erosion benefit categories are *dust cleaning* and *soil productivity*. Values are based on FPR-level[2] estimates.

[1]HUCs are watersheds defined by the U.S. Geological Survey's 8-digit hydrologic unit codes.
[2]FPRs are USDA's multi-state Farm Production Regions.

contingent valuation, hedonics, cost-based approaches, and factor-income approaches (Swinton et al. 2007). It is beyond this chapter to discuss these methods, but suffice it to say that each method is applicable to different situations and that the estimates derived have varying levels of confidence and comparability (Randall 2007).

In the only national-level estimate of conservation values with regional applications, Hansen and Ribaudo (2008) estimated the values of various soil conservation benefits. Although their focus was primarily on the impacts on end uses of water, they do provide a value for reducing soil erosion. Different values were derived for water and wind erosion, and these values varied by region (Figs. 1 and 2). Huszar (1989) estimated that off-site economic costs exceeded on-site costs from wind erosion and concluded that if public action were to be warranted, it should be aimed at reducing off-site impacts.

One of the issues in many valuation studies using willingness-to-pay measures is that while values are estimated, they do not address how values will change as supply and demand for that ecosystem good and service change. As an example, Loomis (2005) estimated values for outdoor recreation on public lands based on numerous studies. While these values may be valid, local conditions and the quantity of these goods and services nearby will affect these values.

In a study to compare the value of ecosystem services from restored versus native land, Dodds et al. (2008) estimated values based on a broad literature search. Estimated values for rangeland regions are shown in Table 6. It is important to note that they generally estimated lower values for restored lands. The implication of these values may be that society values the maintenance of native rangelands more than restored or that restored lands have not been shown to be as productive in producing these ecosystem goods and services as intact native rangelands.

Aesthetics are often cited as one of the important ecosystem services derived from rangelands. Most studies dealing with aesthetics have used contingent valuation. In one study that sought to actually quantify what ranch buyers would pay for a "quality-of-life" amenity that comes with owning the ranch, Torell et al. (2005b) found that ranch location, its scenic view, and the desirable lifestyle had more of an influence on ranch price than its potential to produce income. One of the implications of this is that while ranch owners may not respond to conservation practice implementation to improve ranch income, they may respond to practices that enhance these factors.

Social Valuation

There is a small but growing literature on the valuation of ecosystem goods and services with metrics other than monetary valuation. These are presented here under social valuation since they seek to find alternative metrics for evaluating trade-offs among different products or outcomes.

One such approach has been to define a social-ecological system and solving a system of structural equations (Asah 2008). Finding the relationships between social systems and ecological systems remains a challenge. This method sought to relate a management goal with social knowledge of ecological responses. The results, however, are difficult to extrapolate because of the "place-specific nature of human-environment interactions" (Asah 2008).

Another system uses what are termed "holistic ecosystem health indicators" to integrate ecological, social, and interactive indicators (Munoz-Erickson et al. 2007). The ecological indicators use biophysical measurements; the social indicators use demographics, economics, and quality-of-life metrics; and the interactive indicators use land use practices, policy, and collaboration measurements. Each of the measurements is weighted to derive a measure of overall holistic ecosystem health.

RECOMMENDATIONS

Use of social and economic information can be incorporated into the NRCS conservation planning processes in a variety of ways. NRCS has done a commendable job of considering the ecosystem services in their planning and management processes. Acknowledgment of potential ecosystem services in the Conservation Practice Physical Effects Worksheets is an important step forward. We encourage the NRCS to continue to develop these worksheets, to refine their physical effects and rationale, and to define them on a more site-specific basis (i.e., by major land resource areas or ecological sites, as appropriate). We further recommend that the economic and cultural categories be expanded values of

TABLE 6. Estimated values of ecosystem services per native hectare per year (in 2005 dollars) (Dodds et al. 2008).

	Great Plains		North American Deserts	
Ecosystem service	Native	Restored	Native	Restored
Gas regulation	7	6	—	—
Disturbance regulation	7	7	2	1
Water supply	28	19	85	25
Nutrient cycling	22	15	60	18
Soil erosion control	241	175	237	65
Commodities	3 853	2 490	—	—
Biodiversity	46	50	—	—
Recreation	1 003	1 003	16	16

ecosystem services and social impacts at the individual, ranch or farm, and community levels.

