ACCOUNTING FOR NATURE'S VALUE

WITH RANGELAND CONSERVATION PRACTICES IN THE WESTERN RANGE AND IRRIGATED REGION

EARTH ECONOMICS

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1. EXECUTIVE SUMMARY

The term "nature's value" refers to the observation that healthy ecosystems provide a broad range of services—such as air quality, water storage and filtration, and biological control—which benefit local, regional, and even global natural and human communities. Integrating the economic value of such services—commonly referred to as "ecosystem services" or "nature's contributions to people"—into land use planning and resource management could result in more informed decisions about resource allocation and the strategies needed to balance agricultural productivity and ecosystem health. Yet full consideration of ecosystem services in conservation planning and policy decisions is often limited by the lack of comprehensive, rigorous empirical information regarding the economic value of the services provided.

The purpose of this report is twofold: to improve and expand the initial rangeland ecosystem service valuation framework set forth in Fletcher et al. (2020) and to demonstrate its replicability. We include more (and more detailed) data on the effects of conservation practices on rangeland health, based on published science. We also include additional conservation practices beyond those considered by Fletcher et al. (2020) in the analysis and improved benefit-transfer estimates developed by applying function transfer methods. We show how the framework can be expanded to rangelands managed by other federal agencies (e.g. the Bureau of Land Management (BLM)) and to a second land resource region (LRR)-- the Western Range and Irrigated Region (LRR D).

We focus on three NRCS rangeland conservation practices that predominate in LRR D—Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing—and thirteen ecosystem service benefits are provided by rangelands. We also incorporate BLM's "conservation treatments" that are very similar to the three NRCS conservation practices above. This approach integrates consideration of a broad range of potential benefits of conservation on local communities and economies, highlighting the range of data types, assumptions, and linkages required to produce rigorous ecosystem services valuation estimates in a comprehensive manner. It is also reflective of typical grazing operations in the region, which include a mix of public (BLM and U.S. Forest Service), private, and state lands (herein grouped with private lands).

This study reveals important data gaps and challenges to linking conservation practices with changes to ecosystem function and the value of ecosystem services. While there are limitations stemming from data availability and granularity, and critical assumptions about the relationships between elements of the framework constrain precision, the framework and estimates provide a broad sense of the economic importance of conservation actions. It also offers an example of how it is possible to use available data in a cost-effective manner to better understand programmatic effects on ecosystem services.

Linking management practices to changes in ecosystem services provisioning to estimate the economic value of conservation practices, offers NRCS a compelling way to communicate conservation successes and accomplishments to the American public, especially those farmers and ranchers who voluntarily implement conservation practices. Conservation success is typically reported in terms of acres-treated or numbers of practices applied, but such metrics rarely show how ecosystem services produce off-site public benefits. In addition to reporting acres treated, this framework—and associated value estimates—enables federal agencies to report that practices have improved certain ecosystem services that benefit downstream communities, or the range of per-acre values provided by voluntary conservation actions by ranchers.

This analysis relies upon available NRCS and BLM practice data, peer-reviewed research, and multiple assumptions about complex functional relationships to bridge gaps in existing research on ecosystem valuation, the impacts of conservation practices, and ecosystem health. These estimates suggest that rangeland conservation practices may significantly improve the ability of rangelands to produce an array of ecosystem services. It also identifies critical areas for future research to strengthen analyses of this kind. An improved understanding of the broader value of ecosystem services provided by conservation practices may support goals shared by producers who implement conservation practices and those living downstream and in nearby communities. This can lead to better-informed decision making and support innovative funding mechanisms to ensure that producers, their neighbors, and the broader public benefit from conservation practices.

The need to quantify the non-market value of nature's benefits has been recognized in several key pieces of federal legislation, agency handbooks, departmental memos, and even in national strategies. Estimates of the economic value of ecosystem services could be integrated into conservation planning and policy decision-making in several important ways:

- Improving field-level conservation planning through more comprehensive assessments of the potential practice benefits.
- Informing resource allocation towards and across conservation efforts, based on improved understanding of the benefits of conservation to local communities and economies.
- Broadening financial assistance programs to include incentive payments to producers for improving ecosystem functioning.
- Refining landscape-level assessment of conservation planning priorities based on a better understanding of complementarities across conservation practices.
- Making reporting metrics more robust to convey the breadth of voluntary conservation effects, beyond individual farms and ranches to downstream communities (and others) who benefit when ecosystem services are maintained or improved.

Implementing this framework, we estimate that between 2011 and 2020, Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing on private rangelands in LRR D increased the value of selected ecosystem services between $454 million and $1 billion overall, averaging $46 million to $109 million per year across both private and BLM-managed rangeland. **This represents an average increase of $45.70 per acre per year on private rangelands and $5.71 per acre per year on BLM-managed rangeland, compared to the period before these practices were applied.**

2. INTRODUCTION

Conserving the benefits provided by nature—such as clean and abundant water and healthy soils—is a goal of many federal agencies that play a role in managing rangelands in the United States. It is part of the vision of the Natural Resources Conservation Service (NRCS), the overall goal of the Bureau of Land Management's (BLM) rangeland management program. It has also been a key focus of many recent federal policies: Executive Order 14072, *Strengthening the Nation's Forests, communities, and Local Economies*; the *National Strategy to Develop Statistics for Environmental-Economics Decisions*; and OMB's *Guidance for Assessing Changes in Environmental and Ecosystem Services in Benefit-Cost Analysis*—all designed to support federal agencies in accounting for and valuing nature's benefits, among others.

However, the agricultural sector is unique in that while it has the potential to degrade natural ecosystems, it also presents many opportunities to design resource management strategies that incorporate ecosystem services—defined as the contributions of nature to human well-being—into decision-making and land use planning. Both NRCS and BLM have developed conservation practice standards to help fulfill conservation plan objectives and maintain rangeland health. A conservation practice is a management activity, or structural or vegetative measure, conducted to reduce the degradation of natural resources, including soil, water, air, plant, animal, and energy resources. Conservation practice standards outline the technology used, purpose, applicability, and requirements for use of the practice (NRCS 2023). Efficient and effective targeting and implementation of conservation programs, however, requires a comprehensive understanding of the conservation benefits associated with different resource management strategies—including ecosystem services.

The *Year One America the Beautiful Report*—an update on progress for the federal America the Beautiful Challenge—lists many NRCS and BLM achievements that furthered conservation goals in terms of acres treated or dollars invested. Such metrics rarely show how ecosystem services produced in one location can also produce benefits "beyond the fence line" to neighbors, nearby communities, and the general public. A full accounting of ecosystem services impacts—and the value of those impacts—is hampered by research gaps; the availability and granularity of data; and an incomplete understanding of biophysical, social, and economic interactions at multiple levels. Measuring the site-level outcomes of conservation practices on rangelands is difficult and costly to track. Over the years, the science around ecosystem services has been building and improving rapidly. Nonetheless, including quantification and monetization of these outcomes within decision-making processes broadens our awareness and understanding of the contribution of conservation practices to local communities and economies.

To advance our capacity to quantify changes in ecosystem services attributable to conservation practices, this study lays out a framework to link conservation practices to their impacts on nature, and to estimates of economic value of those impacts. Ecosystem service values are often left out of decision-making. As a result, the value of maintaining healthy, functioning natural systems is underrepresented in policy and planning decision-making. Including ecosystem services in conservation planning and reporting efforts can communicate the cost-effectiveness of conservation practices, so that ranchers who adopt best management practices can sustain both their livelihoods and healthy, productive ecosystems.

2.1 ECOSYSTEM SERVICES: RANGELANDS' BENEFITS TO PEOPLE

Nature—including rangelands—provides a wide range of goods and services that benefit individuals and communities at local, regional, and even global scales, such as air and water filtration, food production, natural disaster risk reduction, climate stability and resiliency, cultural and recreational experiences, and others. Collectively, these are referred to as "ecosystem services" or "nature's contributions to people." Many of these benefits accrue on-site to producers but may also pass beyond the fence line to those living nearby and downstream. Responsible management practices that support these ecosystem services provide benefits to more than just those who own or manage rangelands.

Without healthy ecosystems, many of the benefits provided by nature may need to be replaced by built infrastructure, often at greater cost, once construction, operation and maintenance, and eventual replacement costs are considered. Because ecosystems are living, adaptive systems, natural assets may be more resilient and less costly to maintain than built infrastructure. Acknowledging nature's economic contribution allows the consideration of nature-based solutions when evaluating the relative merits of investing in conserving natural systems versus infrastructure development, while raising awareness of the intrinsic connections between communities and these natural assets.

There are many frameworks by which to categorize ecosystem services. Some of those commonly cited include the Millennium Ecosystem Assessment framework (Alcamo et al., 2003), The Economics of Ecosystems and Biodiversity framework (De Groot et al., 2010), the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services framework (IPBES, 2017), and the Common International Classification of Ecosystem Services (Haines-Young & Potschin, 2018). The number of categories recognized varies widely; for instance, the MEA and TEEB frameworks name 21 distinct groups, while CICES includes 90.

Our report focuses on the thirteen ecosystem services defined in Table 1. We combine definitions from several frameworks, as well as those reported in the literature used for the analysis. This does not represent a comprehensive list of all the services rangelands provide. For instance, we have been unable to include critical services like erosion risk reduction. This does not mean that services outside this list are not provided or are not valuable—it only reflects the limitations of the data included in the analysis.

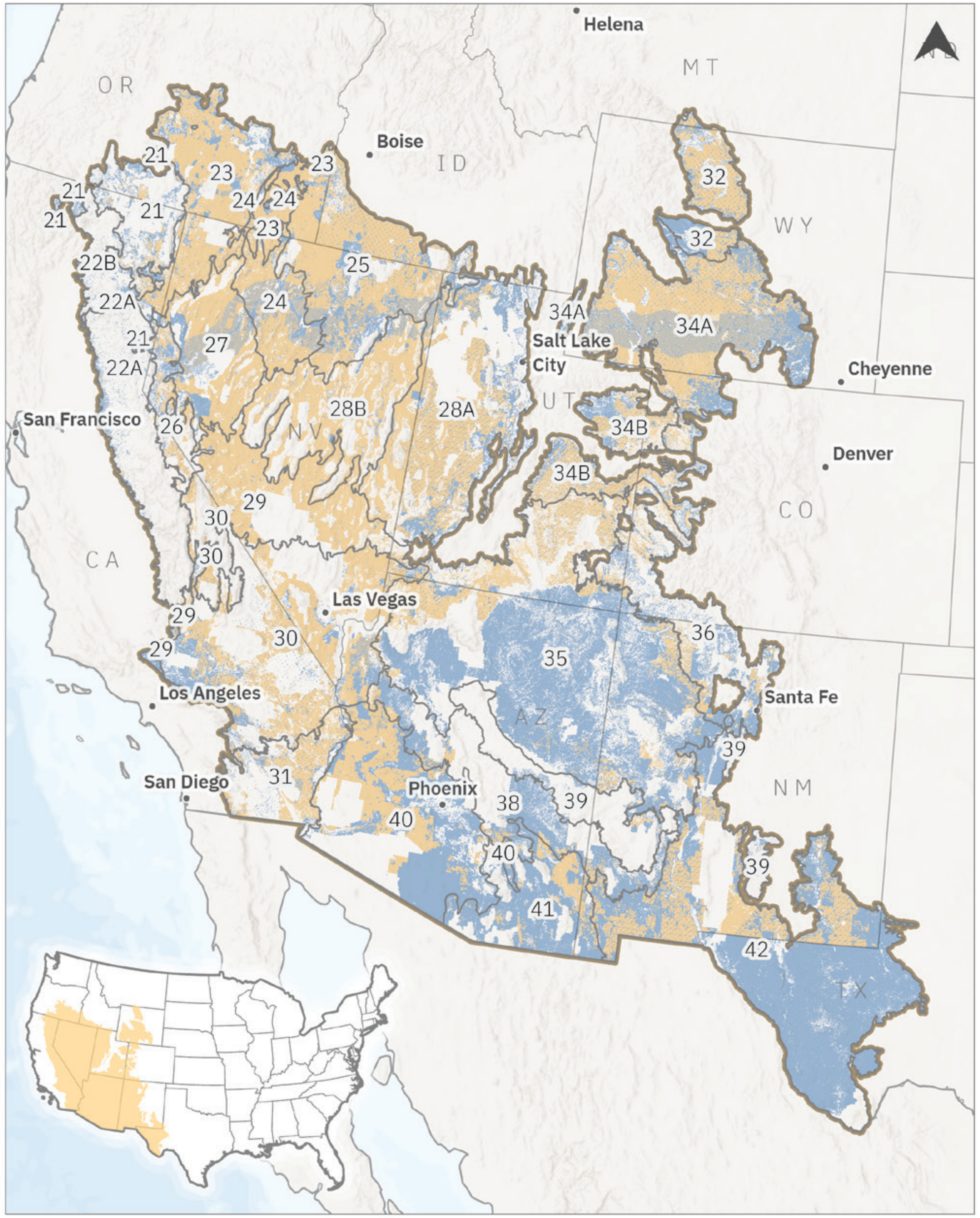
TABLE 1. ECOSYSTEM SERVICES AND THEIR POTENTIAL ECONOMIC AND/OR ENVIRONMENTAL BENEFITS TO PEOPLE

ECOSYSTEM SERVICE	DEFINITION
Aesthetics	Appreciation of natural features
Air quality	Ability to create and maintain clean, breathable air
Biological control	Regulation of pests by natural ecosystems or organisms
Carbon sequestration	Ability to remove and store carbon from the atmosphere
Fire risk reduction	Reduction in the risk of wildfire impacts on humans and infrastructure
Forage production	Production of food used for domesticated and wild animals
Habitat	Protection of biodiversity and habitats for species
Recreation	Physical enjoyment of ecosystems through outdoor activities
Social	Ecosystems' role in the desire to preserve ecosystems or satisfaction derived from knowledge that an ecosystem exists[i]
Soil fertility	Maintenance of soil structure and deposition of nutrients through nutrient cycling
Soil retention	Retaining arable land through erosion prevention
Waste treatment	Filtration of harmful pollutants and particles in water and soil
Water supply	Regulation of water flows by ecosystems used for drinking, irrigation, etc.

Source: Compiled from Daly and Farley 2004, de Groot 2002, and Boehnke-Henrichs et al. 2013.

[i] This category is based on 'supporting identities', from IPBES (2017); 'Existence value' and 'Non-use values' from Newcomer-Johnson et al. (2020); `Spiritual experience and sense of place` from De Groot et al. (2010); `Social relations`, `Sense of place` and `cultural heritage values` from Alcamo et al. (2003); `Cultural` section of CICES: Haines-Young and Potschin (2018).

FIGURE 1. BOUNDARY AND LOCATION OF LRR D AND ASSOCIATED MLRAS WITHIN THE U.S.



SOURCES: USDA, BLM, USFS, Esri, USGS, Natural Earth

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150 MILES

2.2 STUDY AREA: THE WESTERN RANGE AND IRRIGATED REGION (LRR D)

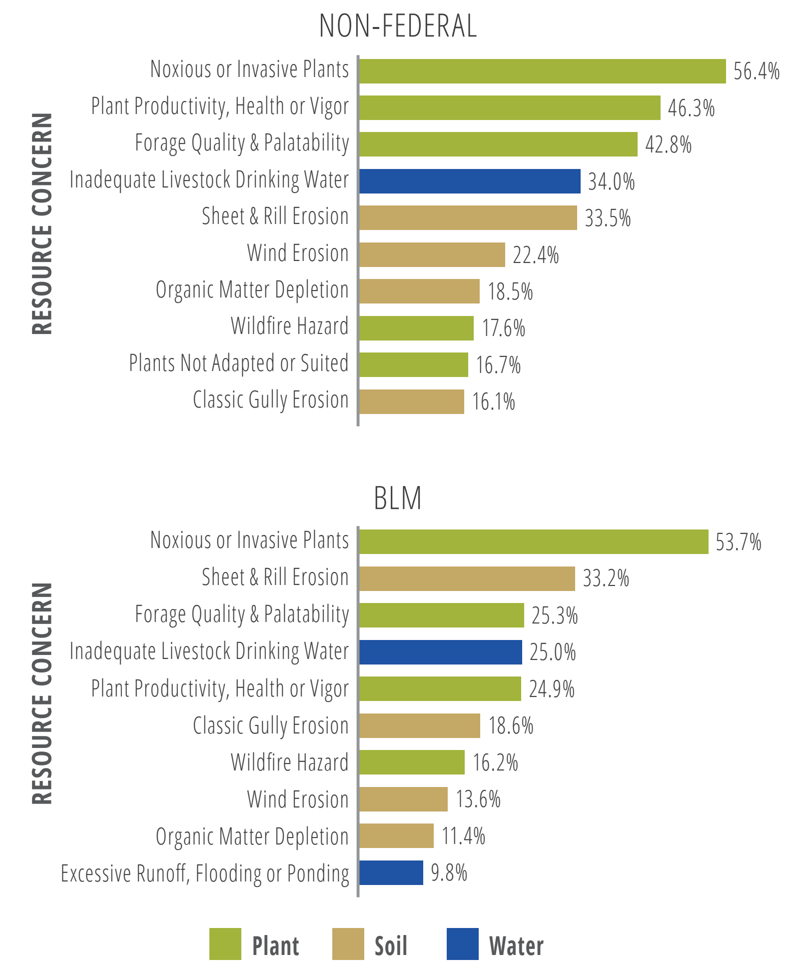
This report is focused on what NRCS identifies as the 'Western Range and Irrigated Region', or, more specifically, Land Resource Region D (LRR D).[ii] LRR D is located in the semi-desert region of the southwest (NRCS, 2006), spanning 549,725 square miles (more than 351 million acres) across Arizona, Nevada, California, New Mexico, Utah, Wyoming, Texas, Oregon, Colorado, Idaho, and Montana, covers 23 Major Land Resource Areas (MLRA)[iii], and includes all or part of 118 counties (Figure 1). Land ownership in this region is roughly 58 percent private or tribally-owned; 30 percent is federally-owned, and 11 percent is owned by state or local government. The Bureau of Land Management (BLM) manages most federal rangelands in the region. LRR D is characterized by semi-desert or desert terrain consisting of plateaus, plains, basins, and isolated mountain ranges, with average annual temperatures ranging from 40 to 60 degrees Fahrenheit and averaging 6 to 42 inches of annual rainfall. Shrubland and grassland are the most common ecosystem vegetation types.

Through the National Resources Inventory (NRI), NRCS collects and produces scientifically credible information on the status, condition, and trends of land, soil, water, and related resources on the nation's non-federal lands in support of efforts to protect, restore, and enhance the lands and waters of the United States. Resource concerns are defined as "an expected degradation of the soil, water, air, plant, or animal resource base to an extent the sustainability or intended use of the resource is impaired" (NRCS, n.d.).

The 2004-2018 NRI Grazing Land Onsite data study identified the most significant resource concerns on non-federal rangelands in LRR D. The BLM conducts their National Terrestrial Assessment, Inventory and Monitoring Survey (NTAIMS)[iv] data collection effort on rangelands using the NRI statistical framework as its basis. NRI data points that are located on federal lands managed by the BLM are shared with the BLM, thus becoming NTAIMS points. They have been inventoried annually since 2011 to establish baseline and trend data following the same, but not all, protocols within the NRCS NRI non-federal grazing land onsite data study. Figure 2 shows the percent of non-federal and BLM rangeland acres affected by the most prevalent rangeland resource concerns in LRR D, colored by type of resource concern.

FIGURE 2. PERCENT OF RANGELAND ACRES AFFECTED BY THE TOP TEN PREDOMINANT RESOURCE CONCERNS WITHIN LRR D

Source: NRCS 2004-2018 NRI Grazing Land Onsite Data Study; USDI BLM 2011-2019 NTAIMS Data Study.



[ii] The NRCS land classification system divides the United States into ecological regions, with Land Resource Regions (LRRs) as the largest units, and ecological sites or soil map units the smallest. Land Resource Regions are "geographically associated Major Land Resource Areas (MLRA) which approximate broad agricultural market regions" (NRCS, 2006).

[iii] A MLRA is described as having similar topography, geology, climate, water, soil, biological resources, and land use within its boundary (NRCS, 2006).

[iv] Formerly known as the Landscape Monitoring Framework. See Karl et. al., (2016) and (Yu et al., 2020) for more details.

The NRCS provides conservation technical and financial assistance to agricultural producers on approximately 409 million acres of non-federal rangeland in the United States, as well as federally-owned and managed rangelands where there is a direct benefit to associated private lands. The Bureau of Land Management (BLM) is responsible for the management of about 245 million acres of U.S. public lands for a variety of uses, including livestock grazing, energy development and reclamation, wildlife habitat, timber harvesting, and outdoor recreation, while also conserving natural, cultural, and historical resources.

Each agency makes considerable investments in conservation to address these resource concerns. An internal NRCS study performed by CEAP-Grazing Lands in 2016 approximated that NRCS invested an average of $71 million each year for conservation assistance on federal lands from 2005 through 2015 (unpublished NRCS data).[v] BLM provides funds for conservation treatments on their lands, by authority of the Taylor Grazing Act (1936), Federal Land Policy and Management Act (FLPMA; 1976), the Public Rangelands Improvement Act (PRIA; 1978), and subsequent legislation. Yet, many resource concerns span jurisdictional boundaries—particularly noxious or invasive species, the resource concern with the largest affected acres for both agencies in this area. Because of this and other reasons, the cooperative coordinated management of private, federal, state, and tribal lands for livestock production and ecosystem services in LRR D is a necessity.

[v] This study included all 50 states, all federal land agencies (e.g. Bureau of Land Management, US Forest Service, Bureau of Reclamation, etc.), and other land uses in addition to rangeland.

3. ANALYTICAL FRAMEWORK

We developed a framework to estimate the changes in ecosystem service value associated with grazed rangeland and rangeland management. Due to the interconnected nature of managed grazing lands in the West, this study focuses on both non-federal and BLM-managed federal rangelands in LRR D.

This analysis explores the relationship between NRCS conservation practices and BLM conservation treatments (referred to collectively as "conservation practices" here) applied on rangelands in LRR D and measures of ecosystem health and value at the MLRA level. Although NRCS conservation practice application data are collected at the field level, we protected producer confidentiality of NRCS data by removing identifying information before reporting the data by county and FSA Farm Tract. For BLM conservation investments, we accessed the BLM Land Treatment Digital Library (LTDL) and extracted practices within LRR D that are similar to the specific NRCS conservation practices selected. To protect producer confidentiality across all conservation practice data, we report only MLRA- and LRR-level results. The working definition of rangeland and its extent within the study area are detailed in section 3.2.1.

The following sections describe the framework developed to integrate NRCS and BLM rangeland conservation practice data with ecosystem service data within the study area, as well as the methods we used to calculate non-market ecosystem service benefits (i.e. nature's value) of baseline conditions (where we assumed no conservation practices have been applied) to the treatment condition (following implementation of conservation practices).

FIGURE 3. GENERAL STEPS IN THE ANALYTICAL FRAMEWORK

3.1 OVERVIEW

Broadly speaking, there are four steps to the methodology described in this chapter:

1. determine baseline ecosystem health attributes and baseline value estimates of the ecosystem services provided on rangelands within the study area;
2. determine the landcover types and acres affected by each contract or project;
3. estimate the magnitude of change in ecosystem function associated with implementation of specific rangeland conservation practices; and
4. quantify any change in nonmarket ecosystem service benefits attributable to the implementation of conservation practices (for which supporting research exists).

This following section provides a brief description of the analytical framework. Subsequent sections elaborate on each step, illustrating how each can be applied to quantify the impacts of NRCS and BLM conservation practices on the value of the ecosystem services. The framework adapts previous work conducted by the U.S. Forest Service and others on the scaling of ecosystem service valuation estimates (Aplet et al., 2000; Esposito et al., 2011; Phillips & McGee, 2014), incorporating site data collected by NRCS and BLM instead of generalized indices of ecosystem health.

The framework begins by identifying landcover[vi] characteristics within the study area, and then deriving a baseline measure of the "health" of rangeland ecosystems. Here, rangeland health is represented by three attributes, as documented in

[vi] We have chosen to unify the noun phrase "land cover" into a single word in this report to simplify subsequent references to landcover-attribute combinations. When referencing external sources, we defer to their original phrasing (e.g. the National Land Cover Database).

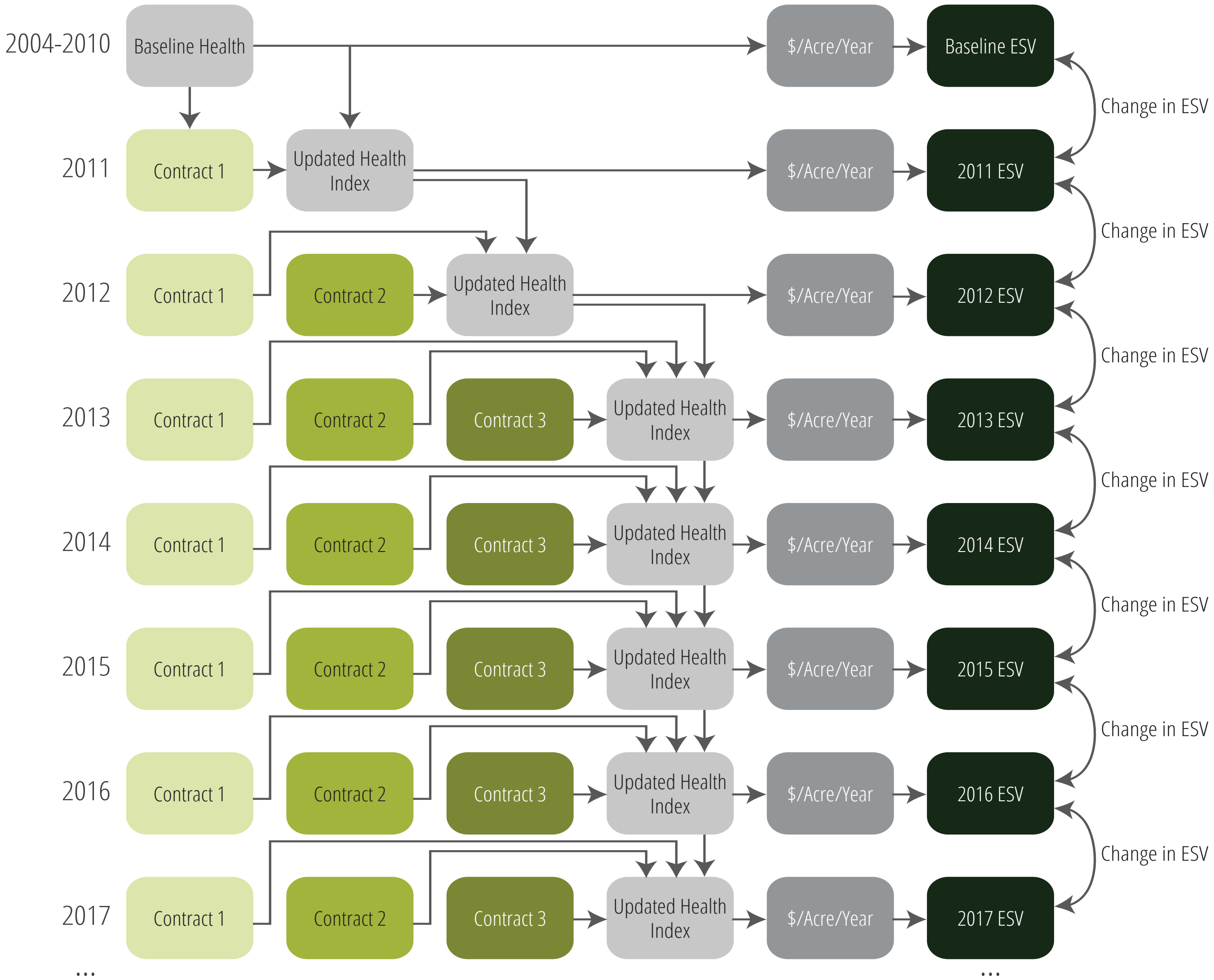
Interpreting Indicators of Rangeland Health (Pellant et al., 2020):

- **Soil and site stability (SSS)** describes the capacity of an area to limit redistribution and loss of soil resources (including nutrients and organic matter) by wind and water.
- **Hydrologic function (HF)** characterizes the capacity of an area to capture, store, and safely release water from rainfall, run-on (runoff from other locations) and snowmelt (where relevant), to resist reductions in this capacity, and to recover this capacity after reductions occur.
- **Biotic integrity (BI)** is defined as the capacity of the biotic community to support ecological processes within the normal range of variability expected for the site, to resist a loss of capacity to support such processes, and to recover this capacity after losses have occurred. The biotic community includes plants, animals, and microorganisms occurring both above and below the ground.