The method currently being used by NRCS to evaluate the benefits and costs contains the main features present in all the standard economic analyses (NPV) with the addition of values for selected ecosystem goods and services identified in each practice's description (H. Gordon, personal communication, 2008). The NRCS should continue to refine how it incorporates ecosystem goods and services into its conservation practice analysis. While the current economic analysis spreadsheet incorporates factors from the Physical Effects Worksheets and attempts to place monetary values on each item, justification for those monetary values needs to be developed and standardized. While the approach is sound, without a sound basis, values for the various ecosystem goods and services can easily be manipulated to justify any project. It should also be recognized that a complete benefit-to-cost assessment of selected conservation projects is not possible until valid estimates of economic value is assigned to presently unvalued ecosystem services.

The NRCS should seriously review its cost-share policies and requirements. If they are truly designed to pay for that portion of a conservation practice that benefits society, then the percentages should reflect that split. There are cost-share options that could be examined.

We recommend serious consideration of the first interpretation of cost share below:

1. Set cost share based on the split between expected private and public benefits. In some instances, this could range from 0% to 100%. Thus, rather than trying to estimate ecosystem goods and services values for each project, they could be determined for each practice through setting the appropriate cost share. Private landowners could then determine if their share of the total project cost would be covered by changes in their private benefits over the life of the project.
2. Determine whether conservation is a priority and use cost-share amounts to promote adoption of those practices with the highest public benefit. If the only purpose behind cost share is to get the private entity to become invested in the practice, the level of investment should be examined with a view toward the impacts that higher, the same, or lower cost shares would have on adoption rates.

KNOWLEDGE GAPS

Knowledge gaps in the social and economic realms of conservation practices are numerous. Here we cover a few specific research needs for economics, ecosystem services, and social science.

Economics of Conservation Practices

Since most rangeland conservation practices will be implemented by private ranchers throughout the western United States, it is necessary to understand how changes affect the overall economics of the ranching operation. Economic analysis of conservation practices can start with a basic efficiency estimate, such as present net worth, benefit-to-cost analysis, or internal rate of return. While that will provide a basic estimate of profitability, ranchers also need to understand how the change will affect their entire operation, and society needs to understand the larger-scale social benefits and costs.

Bighorn sheep near Gabbs, Nevada. (Photo: Tim Torell)

In terms of the actual conservation practice, the methodology for economic analysis as related to livestock production is well understood. Research that is needed at this level is knowledge about the physical responses (e.g., additional production and seasonality), the costs of inputs and outputs, and the timing of benefits and costs. Caton et al. (1960) noted that early attempts by agricultural economists participating in a West-wide regional research project to quantify the economics of rangeland management practices were hampered by a lack of response data. Potential benefits of many improvements could not be assessed because long-term studies had not been undertaken to quantify the forage and livestock responses that were realized from the various practices. This limitation continues.

The economic impacts of the conservation practice on the entire ranching operation require additional research. Most of the relationships are biological in nature (e.g., livestock production and forage production) and require knowledge about annual cycles and longer-term responses. Each livestock production cycle will have its own unique attributes, depending on geographic location, type of animal species, and production goals. In order to model within a year, these factors must be understood. When management changes with resulting changes in herd size, it may take several years for the ranch to come to a new equilibrium herd size. On the forage side, within-year variation of production affects the amount of feed available for the herd. Within the year and across years, temperature and precipitation will also affect the amount of feed available for livestock. Many of the conservation practices will affect the potential amount of forage production.

Social Aspects of Conservation Practices

There are many knowledge gaps related to social aspects of conservation practices.

The problem with many of these is that the knowledge that is needed is place based. What might be socially acceptable for a national program may not be at the local level. While there have been a few studies looking at adoption of conservation practices, these need to be done at more locations and specific to NRCS conservation practices on rangelands. In addition to location, such information for individuals, operations, and social systems will be useful in designing programs.

Ecosystem Goods and Services Valuation

Quantifying societal benefits and the economic value of previously nonquantified ecosystem services are the areas where economic evaluation of conservation practices is most lacking. As noted earlier, livestock production benefits do not justify the total cost of many conservation practices (McBryde et al. 1984; Lee et al. 2001; Torell et al. 2005a; Skaggs 2008). Economic evaluations are incomplete, and the assumption is made that ecosystem services not quantified in the analysis offset the excess cost of the practice not justified from marketable goods and services. This may not be the case. Only by quantifying and assigning economic value to selected ecosystem services that are currently only qualitatively noted can a complete economic assessment be made. Quantifying benefits accruing to the public at large could also be used to justify cost-share percentages on a case-by-case (or location-by-location) basis. This would mean an expanded use of nonmarket valuation techniques (see Champ et al. 2003). We note that while it would be advantageous to know how people value each of the goods and services produced by different conservation practices, it is probably neither likely nor feasible. Yet expanding the economic analysis to quantify all potential benefits has several important implications. First, the quantification may show a very positive benefit-to-cost ratio from society's point of view, suggesting that even more should be done. In other cases, it may demonstrate that the conservation practice is not justified. If not valued by society, based on economics, transfer payments and willingness to pay, inaction, and a deteriorated ecosystem may be the preferred state. Dismal federal and state budget situations highlight that trade-offs exist.

Most of the knowledge base on the values of nonmarket goods and services is time and space specific given the methodologies currently in use. Aggregation of numerous studies can provide value ranges that various researchers have estimated using methods such as travel cost, contingent valuation, or hedonic models. Whether these are appropriate for the NRCS to use in evaluating their conservation practices through extrapolation is a question for investigation. The basic knowledge gaps are the values for each ecosystem good or service in each location and at the specific time.

CONCLUSIONS

Use of Economic Information in Resource Planning and Management

Economic values for market and nonmarket ecosystem goods and services will vary by location and time. What might be available at the scale necessary for ranch-level planning are indicators of relative values. There have been many studies of individual benefits for specific goods and services. Those that exist in markets provide what people are truly willing and able to pay. Some methods, such as hedonic pricing and willingness-to-pay studies, can help determine values of specific characteristics.

Relative values of the various goods and services can provide information to planners and managers if they are collected appropriately and in a consistent manner. There are numerous peer-reviewed articles for many of the conservation practices that have done an economic analysis of the specific practice comparing the cost of the practice with the estimated market benefits. We have not reviewed those studies because they are not particularly enlightening for this project. The method currently being used by the NRCS to evaluate the benefits and costs contains the main features present in all the standard analyses with the addition of values for the various ecosystem goods and services identified in each practice's description (H. Gordon, personal communication, 2008).

If economic feasibility of various conservation practices is important, it would be prudent for NRCS to become seriously involved in finding ways to assess the economic value of ecosystem goods and services beyond those found in

"Quantifying societal benefits and the economic value of previously nonquantified ecosystem services are the areas where economic evaluation of conservation practices is most lacking."

Old cabin and scenic view, Owyhee County, Idaho. (Photo: John Tanaka)

the marketplace. As noted, when considering only forage value or livestock production from rangelands as the primary benefit, many conservation practices will not show a favorable benefit-to-cost ratio for conservation programs. As the ecosystem goods and services are considered and valued, the NRCS can allocate the conservation practice costs to the private landowner and to taxpayers through its cost-share mechanism based on the expected proportion of benefits going to the different parties. For example, if a prescribed grazing practice does not increase livestock production and net returns but produces significant social benefits, the taxpayers may be allocated a larger proportion of the costs of implementation than if livestock benefits were higher. Some of the social benefits may go to the private landowner, and this should also be taken into account (e.g., maintaining a way of life). At present, while cost-share mechanisms would seem to implicitly recognize these social benefits, there is little information to justify the cost-share percentages on a case-by-case (or location-by-location) basis. It is not reasonable to assume that the social benefits from a given conservation practice is the same everywhere or that they offset substantial practice costs in many cases.

Quantitative estimates of the value of ecosystem service are largely nonexistent, but quantification of traditional market values are lacking as well. Efforts to quantify the economics of rangeland management practices have been hampered by a lack of response data. Long-term range and grazing studies that monitored the production response of management practices were found to be a major shortcoming of economic assessments of rangeland management practices that have continued since the 1960s. These economic assessments of rangeland management practices are based on very limited data and usually use simulated biophysical data. Long-term range and grazing studies are becoming even less common. Response relationships among conservation practices and other ecosystem goods and services are even rarer or nonexistent.

Use of Social Information in Resource Planning and Management

Social values and attitudes have clearly impacted adoption of rangeland management practices and how rangeland policy and management has progressed. As an example, Young and Clements (2009, p. 178.)

concluded, "Public rangeland management agencies did not drop the use of herbicides because they were afraid of the environmental consequences of using pesticides; they dropped them because they were afraid of comments from a highly vocal but not necessarily knowledgeable portion of the general public. Congress stopped appropriating money for the improvement of publicly owned rangelands to avoid criticism from environmental groups." The numerous other examples described above clearly show that social factors have motivated land managers to behave in ways beyond profit-maximizing behavior.

If social information is going to be used in resource planning and management, social indicators need to be added to the list of benefits along with a description of how to use and interpret the indicators. Our assumption is that most resource managers do not know what indicators would be appropriate or how to use them in decision making. The significant lack of social research on the effects of the conservation practices on landowners will make implementation of this recommendation difficult in any sort of quantitative manner.

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