The baseline status of these three attributes of rangeland health is established from several indicators of biological and physical function, then normalized to generate a unified index of rangeland health, calculated as the average of the measures of the individual factors described above (section 3.2.2).

Benefit Transfer Methods (BTM) are then used to estimate the baseline economic value of ecosystem services (section 3.2.3—3.2.4) produced on rangelands within the study area, given the expert-derived level of

FIGURE 4. SIMPLIFIED FLOW CHART OF THE ANALYTICAL FRAMEWORK



rangeland health for each of those rangeland areas (section 3.2.2). BTM is broadly defined as the use of data or information in settings other than those where it was originally collected (Johnston et al. (eds.), 2015). As a means of indirectly estimating the value of ecological goods or services (Rosenberger & Loomis, 2003), BTM is widely used in the field of ecosystem service valuation and is particularly relevant in contexts where data is scarce and limited time and resources preclude new, site-specific primary valuation research for each study area (Jadhav et al., 2017).

The second step of the analytical framework involves evaluating the impacts of applied conservation practices on the derived indices of rangeland health. NRCS and BLM data on the applied practices are used to identify the areas of impact of conservation practices. A literature review then informs estimates of the impacts of specific practices on rangeland health attributes. Due to the limited amount of relevant peer-reviewed literature, this analysis is restricted to impacts of three rangeland management practices, all of which are applied extensively in LRR D: Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing.

Studies are then selected to estimate the proportional change in rangeland health attributes (percent change per year) associated with each practice. Since practice effectiveness can vary over time after practice implementation, we estimate change rates annually for up to 5 years post-implementation. These rates are then used to estimate changes to both baselines: health indices and the monetary values assigned to those indices.

The baseline rangeland ecosystem health estimates for the analytical framework have been calculated based on 2004-2010 data for the NRCS practices, and 2011-2015 for the BLM data. Annual impacts of conservation practices on rangeland health and the ecosystem service values associated with those health improvements are then sequentially calculated, as conservation practices were applied each year from 2011 through 2020 (NRCS) and from 2016 to 2020 (BLM). Regional totals for the value of changes in ecosystem services are calculated as the sum changes in value across each rangeland vegetation type within the study area, over the full period of analysis.

The generalized analytical framework outlined in Figure 3 has been applied for each year of NRCS-certified contracts (2011-2020), and for BLM practices (2016-2020) with the benefits associated with each subsequent year based on conservation practices implemented in prior years. These steps are detailed in the following sections. Figure 4 shows an example of how the ecosystem service values (ESV) of three hypothetical NRCS contracts (implementing a combination of practices considered in this report) are calculated within the framework.

As with any attempt to estimate ecosystem service values and land health trajectories, the effectiveness of this approach depends on sufficient site data and related literature, including—but not limited to—primary studies and environmental factors. Section 4 provides a discussion of limitations encountered in this analysis.

Tracing linkages between ecosystem health, conservation practices, and economic values necessarily requires assumptions about functional relationships between variables that are not well-understood. Highlighting such intricacies and the need for greater understanding of critical relationships underscores the research necessary to improve our ability to generate such estimates.

[ii] We have chosen to unify the noun phrase "land cover" into a single word in this report to simplify subsequent references to landcover-attribute combinations. When referencing external sources, we defer to their original phrasing (e.g., the National Land Cover Database).

In the discussion that follows, we detail the assumptions that we applied to this analysis and summarize critical assumptions and uncertainties in the following chapter.

3.2 BASELINE ANALYSIS

The scope of this report is limited to rangelands managed on Bureau of Land Management (BLM), Bureau of Indian Affairs (BIA), State trust, and privately owned land in LRR D, and a subset of NRCS and BLM conservation practices applied to those rangelands from 2011–2020 (NRCS), and 2016–2020 (BLM). The different date ranges of the NRCS and BLM assessments are because baseline rangeland health data collection began later on BLM lands. To assess changes in the quantity and value of ecosystem services produced by these lands over time, we first determine the baseline condition of rangeland in the study area prior to implementation of NRCS and BLM conservation practices during the time period of analysis. The following sections describe the estimation of baseline levels of rangeland health and the economic value of ecosystem services. These baselines are the reference conditions to estimate changes in ecosystem function (section 3.3) and ecosystem service provisioning (section 3.4) associated with implementation of BLM and NRCS conservation practices on rangelands in LRR D between 2016-2020 (BLM) and 2011 to 2020 (NRCS).

NRCS defines rangeland as:

"A broad land cover/use category on which the climax or potential plant cover is composed principally of native grasses, grass-like plants, forbs or shrubs suitable for grazing and browsing, and introduced forage species that are managed like rangeland. This would include areas where introduced hardy and persistent grasses, such as crested wheatgrass, are planted and such practices as deferred grazing, burning, chaining, and rotational grazing are used, with little or no chemicals or fertilizer being applied. Grasslands, savannas, wetlands, deserts, and tundra are considered to be rangeland. Certain communities of low forbs and shrubs, such as mesquite, chaparral, mountain shrub, and pinyon-juniper, are also included as rangeland" (U.S. Department of Agriculture, 2020, pp 3-2).

3.2.1 CHARACTERIZING TYPES OF RANGELAND IN THE STUDY AREA

Using the NRCS definition, rangeland in the study area consists of multiple landcover types. The "Rangelands in the coterminous U.S." dataset created by the U.S. Forest Service (USFS) (Reeves & Mitchell, 2011) maps the extent of rangelands but does not indicate landcover types. To determine the landcover within LRR D rangelands, we intersected the 30-meter National Land Cover Database 2016 (NLCD) (Jin et al., 2019) (see Table 2 for the land cover types defined in NLCD) with the three rangeland categories in the USFS rangeland data (Rangeland, Transitional Rangeland, and Afforested Rangeland) using Esri's ArcGIS software. Following the NRCS definition of rangelands, we counted any forest, shrubland, or grassland acre intersecting the three USFS rangeland types to count as a "rangeland acre" in the study area.

Rangeland in the study area covers nearly 193 million acres, including 103 million acres that are BLM-managed, and 90 million acres that are non-federal rangelands. More than 96 percent of this area is grassland or shrubland. Figure 5 shows a map of where these landcovers are found in the study area; Table 3 shows the proportion of each landcover type relative to the total area.

TABLE 2. LAND COVER DEFINITIONS OF RANGELAND LAND COVER TYPES, AS USED IN THE NATIONAL LAND COVER DATABASE (2016)

LANDCOVER	DESCRIPTION
Forest	Areas dominated by trees generally greater than 5 meters tall and greater than 20% of total vegetation cover.
Shrubland	Areas dominated by shrubs; less than 5 meters tall with shrub canopy typically greater than 20% of total vegetation. This class includes true shrubs, young trees in an early successional stage or trees stunted from environmental conditions.
Grassland	Areas dominated by graminoid or herbaceous vegetation, generally greater than 80% of total vegetation. These areas are not subject to intensive management such as tilling, but can be utilized for grazing.
Wetlands	Vegetated areas where the soil or substrate is periodically saturated with or covered with water.

Source: Jin et al., (2019)

TABLE 3. RANGELAND LANDCOVER EXTENTS IN THE STUDY AREA

LANDCOVER	STUDY AREA	PERCENT OF NON-FEDERAL LAND	PERCENT OF BLM LAND
Forest	4,522,458 (2%)	3,175,255 (4%)	1,347,203 (1%)
Grassland	27,884,697 (15%)	13,736,357 (15%)	14,148,340 (14%)
Shrubland	160,525,766 (83%)	72,660,025 (81%)	87,865,741 (85%)
Total	192,932,921 (100%)	89,571,636 (100%)	103,361,285 (100%)

FIGURE 5. RANGELAND LANDCOVER TYPES IN THE STUDY AREA

Sources: USDA, BLM, USFS, Esri, USGS, Natural Earth
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3.2.1.1 CHARACTERIZING RANGELAND PROFILES OF TREATMENTS

Since our goal is to estimate the economic value of ecosystem services provided by rangelands by landcover type, we must know the acreage of landcover types affected by each treatment applied. However, NRCS and BLM conservation practice data do not distinguish by the type of landcover treated, only total area. We combined available GIS information with the landcover layers determined in section 3.2.1 to make assumptions about the landcover types affected by treatments.

Many BLM treatments are associated with geospatial boundaries available in the Land Treatment Digital Library (Pilliod et al., 2019). We intersected these boundaries with the landcover layers described in section 3.2.1 to find the total acres of each landcover affected by any given project and treatment.

NRCS contract data did not delineate treatment area boundaries. Some NRCS contract data were associated with the Farm Service Agency (FSA) Tract and represent the smallest unit of analysis we could obtain for this data. Where FSA tracts are not identified, the next smallest unit of analysis is the county level. To approximate the landcover types treated by NRCS contracts, we calculated the proportions of landcover acres present (from the data described in section 3.2.1) in either the FSA tract (where available) or the relevant county. Table 4 presents a hypothetical example of a rangeland profile on an FSA tract. In this example, one acre of rangeland is assumed to include 0.85 acres of shrubland, 0.13 acres of grassland, and 0.02 acre of forest. We scaled the distribution of landcover for NRCS treatment contracts for larger areas accordingly.

3.2.2 CALCULATING A MEASURE OF BASELINE RANGELAND HEALTH

Many primary valuation studies omit detailed site conditions or do not present information on environmental quality in a comparable manner (Newbold et al., 2018). Thus, studies encompass a wide range of environmental quality, including healthy landscapes (i.e. those at fully functional, "reference" conditions (or nearly so) and hypothetical reference conditions for degraded sites. Following previous work, we assume that monetary estimates in the research literature (see section 3.2.3) represent values at "reference" conditions and discount these monetary estimates to reflect baseline conditions using a proxy index of ecosystem health (Aplet et al., 2000; Esposito et al., 2011; Phillips & McGee, 2014). Despite the literature potentially including multiple levels of environmental quality, this is a conservative assumption as even values not at "reference" conditions are discounted. In this section, we describe our approach for calculating an index factor of the relative health of the rangeland areas in the study area for 2004-2010 (NRCS) and 2014-2015 (BLM).

Direct measures of the three attributes of rangeland health—**soil and site stability (SSS), hydrologic function (HF),** and **biotic integrity (BI)**—are difficult to determine due to the complexity of the underlying processes. Related biological and physical characteristics were used as indicators of overall ecosystem function. We based the baseline status of the three attributes of rangeland health on seventeen indicators from NRCS' National Resources Inventory (NRI) *Grazing Land Onsite Data Study, Rangeland Health Assessment Protocol* (NRCS, 2014; NRCS, 2004), which is also followed by BLM in their National Terrestrial Assessment, Inventory and Monitoring Survey (NTAIMS) (Pellant et al., 2020). Both NRI and NTAIMS are statistical surveys of natural resource conditions and trends on non-federal (NRI) and federal (BLM) land within the United States.[vii] We based rangeland health on the NRI and NTAIMS assessments of the degree of departure from ecological reference conditions, as determined from the Ecological Site Descriptions developed by the NRCS and/or BLM for all seventeen indicators (Table 5).[viii]

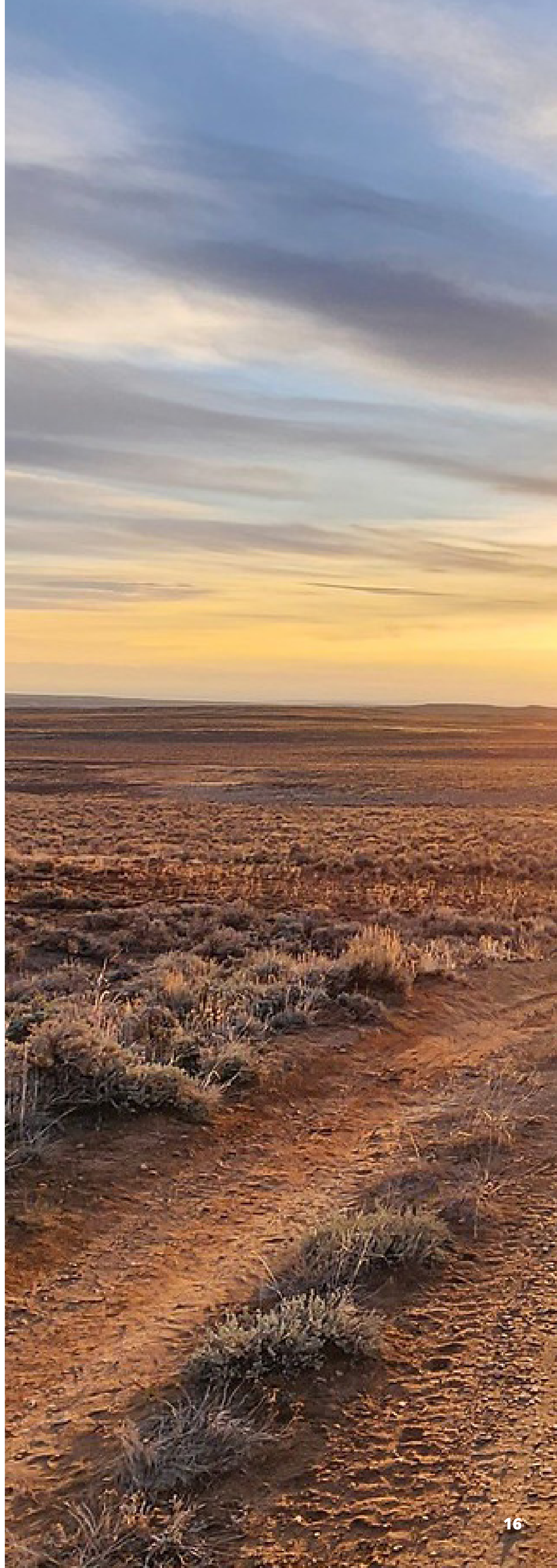
The NRI/NTAIMS data characterize indicators by *departure categories*, which describe the degree of departure from the reference conditions of each ecological site.[viii] The degree to which indicators depart from expected reference conditions are characterized as: 1 (none-to-slight); 2 (slight-to-moderate); 3 (moderate); 4 (moderate-to-extreme); or 5 (extreme-to-total). As departure increases, the ecological site function is inhibited. Depending on the affected indicators and attributes, a site may lose its capacity to: retain soil; store and release water; provide nutrients for plant growth; or cycle energy efficiently.

TABLE 4. EXAMPLE LANDCOVER CONVERSION FACTORS FROM LAND USE TO LANDCOVER FOR A HYPOTHETICAL FSA TRACT

LAND USE	FSA LANDCOVER	CONVERSION FACTOR
Rangeland	Grassland	13%
	Shrubland	85%
	Forest	2%

TABLE 5. RANGELAND HEALTH INDICATORS AND ATTRIBUTES IN THE NRI AND NTAIMS

LANDCOVER	DESCRIPTION
Rills	SSS; HF
Water flow patterns	SSS; HF
Pedestals and/or terracettes	SSS; HF
Bare ground	SSS; HF
Gullies	SSS; HF
Wind-scoured, blowout, and/or depositional areas	SSS
Litter movement	SSS
Soil surface resistance	SSS; HF; BI
Soil surface loss or degradation	SSS; HF; BI
Compaction layer	HF
Plant community composition and distribution relative to infiltration and runoff	SSS; HF; BI
Litter amount	BI
Functional/structural groups	BI
Plant mortality/decadence	HF; BI
Annual production	BI
Invasive plants	BI
Reproductive capability of perennial plants	BI



[vii] Non-federal land includes privately-owned lands, tribal and trust lands, and lands controlled by state and local governments. Federal land, in this report, only represents rangelands administered by the Bureau of Land Management (BLM).

[viii] Reference conditions are determined by Ecological Site Descriptions developed by the NRCS and/or BLM. The rangeland health assessment provides information about how ecological processes are functioning relative to a site's ecological potential. Because ecological potential varies both locally and regionally, NRI and NTAIMS rangeland health assessments are based on the reference plant community and conditions for the ecological site. It is important to note that each ecological site will vary in its response to management actions, inputs or stressors placed upon it.

We then combined baseline NRI/NTAIMS data for the indicators[ix] into rangeland health indices as follows:

1. Because the indicator data were collected at multiple sites over a seven-year (NRI) and four-year (NTAIMS) period, we identified the median departure values for indicators associated with soil and site stability (SSS), hydrologic function (HF), and biotic integrity (BI) as reflective of the general health of each landcover type in each MLRA.

2. To normalize the departure scores for each attribute at each point, the median attribute value of the MLRA was subtracted from 6, and then divided by 5, producing a 0.2–1 index, with 0.2 representing the greatest departure from reference conditions (Table 6). For example, if the median departure value for the indicators associated with a given health attribute is 2.5, the resulting index value is 0.70. The estimation process is mathematically represented as:

$$Score^{i}_{normalized} = \frac{6 - Score^{i}_{median}}{5}$$

Where $Score^{i}_{median}$ is the median of all values corresponding to the attributes associated with index i, and i is one of SSS, HF, or BI.

These indices reflect the relative position of each ecological site[x] along a continuum of departure from reference conditions, allowing us to make general statements such as "an index value of 0.80 is closer to the full potential (1.0) of the ecological site than a value of 0.20," or, "higher index values indicate healthier sites."

Attribute indices for each MLRA need to be similarly aggregated to produce average MLRA scores, weighted by the acres represented by each NRI or NTAIMS measurement.[xi] We then took the average index score across all rangeland health attributes to represent the overall baseline health of rangelands in each MLRA, where each attribute contributes an equal weight to the overall health of a rangeland ecological site. While some attributes may be more influential than others for a given ecological site, we lacked the data and supporting literature necessary to develop a more sophisticated metric.

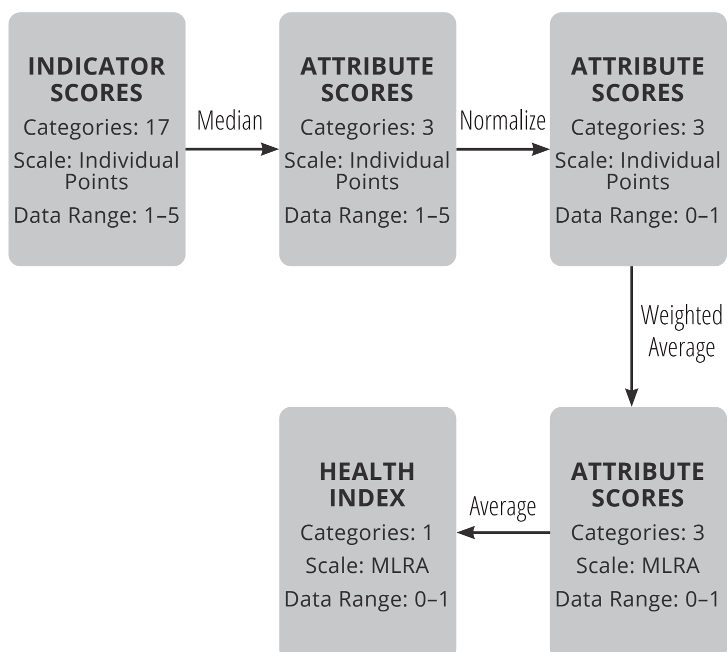
Accordingly, we urge caution when interpreting the health index scores. While multiple factors influence rangeland health attribute ratings, none describe historical use and management, current management, significant weather events and their impact on site condition at the time of assessment, or other relevant considerations. The rangeland health index scores developed here are not intended to be used as point estimates of rangeland health for specific sites, as they have been aggregated to the MLRA scale as a proxy for average rangeland health across broader areas. Moreover, specific index scores are of less interest than the *relative change* in indices attributable to application of conservation practices.

TABLE 6. RANGELAND HEALTH DEPARTURE CATEGORIES AND BASELINE HEALTH INDEX USED IN THIS STUDY

DEPARTURE CATEGORY	NRI SCORE	HEALTH INDEX SCORE
None to Slight	1	0.81 to 1.00
Slight to Moderate	2	0.61 to 0.80
Moderate	3	0.41 to 0.60
Moderate to Extreme	4	0.21 to 0.40
Extreme to Total	5	0.20*

**The lowest value on the rangeland index is 0.20, because despite the severity of degradation, most rangelands still have a capacity to perform basic functional processes related to soil stability, water capture/storage, biotic integrity, and nutrient and energy cycling. If the score reached zero, then it would likely be due to a land use change, in which we would place the land into a different land use category, not rangeland.*

FIGURE 6. FLOWCHART DEPICTING PROCEDURES FOR DETERMINING BASELINE HEALTH INDEX SCORES



3.2.2.1. BASELINE HEALTH INDEX RESULTS FOR THE STUDY AREA

Figure 7 and Figure 8 show the distribution of MLRA-aggregated rangeland health index scores throughout the LRR for both non-federal and BLM-managed rangelands. For non-federal rangeland, MLRA-aggregated rangeland health indices all fell at or above 0.7, with Biotic Integrity exhibiting the greatest variation across MLRAs. Overall, the MLRA-level scores for soil and site stability and hydrologic function were high, with the median score being about 0.9 across all MLRAs for both attributes. For BLM-managed rangeland, scores were at or above 0.5. Differences in the baseline scores of MLRAs are due to different datasets being used as NRI is only applicable on non-federal rangeland and AIM is conducted only on BLM rangelands. Also note that each dataset had a different timeline to construct these baseline estimates (2004-2010 for NRI and 2014-2015 for AIM).

FIGURE 7. BOXPLOT OF MLRA-LEVEL RANGELAND HEALTH ATTRIBUTE INDEX SCORES FROM BASELINE NRI (N=22) AND NTAIMS (N=21) DATA

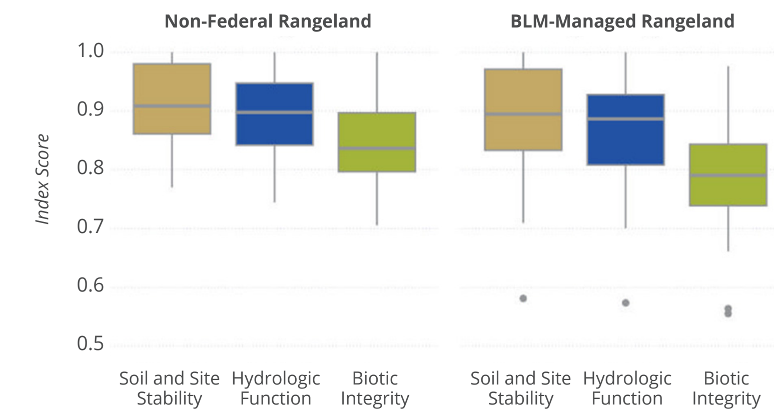
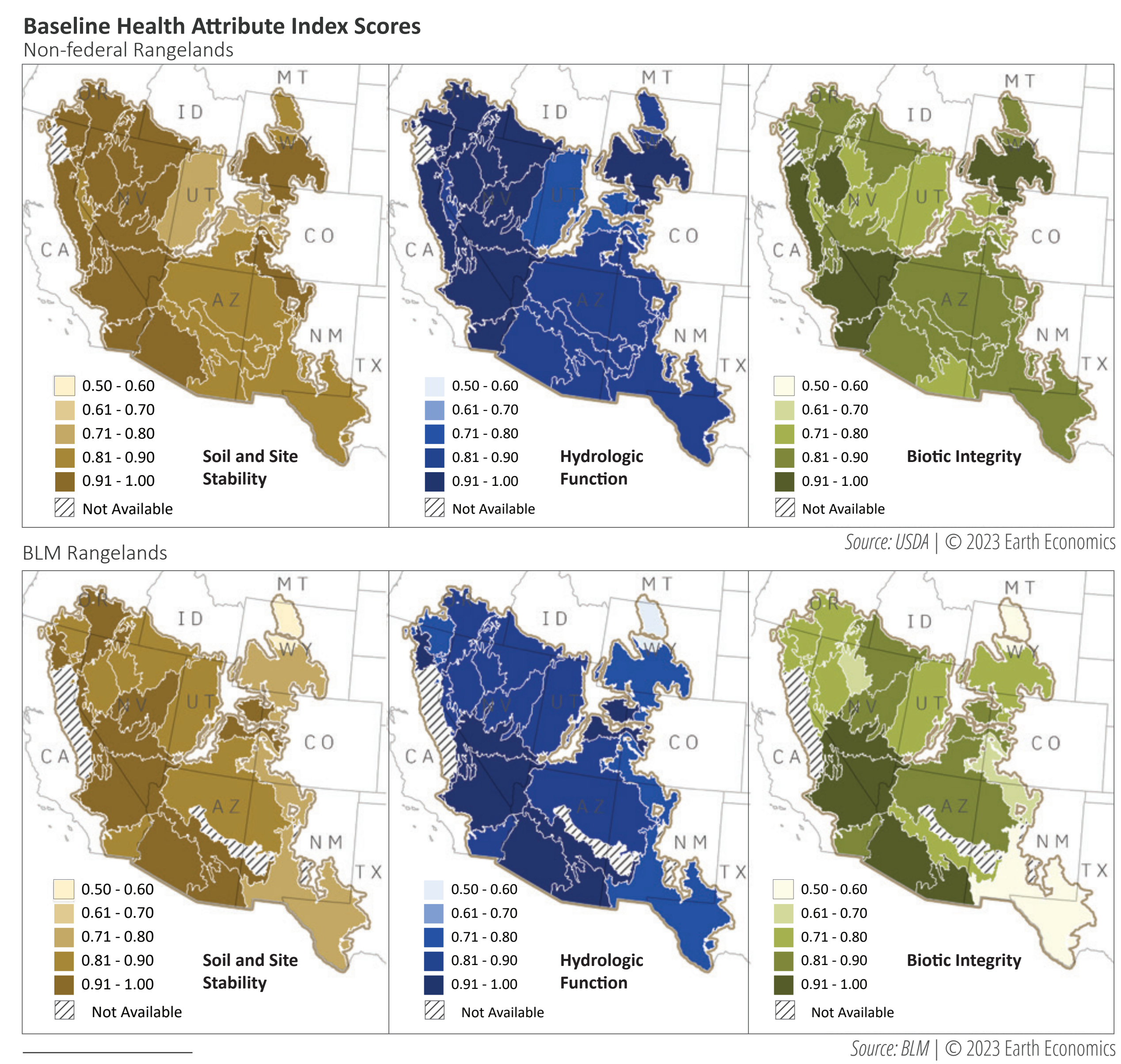


FIGURE 8. VARIATION IN MLRA BASELINE HEALTH ATTRIBUTE INDEX SCORES FOR RANGELAND IN THE STUDY AREA[xii]



[ix] In 2014, the NRI protocol for determining attribute scores changed. The NRI baseline attribute scores are constructed from 2004-2010 data, while the AIM baseline uses the updated methodology.

[x] There is a range of 20 to 302 (average of 77) ecological sites per MLRA throughout LRR D.

[xi] The NRI/NTAIMS statistical framework provides an estimate of the number of acres each point represents.

[xii] MLRA's that did not have NRI or AIM data are labeled as "Not Available."

3.2.3 NON-MARKET BENEFIT VALUATION METHODOLOGY

We used Benefit Transfer Methods (BTM) to associate published research on the economic value of ecosystem services (by ecosystem type) with comparable ecosystems (known as "transfer sites") within the study region (Johnston et al. (eds.), 2015). As a secondary research method, BTM results can be somewhat imprecise, but applying conservative (and transparent) criteria for selecting eligible primary studies generates reasonable estimates that may be sufficient to inform decision making. As with all research, such estimates may be improved as more research and data become available.

3.2.3.1 IDENTIFYING STUDIES FOR USE IN BTM

We begin our BTM process by screening published studies for similar landcover classifications (e.g. wetland, forest, grassland) as those found within the study area (i.e. transfer sites), as defined in section 1.2.1. In our initial search, we included peer-reviewed valuation studies, published reports, and gray literature on the value of ecosystem services in an initial search of valuation literature conducted within the United States. Following best practices, these studies were subjected to a double-review process in which two analysts independently reviewed and coded studies for inclusion into the valuation dataset (Boyle & Parmeter, 2017) to ensure that values have been selected based on commensurate site attributes (Plummer, 2009) and best-available methodologies.

The criteria for evaluating data within studies for inclusion in the analysis have been summarized below. Studies that failed to meet these criteria were excluded. Appendix B lists the studies selected for inclusion in the analysis.

SIMILARITY OF ECOSYSTEM GOODS AND SERVICES

At the most basic level, the ecosystem services valued at both study and transfer sites should be commensurate, with similar goods, services, and uses at both considered critical for valid transfers (Plummer, 2009; Brouwer, 2000; Spash & Vatn, 2006; Boyle & Bergstrom, 1992). During review, we identified the ecosystem services produced by each published study site. If those services could not plausibly be provided by rangelands within the study area, those values were excluded from further consideration.

SIMILARITY OF LANDCOVER TYPES

Landcover types at both the study and transfer sites must also be similar, with errors in estimates of ecosystem service values diminishing as similarities between study and transfer sites increase (Rosenberger & Loomis 2003; Plummer, 2009; Rosenberger & Stanley, 2006). The ecosystems central to this study are described in the 2016 NLCD framework outlined in section 3.2.1. In addition, proximity to urban centers can affect ecosystem services values, so our dataset focuses on general or rural study sites. If a landcover reported in the primary literature did not fit into this framework, that study was excluded.

CREDIBLE AND APPROPRIATE METHODOLOGY

To be selected, published studies must be from a credible source (for example, those having a rigorous review process for publication, minimal bias, or providing clear citations) and apply accepted economic valuation methods to high-quality data (Plummer, 2009; Boyle & Bergstrom, 1992; Brookshire, 1992; Freeman, 1984). Primary valuation methods, refined within the environmental and natural resource economics communities over decades, fall into three broad categories: 1) direct market valuation, 2) revealed preferences, and 3) stated preferences (Pascual et al., 2010). Table 7 provides descriptions of the most common valuation techniques.

TABLE 7. COMMON PRIMARY VALUATION METHODS

METHOD	DESCRIPTION	EXAMPLE
DIRECT MARKET VALUATION		
Market Price	Valuations are directly obtained from the prices paid for the good or service in markets.	The price of wheat sold on open markets.
Replacement Cost	Cost of replacing open space services with engineered systems.	The cost of replacing a watershed's natural filtration capacity with an engineered filtration facility.
Avoided Cost	Costs avoided or mitigated by open space services that would have been incurred in the ab-sence of those services.	Grasslands absorb and retain water, reducing flooding and recovery costs.
Production Approaches	Value created from an open space service through increases to dependent economic outputs.	Better grazing land health may increase stocking rates for livestock.
REVEALED PREFERENCE APPROACHES		
Travel Cost	Costs incurred to consume or enjoy open space services reflects a minimum implicit value of the service.	Tourists who travel to visit a locale must value that resource *at least as much* as the cost of traveling there.
Hedonic Pricing	Value implied by the additional price consumers are willing to pay for the service in related markets.	Property values near lakes and parks tend to exceed similar properties without such nearby amenities.
STATED PREFERENCE APPROACHES		
Contingent Valuation	Value elicited by posing hypothetical, valuation scenarios.	What people are willing to pay to protect wilderness from development.
Conjoint Analysis	Values estimated from choosing or ranking different scenarios of ecosystem service amounts.	Choosing between restoration scenarios providing varying levels of forage yields.

Note: adapted from Farber et al. (2006)

The economic research literature provides guidance on which valuation methods are best-suited to each ecosystem service. For example, when valuing recreation, the travel cost approach is more appropriate than hedonic pricing. Table 8 lists each ecosystem service and the most appropriate valuation methodologies as identified in, or inferred from, the literature (Farber et al., 2006). There is some evidence that aligns with these transferability estimates on the transferability of the valuation methods themselves. Lewis and Landry (2017) conduct a hedonic analysis of river proximity and a subsequent validity test of benefit transfer and suggest caution when applying function transfer to hedonic models—which are common when evaluating aesthetic and recreation services, which have low transferability. Newbold et al. (2018) note that stated preference valuation formats tended to have larger transfer error (see section 4.1.1 for a discussion of transfer error), which would imply low transferability of those methods, and are commonly used to value services such as habitat and social values. We included large-scale meta-analyses where possible, which provide generalized estimates across multiple study sites, as grasslands and shrublands are some of the least-studied ecosystems in terms of ecosystem service valuation.

The double-review process also assessed study methodologies. While no hard-and-fast rules exist on criteria for rejecting or accepting studies in BTM based on the methodologies applied, we reviewed each study's methodology to note any weaknesses (e.g. small sample sizes, low response rates, weak explanatory power). The studies included in the database were all considered to have applied appropriate methods with sufficient rigor to high-quality data. See Appendix B for details on each study used.

STUDY SITE LOCATION

We limited the selection of valuation studies to those conducted in the United States. Studies conducted within the LRR D study area were evaluated first. Studies based elsewhere in the continental United States were included on a case-by-case basis but limited to those with at least medium transferability and matching other key criteria described in this section. For example, ecosystem processes with broadly distributed benefits (e.g. carbon sequestration) are highly transferable, while services with more localized effects (e.g. habitat for specific species), tend to be less so (see Table 8).

STUDY SITE DEMOGRAPHICS

Benefit transfers are thought to be more accurate when demographics, social attitudes, and consumer beliefs are similar at both transfer and study sites (Boyle & Bergstrom, 1992; Loomis & Rosenberger, 2006; VandenBerg et al., 1995), yet the significance of these variables has been mixed in valuation models (Schmidt et al., 2016; Unsworth & Petersen, 1995; Wilson & Hoehn, 2006). Unfortunately, it is unusual for such sociocultural characteristics to be reported in the literature identified for this study, apart from easily obtainable data such as income or population density. Limiting study location (see "Study Location") can help to partly address the effects of cultural attitudes and beliefs. Although LRR D is large and includes many different social and economic settings, we chose not to adjust ecosystem service values by socioeconomic or sociocultural values unless a study used in function transfer already included variables (e.g. income) to avoid potential bias. Including local income data in these function transfers allows for determination of a range of values that captures the different socioeconomic contexts in LRR D.

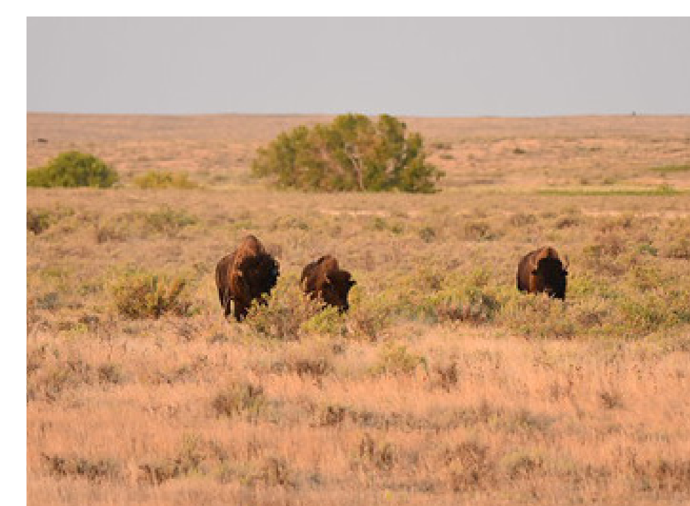
PUBLICATION YEAR

All things being equal, value transfers will be more accurate if the time between the original publication year and the present is smaller (Wilson & Hoehn, 2006; Richardson et al., 2015). Accordingly, we omitted literature published prior to 2000, and prioritized more recent studies for inclusion. The oldest selected study was published in 2002.

TABLE 8. TRANSFERABILITY AND VALUATION METHODS FOR ECOSYSTEM SERVICES

ECOSYSTEM SERVICE	MOST APPROPRIATE VALUATION METHOD	TRANSFERABILITY ACROSS SITES
Aesthetics	H, CV, TC, ranking	Low
Air quality	CV, AC, RC	High
Biological control	AC, P	High
Carbon sequestration	CV, AC, RC	High
Fire risk reduction	AC	Medium
Forage production	M, P	High
Habitat	CV	Low
Recreation	TC, CV, ranking	Low
Social	CV, ranking	Low
Soil fertility	AC, CV	Medium
Soil retention	AC, RC, H	Medium
Waste treatment	RC, AC, CV	Medium to High
Water supply	AC, RC, M, TC	Medium

Key: AC, avoided cost; CV, contingent valuation; CA, conjoint analysis; H, hedonic pricing; M, market pricing; P, production approach; RC, replacement cost; TC, travel cost. This table is adapted from Farber et al. (2006).



3.2.3.2 ASSIGNING MONETARY VALUES TO ECOSYSTEM SERVICES

Using these criteria, we selected the best-available studies for each landcover-ecosystem service combination, standardizing all estimates to units of dollars-per acre-per year ($/acre/year), adjusted to 2021 U.S. dollars using the World Bank GDP inflation and deflation factors.

It was possible to adjust the outputs of several studies using function transfer, a benefit transfer method which uses statistical models estimated for individual study sites (aka "value functions") in conjunction with information on transfer site characteristics to estimate the unit value of an ecosystem service at the transfer site. This approach offers many advantages, including the ability to tailor value estimates to the transfer area. Some research suggests that function transfers can be more accurate than point estimates (Kaul et al., 2013). Where function transfers were not available, we used the simpler point transfer approach. Appendix B describes each study included in the ecosystem service valuation dataset, its characteristics, and function transfer processes, where these were applied.

We report maximum and minimum values across all selected studies in any given combination of landcover and ecosystem services as the final estimates for each category. In other words, if a landcover-ecosystem service combination had multiple appropriate values in the dataset, we took the range of those values. Reporting value ranges underscores variability in location, methods, and socioeconomic characteristics of the selected studies. The unit values of all ecosystem services were then summed for each associated landcover type to calculate the total annual value produced by an acre of each landcover. We then adjusted these totals by the baseline rangeland health index scores, which were then scaled by their extent within each MLRA to estimate the total baseline value of ecosystem services across each MLRA (section 3.2.4). We lacked information on recreation access to private rangelands. Since all studies selected to estimate recreation value were conducted on public lands, we apply recreation value only to BLM-managed rangelands.

Accordingly, a total of 34 value estimates from 16 studies on grassland, shrubland, forest and wetland ecosystem services have been included in the dataset. Table 9 summarizes the combinations of landcover and ecosystem services that were able to be valued based on literature meeting the inclusion criteria (described in section 3.2.3.1). Highlighted combinations represent combinations valued in the benefit transfer dataset. Again, although this dataset represents the best-available approximation of ecosystem service valuation estimates in the study area, it may be extended and improved as new primary analyses and better data become available.

That specific combinations of landcover and associated attributes and ecosystem service value are not included here does not necessarily mean that such ecosystems do not produce a given service—or that the service is not valuable. Rather, absence may simply reflect a lack of peer-reviewed data relevant to that combination. For example, shrubland is known to provide valuable services (e.g. flood risk reduction), yet there are few valuation studies of this landcover type, none of which we determined were suitable for LRR D sites. Thus, caution should be exercised when comparing the total value of ecosystem services across landcover types, as differences in values may reflect information gaps, rather than real differences in ecosystem productivity or the economic value of such services. Ongoing investments in primary valuations are needed to fill gaps in our ability to estimate the full range of ecosystem service values. See section 4 for a detailed discussion on study limitations.

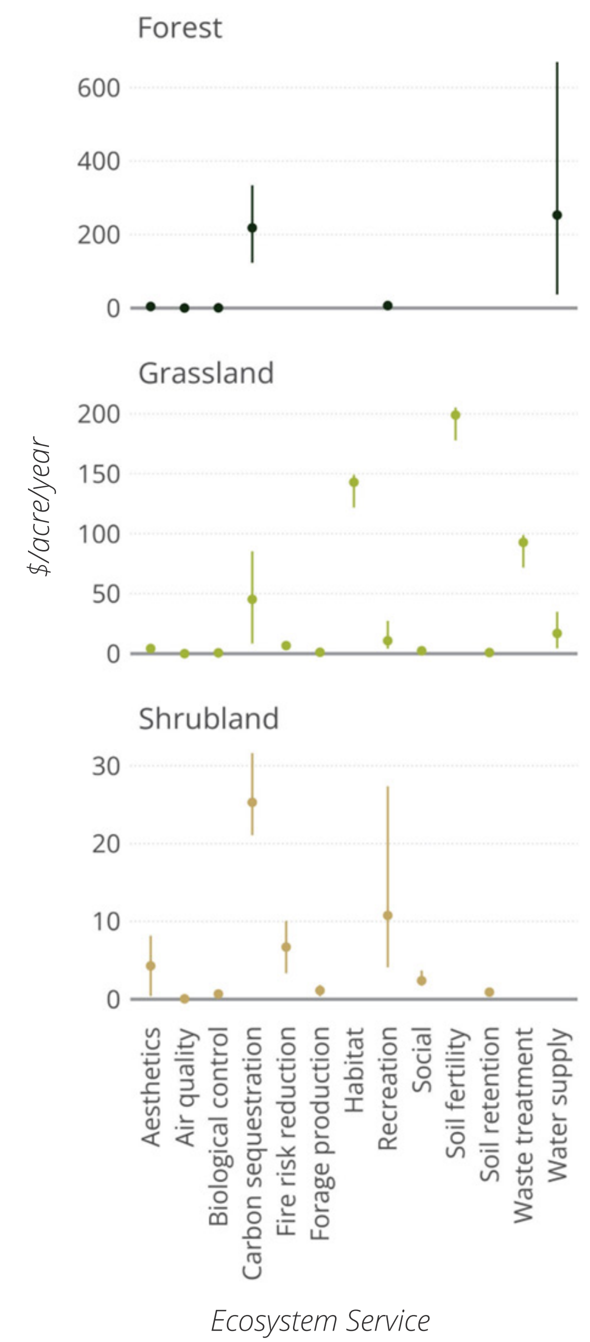
Figure 9 plots the distribution of ecosystem service value (ESV) estimates included in the dataset, across the relevant landcover types. Dots indicate the mean value among all studies while whiskers represent the range.

TABLE 9. ECOSYSTEM SERVICE AND GENERAL LANDCOVER COMBINATIONS VALUED IN THE STUDY AREA

ECOSYSTEM SERVICES VALUED IN THIS STUDY	FOREST	GRASSLAND	SHRUBLAND
Aesthetics	•	•	•
Air quality	•	•	•
Biological control	•	•	•
Carbon sequestration	•	•	•
Fire risk reduction		•	•
Forage production		•	•
Habitat		•	
Recreation	•	•	•
Social		•	•
Soil fertility		•	
Soil retention		•	•
Waste treatment		•	
Water supply	•	•	

• Indicates landcover-ecosystem service combination included in the ESV dataset.

FIGURE 9. RANGE OF ECOSYSTEM SERVICE VALUES (ESV) USED IN THIS STUDY



Lines indicate the range (minimum and maximum) of values in the dataset for each category while points indicate the mean value. Variation is displayed across all values used in this study for a given landcover type and ecosystem service combination.

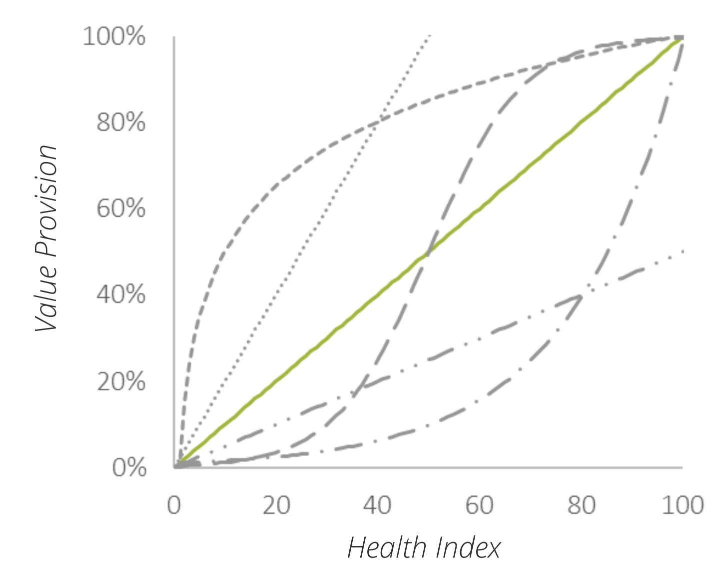
3.2.4 CALCULATING BASELINE ECOSYSTEM SERVICE BENEFITS

As explained in section 3.2.2, we assumed that the valuation literature represents values of ecosystems at reference conditions. Once these ecosystem service unit-value estimates (at reference conditions) were identified, they were then discounted by baseline MLRA rangeland health attribute indices to approximate the baseline ecosystem function and associated ecosystem service value for each MLRA (Aplet et al., 2000; Esposito et al., 2011; Phillips & McGee, 2014).

Following our earlier work in LRR H, we assumed that relationships between rangeland health and ecosystem service provisioning are linear, meaning that there is a one-to-one relationship between rangeland health index scores and the value of the services produced by those ecosystems (Aplet et al., 2000; Esposito et al., 2011; Phillips & McGee, 2014). We recognize that such relationships likely vary in reality—ideally, each combination of ecosystem service and ecosystem health indicator would have its own response curve. However, research on such dynamics is quite limited, and no non-arbitrary way could be found to select other response curves. At a minimum, the linear assumption adopted here provides a straightforward and consistent substitute that could be adapted as better response curve models become available. Overall, this approach represents a conservative means of adjusting ecosystem service values to site-specific conditions. Where primary valuations have been based on less than fully-functioning ecosystems, the value of ecosystem services produced can be assumed to be undervalued—discounting undervalued benefits will produce under-estimates of the total value provided.

BOX 1. RANGELAND HEALTH-ECOSYSTEM SERVICE RELATIONSHIPS

The hypothetical curves in the figure below demonstrate the range of potential relationships between ecosystem health and productivity. The true relationships are unknown. Ideally, one could specify a unique response curve for each health attribute-ecosystem service combination (39 different relationships for the 13 services valued in this report). This could further be differentiated by the landcover type providing the service as well. This report assumes a one-to-one relationship (the green line), but the gray dashed curves represent other possible options, including non-linear and greater- or less-than one-to-one linear relationships.



In this way, we adjusted ecosystem service values by the average MLRA-level health attribute score for each landcover type present, meaning each of the three range health attributes contributes equally to ecosystem service value. For example, an MLRA with an average rangeland health index of 0.5 would be credited with half the ecosystem service value it might have with fully-functioning (1.0) rangeland ecosystems. These dollar-per-acre values were then scaled by the acreage of the associated landcover-attribute combination within each MLRA (e.g. forests within riparian zones). The economic value per landcover-attribute combination (the sum of all valued ecosystem services for that combination) were then summed across all landcover types in each MLRA to produce a total ecosystem service value per MLRA, as follows:

$$(1)\ ESV = \Sigma_{m\,n,j}(Acres_{nj} \times BH_j \times D_{mn})$$

Where:

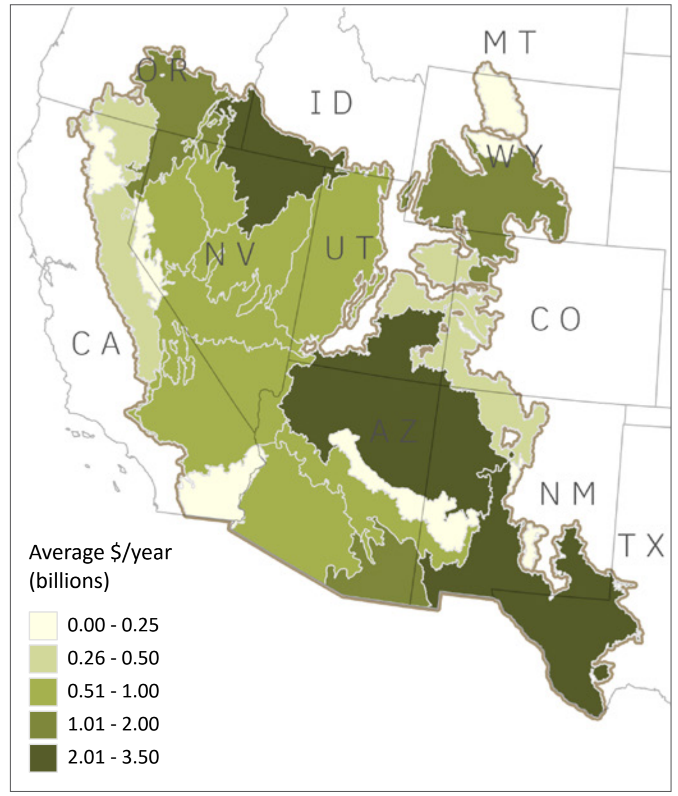
- *ESV* — total baseline ecosystem services ($/year) produced in LRR D
- $Acres_{n,j}$ — the number of acres of landcover-attribute combination *n* in MLRA *j*
- BH_j — the weighted average of the median health attribute index scores in MLRA *j*
- $D_{m,n}$ — the dollar-per-acre-per-year value of each ecosystem service m provided from each landcover-attribute combination *n*

The value of annual ecosystem services represents the continuous year-over-year contribution of rangelands in the study area to human well-being at current rangeland health conditions. Again, these are conservative estimates, as it was not possible to value every ecosystem service on all landcover-attribute combinations—the contributions presented here are only partial estimates.

3.2.4.1 RESULTS

The analytical framework estimates that the aggregated baseline ecosystem service value provided by rangelands within the study area (section 3.2.1) ranges from $13.9 billion to $30.0 billion each year. Figure 10 shows the distribution of the average baseline ecosystem service values (in millions of $/year) provided by rangeland for each MLRA within the study area. Table 10 shows the annual monetized baseline ecosystem service value broken down by landcover type in the study area as well as the average per-acre-per-year ecosystem service value, calculated over the subset of ecosystem services included in this analysis.

FIGURE 10. DISTRIBUTION OF THE TOTAL DISCOUNTED BASELINE ECOSYSTEM SERVICE VALUE IN THE STUDY AREA ACROSS BOTH NON-FEDERAL AND BLM RANGELANDS



Sources: USDA, BLM, USFS, USGS | © 2023 Earth Economics

TABLE 10. BASELINE ANNUAL MONETIZED RANGELAND ECOSYSTEM SERVICE VALUE FOR STUDY AREA

RANGELAND OWNERSHIP	LANDCOVER	ACRES (MILLIONS)	$ YEAR^{-1} (MILLIONS) LOW	HIGH	AVERAGE $ ACRE^{-1} YEAR^{-1} LOW	HIGH
Non-Federal	Forest	3.2	$438	$2,746	$138	$865
	Grassland	13.7	$4,607	$7,055	$335	$514
	Shrubland	72.7	$1,757	$3,574	$24	$49
Subtotal		89.6	$6,802	$13,375	$76	$149
BLM	Forest	1.3	$177	$1,078	$132	$800
	Grassland	14.1	$4,559	$7,224	$322	$511
	Shrubland	87.9	$2,398	$6,287	$27	$72
Subtotal		103.4	$7,135	$14,589	$69	$141
Grand Total		193.0	$13,937	$27,964	$72	$145

3.3 CALCULATING THE EFFECTS OF CONSERVATION PRACTICES

3.3.1 PRACTICE APPLICATION

NRCS issues payments to producers when a practice in a conservation plan and contract (that meets specific requirements) has been implemented. We received contract data (after personally identifiable information had been removed) from NRCS for all contracts implemented in LRR D from 2011 to 2020. For BLM conservation investments, we accessed the Land Treatment Digital Library (LTDL), extracting practices for assessment from 2016 through 2020 within LRR D that are similar to the NRCS conservation practices of interest.[xiii]

From 2011-2020, NRCS issued 7,956 payments for the Brush Management, Prescribed Grazing, and Herbaceous Weed Treatment conservation practices, meaning these practices were implemented nearly 8,000 times in LRR D over ten years, roughly 30 percent of all practice applications in the LRR during that period. Between 2016 and 2020, BLM implemented Brush Management and Herbaceous Weed Treatment 525 times.

Although Prescribed Grazing is not tracked in the LTDL, we have assumed it is implemented on all rangelands treated by BLM based on allotment management requirements that specify approved stocking rate, season(s) of grazing, utilization measurements, etc., which are in broad alignment with NRCS Prescribed Grazing requirements.

We compared demographic indicators reported in the NRCS practice data to the median values for LRR D reported in the Agricultural Census (Figure 11). Error bars indicate the standard error reported by the Agricultural Census. Within LRR D, more NRCS contracts were awarded to white men and early career producers than average, with fewer contracts going to American Indians/Alaskan Natives and female producers.[xiv]

3.3.2 AFFECTED ACRES

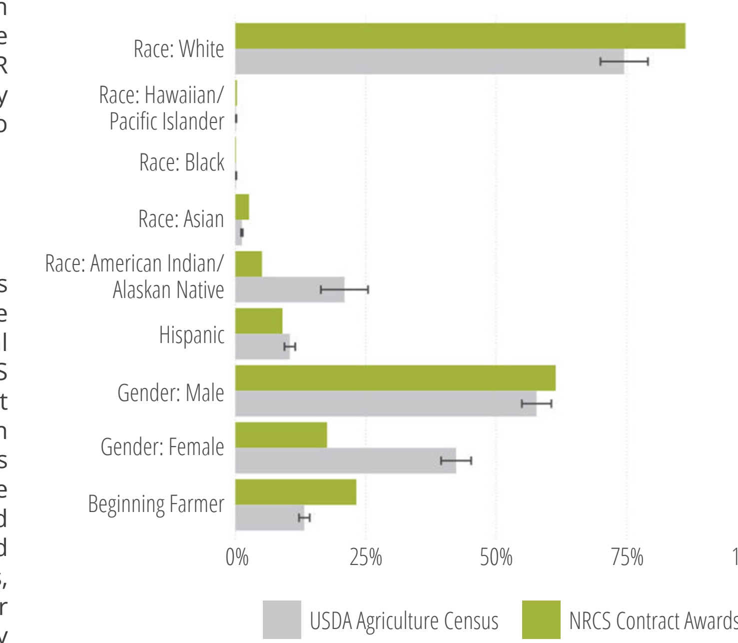
Because contract data have been anonymized and practices could have been implemented repeatedly on the same rangeland acres, we were unable to determine the total unique acres where practices were applied. The NRCS National Planning and Agreement Database (NPAD) does not (consistently) distinguish whether lands treated in a given contract have been treated previously. Brush Management is a practice that often requires repeated treatment to achieve reduction targets for undesirable woody plants. The same could be said of Prescribed Grazing—it may need to be implemented continuously on the same lands to achieve desired objectives, so the same rangelands may receive NRCS cost-share for multiple years on the same contract. Because of this ambiguity (and because cost-share data had been aggregated to the

TABLE 11. NUMBER OF APPLICATIONS OF CONSERVATION PRACTICES ON RANGELANDS IN LRR D FROM 2011-2020 ON NON-FEDERAL RANGELANDS AND 2016-2020 ON BLM-MANAGED RANGELANDS

PRACTICE	NON-FEDERAL RANGELAND	BLM RANGELAND
Brush Management	4,259	80
Prescribed Grazing	2,902	N/A*
Herbaceous Weed Treatment	795	445

** Prescribed grazing is not tracked in the LTDL.*

FIGURE 11. PERCENT OF NRCS CONTRACTS AWARDED TO DEMOGRAPHIC GROUPS COMPARED TO PROPORTION OF LRR D PRODUCERS



[xiii] Practice counts on BLM rangeland are likely underestimates as the LTDL does not include practices tracked in the Vegetation Management Action Portal (VMAP), which is at the time of writing in the process of being integrated into the LTDL.

[xiv] Some categories may not appear to sum correctly due to respondents not reporting such attributes, as in the case of gender.

county level), it was impossible to distinguish whether NRCS cost-sharing supported retreatment or expanded treatment to other rangelands within the same contract.

Lacking finer resolution of NRCS practice data, we recognized the potential to over-estimate ecosystem service benefits by double-counting treated acres. Treating the same location more than once is not expected to yield an equal change in ecosystem function with each repeat application, owing to the law of diminishing returns. To avoid double-counting, we counted implementation acres once per practice per contract, recognizing that this approach may under-value any subsequent effects on ecosystem services. As described in section 3.2.1, the practice data did not distinguish affected landcover types—only total affected acres. Using the average landcover description of land use within each farm tract or county (if no farm tract was identified), we derived the acreage of each landcover affected by a given practice.

BLM's LTDL is a spatial database that associates points, lines, and boundaries with on-the-ground treatments. We used the geodata associated with treatment boundaries in LRR D to clip the landcover layer created from the methods described in section 3.2.1 and quantify acres of landcover types affected by each treatment.

3.3.3 CONSERVATION PRACTICE EFFECTS ON RANGELAND HEALTH ATTRIBUTES

We then conducted a review of the practice-effects literature to quantify the impacts of specific conservation practices on rangeland health attributes. Each study was evaluated by a NRCS CEAP-Grazing Lands technical sub-team for methodological quality and relevance to the research scope. All applicable data was entered into The Conservation Outcomes Research Explorer (CORE).

CORE is a PostgreSQL database under development and created by the Grazing Lands component of the Conservation Effects Assessment Project (CEAP). CORE is intended to act as an ongoing, dynamic NRCS library of categorized research, aligning research data with NRCS conservation practices, resource concerns, land health attributes, and potentially economic evaluation of conservation measures. It houses peer-reviewed, published research studies organized by land use (e.g. rangeland, pastureland, cropland, forestland); all study sites have been geo-referenced to the degree possible based on the data contained in each study. Data for each research study, such as climate, soils, MLRA, ecological sites, treatment methods, and use history can be entered with as much detail as provided in the research. Field data measured by each study are entered into CORE as actual measurements or as graph-estimations directly from published figures and tables. This allows for calculation of actual or relative differences found for measured resource elements as compared to a control (or pre-treatment), so that the percent-change, magnitude of change, and direction of change can be determined for specific treatments. Because each measured data element is linked to at least one rangeland health attribute, and treatments are linked to one or more conservation practices, the CORE database informs our methodology on the effectiveness of a particular practice in improving measures of rangeland health.

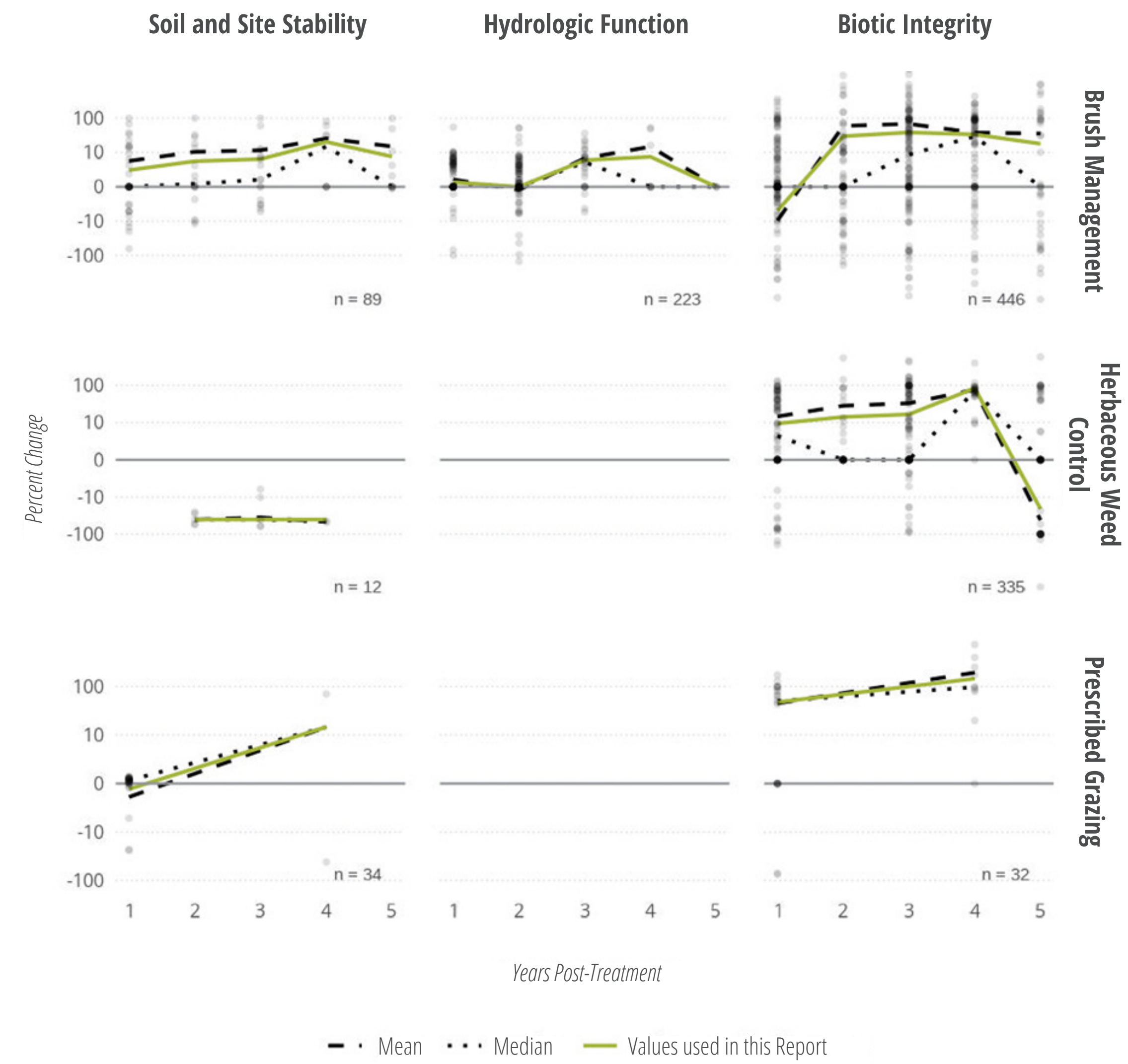
Data within CORE are formatted to support determination of change-over-time metrics—including amount, rate, and duration—for each natural resource element measured in each study. Data captured for soils and other site conditions showcase relative effects on different ecological sites across MLRAs. By linking treatments to conservation practices, the effect of the conservation system can be determined for each action. This database improves on the initial framework developed in Fletcher et al. (2020) by providing direct percent-change metrics from research, rather than relying on qualitative indices of practice effectiveness.

Transcribing studies into CORE requires that study results be characterized as a unitless percent change between applied practices and controls to enable comparison across multiple unit types. We selected twenty-eight studies relevant to the rangeland health attributes and practices for this analysis (see Appendix C for a list of references). From these, we recorded more than 1,000 proportional relationships between specific health indicators and specific conservation practices. These changes could be either positive (e.g. health improvements from reducing bare ground) or negative (e.g. worsening health due to increased erosion).

Proportional changes are also recorded based on the number of years since initial treatment, allowing us to describe a practice's effectiveness over time. We limited the period of effect to 5 years—while practices may be effective for more than 5 years, published research for longer periods was scarce. High data variability meant that it was not possible to fit appropriate statistical models to the data. Instead, we determined mean and median values, health attributes, and years since initial implementation to derive a "practice effectiveness curve" for each practice. We then traced an effectiveness function based on the average of mean and median values for each practice year to determine practice effects on our baseline health index scores. Figure 12 shows the statistics and values used for each combination of health attribute and conservation practice, along with the number of data points used to derive those relationships.

Where contracts apply more than one practice at a time, we assume benefits to be additive. Stacking practices and developing comprehensive conservation plans is a proven approach to treating resource concerns, and literature suggests that effects of multiple conservation practices applied at the same time are greater than single-practice applications. Baffaut et al. (2020) suggest that this practice stacking can counteract negative aspects of some practices with benefits from others, indicating greater benefits when multiple practices are implemented at once. Francesconi et al. (2015) find that nutrient reductions were 1.7 to 10.5 times greater for two- and three-practice systems compared to single-practice systems. Law et al. (2020) and Blanco-Canqui et al. (2011) also find greater benefits with stacked practices for nutrient loading and soil properties, respectively. As with the relationship between range health and ecosystem service provision, more research should be conducted to characterize interactions between stacking practices and environmental effects, whether those are linear, multiplicative, or some other nonlinear relationship. These references support that stacking practices are at least partially additive. Assuming that effects are additive may overestimate the impacts of practice systems, but due to the many gaps in the rest of the framework (e.g. unable to value more than three practices and only thirteen ecosystem services), we believe that overall, the results are still conservative with this assumption.

FIGURE 12. TREND OF PERCENT CHANGE VALUES DERIVED FROM CORE FOR UP TO FIVE YEARS POST-TREATMENT, BY HEALTH ATTRIBUTE AND CONSERVATION PRACTICE*



Note: While the practice year values for each effectiveness function are the average of the mean and median effectiveness values recorded in CORE, some do not appear as such because the y-axes are logarithmic.

3.4 QUANTIFYING ECOSYSTEM SERVICE CHANGES DUE TO CONSERVATION PRACTICES

The final step to identify changes in ecosystem service production and valuation due to conservation practices combines all steps described previously in this report. First, we identified the specific conservation practices implemented and the total acreage of each landcover-attribute combination to which they were applied (see sections 3.2.1). Where practices were applied multiple times on a contract or project, we counted only the earliest application of that practice to avoid double-counting. We applied associated changes in ecosystem health (section 3.3) to the rangeland health baselines (sections 3.2.2-3.2.4) to estimate changes to ecosystem services by conservation practice in each MLRA.

Again, we assumed that implementation of more than one practice has additive ecosystem health benefits. In other words, a contract certifying two practices (e.g. brush management and herbaceous weed treatment) on the same land in a given year was assumed to change the overall ecosystem health by the sum of the health attribute changes associated with each practice that year. It is possible that in some circumstances, such an assumption may overestimate impacts—conversely, conservation practices may have synergistic effects (i.e. greater than the sum of their parts) on at least some ecosystem services and associated resource concerns. In the context of the two practices included in this analysis, we felt that an additive assumption is appropriate, because:

1. we could not determine whether both practices were applied to the same lands;
2. had they been applied to the same acres, research and expert opinion suggests that applying multiple practices to the same land is more effective over time; and
3. the likelihood of diminishing returns and additive benefits must be determined at the field scale.

Such "response curves" would likely vary for each combination of practices and ecoregions. Absent a clear and comprehensive understanding of such dynamics, we acknowledge that assuming that conservation practice effects are universally additive has limitations.

We calculated marginal changes in ecosystem service benefits by multiplying the total change in ecosystem health (averaged across all health attributes) by the associated per-acre ecosystem service value. Benefits were capped when the health index is 1, in other words, there was no potential for improvement in service provisioning in a site at reference condition. We then scaled these marginal changes by the spatial extent of each associated landcover type within a MLRA. To yield the total change in ecosystem service benefits for each MLRA associated with NRCS contracts for that year, we then summed the scaled marginal changes of all ecosystem services provided each landcover within each MLRA (Equation 2):

$$(1)\ ESV\ C_{ijp} = \Sigma_{kmnp} \frac{H_{ijklp}}{3} \times D_{mn} \times A_{np}$$

Where:

ESV_{ijp}	is the change in ecosystem service value in year *i* in MLRA *j* by practice *p*
H_{ijklp}	is the percent change in health attribute *k* from practice *p* for MLRA *j* in year *i*
A_{np}	is the landcover acres *n* affected on the practice *p*
D_{mn}	is the dollar-per-acre-per-year ecosystem service value for service *m* and landcover *n*

Because applied practices are aggregated to the MLRA level to preserve producer confidentiality for the NRCS data, we were unable to determine whether any acres received repeated treatments across multiple contracts. This limited our ability to determine health index scores of any given acre for prior years. Instead, at the end of each year, we calculated new ecosystem health scores for each MLRA by adding the ecosystem health change associated with that year (attributable to practices) to the previous annual health score, weighted by county-level landcover acres treated. These new index values were then used as the baseline for the subsequent year. This process was repeated each year (refer to Figure 4 for a simplified example). Because practices certified in a given year are likely to provide residual effects (see section 3.3), the effects of both newly-certified practices and the residual effects from previously-certified practices were combined until the 5-year limit.

3.4.1 ESTIMATED VALUE ATTRIBUTABLE TO CONSERVATION PRACTICES

To review, this study estimated the value of ecosystem services produced on non-federal and BLM-managed rangelands in LRR D that could be attributed to the implementation of three NRCS conservation practices (Brush Management, Herbaceous Weed Treatment, Prescribed Grazing). Because conservation practices could have effects beyond their initial application year, we estimated impacts for 5 years post-treatment and assessed these benefits relative to annually adjusted baseline estimates of the average annual ecosystem services produced throughout the study area.

For non-federal rangelands, we estimate that Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing practices implemented between 2011 and 2020 increased the non-market value of ecosystem services across the study area between $78 million and $214 million,[xv] an average of $8 million to $21 million per year. On a per-acre basis, this amounts to an additional $5 to $15 annually per treated acre.

[xv] Payments made on contracts implementing these conservation practices during this time period total $131.7 million ($69.9 million, $2.7 million, and $59.1 million for Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing, respectively).

On BLM rangelands, we find that Brush Management, Herbaceous Weed Treatment, and Prescribed Grazing practices implemented between 2016 and 2020 increased the non-market value of ecosystem services on BLM rangelands within the study area between $22 million and $37 million, an average of $6 million to $9 million per year. On a per-acre basis, this amounts to $6 to $11 annually per treated acre. Note that because fewer projects were recorded in the LTDL as Brush Management or Herbaceous Weed Treatment, many MLRAs have null valuation data in Figure 13.

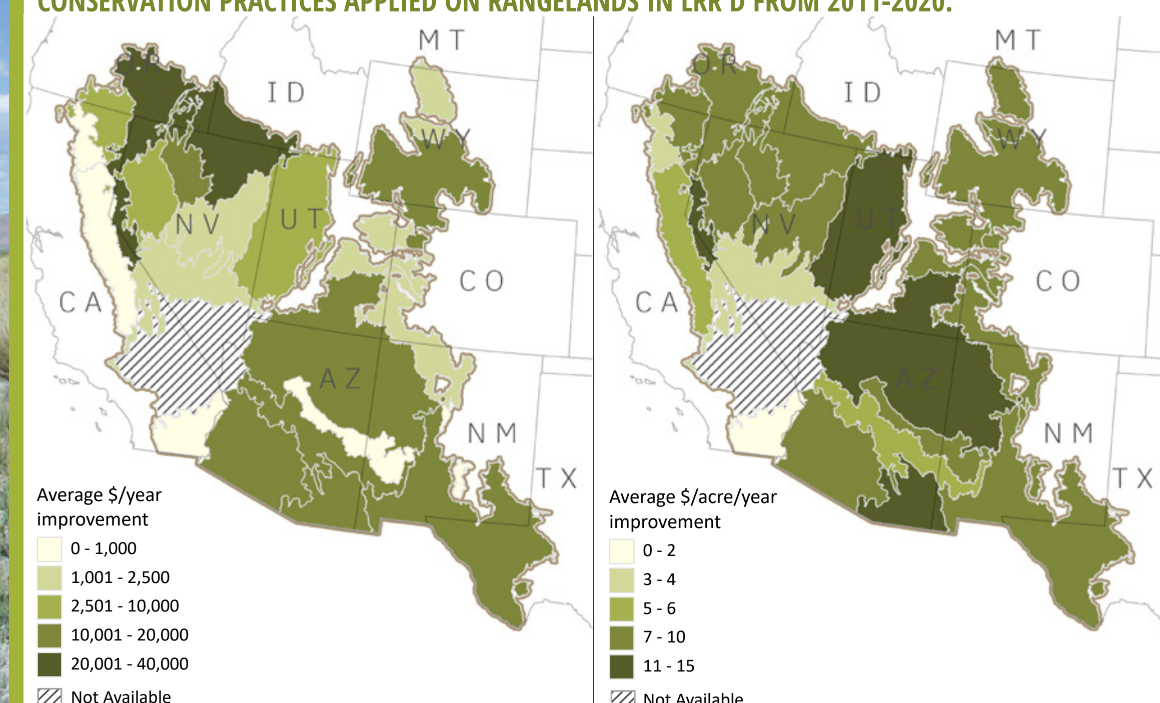
Figure 13 shows the average annual ecosystem service improvement for each MLRA in the study region, combined across non-federal and BLM management. Table 12 shows the average dollar-per-acre improvement by practice. Results for both non-federal and BLM-managed rangelands are similar, despite BLM lands starting in worse condition (see section 3.2.2) and fewer instances of the three practices being implemented annually on BLM lands than on non-federal rangelands. However, the average area of BLM treatments tended to be larger than the affected area for NRCS contracts. Relative to the baseline, these practices increased the value of ecosystem service benefits about 0.1 percent of the total annual baseline ecosystem service value each year.

[x] As mentioned previously, counties were assigned to only one LRR and MLRA boundary to alleviate multiple MLRA and LRR boundary overlaps within counties.

TABLE 12. AVERAGE ANNUAL IMPROVEMENT IN MONETIZED ECOSYSTEM SERVICE VALUE DUE TO THREE RANGELAND CONSERVATION PRACTICES APPLIED IN LRR D ON NON-FEDERAL RANGELANDS FROM 2011-2020 AND BLM-MANAGED RANGELANDS FROM 2016-2020.

RANGELAND OWNERSHIP	NON-FEDERAL RANGELAND		BLM-MANAGED RANGELAND	
	LOW	HIGH	LOW	HIGH
Brush Management (314)	$5	$14	$4	$9
Herbaceous Weed Treatment (315)	$10	$26	$12	$21
Prescribed Grazing (528)	$2	$5	$2	$4

FIGURE 13. DISTRIBUTION OF AVERAGE ANNUAL CHANGE IN ECOSYSTEM SERVICE VALUE (ANNUAL, LEFT; PER-ACRE, RIGHT) DUE TO THREE RANGELAND CONSERVATION PRACTICES APPLIED ON RANGELANDS IN LRR D FROM 2011-2020.



Sources: USDA, BLM, USGS | © 2023 Earth Economics

4. LIMITATIONS AND SENSITIVITIES

All estimation approaches have strengths and weaknesses. Benefit transfer methods (BTM) estimate the economic value of a given ecosystem based on studies of similar ecosystems in similar contexts. As with any effort to generalize, the main limitation in applying BTM to value ecosystem services is recognition that each ecosystem, along with the hydrological, chemical, biological, social, and economic conditions influencing its values, is unique. This may limit the validity of assuming that unit values (e.g. $/acre/year) derived in one location are relevant to other sites, as well as the fact that the per-acre value of services is constant throughout each MLRA. The method also assumes that the values selected from the literature are representative of the area being studied. Bias may also be introduced when selecting valuation studies to form a dataset—however, conducting secondary reviews for appropriateness and rigor limit the potential for this, and reporting ranges rather than single value estimates (e.g. averages) partially mitigates remaining issues. Conducting primary valuation of ecosystem services produced on a site-by-site basis is cost-prohibitive and cannot be completed in a timely manner. BTM is a widely accepted, and replicable, approach to developing estimates that are helpful for informing current decisions about conservation investments and priorities.

The studies on which we based our calculations encompass a range of geographic areas, socioeconomic conditions, analytical methods, investigators, and time periods. Many provided a range of valuation estimates, rather than single-point values. We have preserved this variance here; no studies were excluded because their estimates were deemed "too high" or "too low." We performed a sensitivity analysis relating to the per-acre values from the BTM dataset, as reported in section 4.1.1, below.

A large limitation of this work is the inability to provide a value for every ecosystem service on every landcover type due to gaps in the published literature, as well as sparse understanding on relationships between conservation practices and rangeland health attributes. The effect is to significantly underestimate the full value produced by any given ecosystem or conservation practice. More comprehensive research on the ecosystem services provided by rangelands would almost certainly affect the values estimated here, and quite likely increase estimate totals. Furthermore, lack of literature on relationships between conservation practices, ecosystem health, and ecosystem service value has led to several assumptions throughout the analytical framework—additional research in these areas should improve the specificity of these results.

Our study presents ranges of estimates of the value of ecosystem services over time (years) and space (FSA tracts and MLRAs). These estimates have limits, as indicated above. However, we believe this report improves our understanding of the value of NRCS and BLM conservation practices in two ways. First, the analytical framework offers an example of how it is possible to use available data in a cost-effective manner to develop reasonable estimates of the value of ecosystem services produced across large regions. Second, these specific results may provide a broad indication of the scale of benefits which these conservation practices provide to local and downstream communities, helping us better understand program effectiveness and other important factors associated with national conservation programs carried out at local scales.

With better information about the links between land use, management and conservation practices, and the benefits provided by healthy ecosystems, policy makers and program managers may find it easier to generate support for—or promote the adoption of—more sustainable practices and suites of practices. Estimates of the value of ecosystem services based on the best available literature and established methodologies can broaden understanding of how decisions—and investments—might impact constituents and the land base. This applies not only to the agricultural sector, but also those industries and communities which rely on ecosystem services that are strongly affected by agricultural practices (e.g. water quality). Such information may inform decisions ranging from selecting the most cost-effective practices, to estimating fee-for-service compensation for producers. In a world of limited choices, understanding the relative value provided by alternatives is critical to effective decision making.

4.1.1 SENSITIVITY ANALYSIS OF BASELINE ESTIMATES

No estimation technique is capable of perfect prediction—variance between what is estimated and what is ultimately observed is known as model error. This error is often quantified by calculating the absolute percent difference between a transferred value and actual values at a study site (Magalhães Filho et al., 2021). While there is no consensus on a maximum level of error acceptable for different applications of BTM (Magalhães Filho et al., 2021), studies quantifying error in BTM find that, on average, well-designed benefit transfers can produce errors of up to 42 percent (meaning transferred values are 42 percent lower or higher than study site values) (Boyle & Parmeter, 2017). While the best way to estimate BTM error would be to compare predicted ecosystem service values to observed values, this would require primary valuation studies for each ecosystem service at each study site, which would be prohibitively expensive and time-consuming. Again, one of the strengths of BTM is that it is a cost-effective and timely means of producing reasonable estimates where data does not already exist.

Sensitivity analysis is an important means of assessing the external validity of the study by investigating the outcomes of changing parameters of the analysis (Aschonitis et al., 2016). One approach to conducting a sensitivity analysis for the BTM is to assess how much value estimates vary based on changes in the supporting data (Boyle & Parmeter, 2017). In general,

BTM efforts that are less-sensitive to single data points are considered more robust. Absent corroborating evidence, researchers sometimes choose to omit outlying data points as "recording" errors. Yet in some cases, outliers may accurately reflect study site conditions. Here, we assess BTM sensitivity as the proportional change in total ecosystem service value per acre as each ecosystem-ecosystem service combination valued was dropped from the dataset.

Table 14 shows the average relative contribution of each landcover and ecosystem service value to the overall baseline estimate. The largest total value in the study area comes from the soil fertility service on grasslands, followed by grassland habitat and shrubland carbon sequestration. This is due to the combination of high acreage (see Table 3) and relatively higher per-acre values on the corresponding landcover types (see Figure 9).

TABLE 14. PROPORTION OF BASELINE TOTAL ECOSYSTEM SERVICE VALUATION (ESV) IN THE STUDY AREA.

ECOSYSTEM SERVICES VALUED	FOREST	GRASSLAND	SHRUBLAND
Aesthetics	<0.5%	<0.5%	3%
Air quality	<0.5%	<0.5%	<0.5%
Biological control	<0.5%	<0.5%	<0.5%
Carbon sequestration	4%	5%	17%
Fire risk reduction		1%	5%
Forage production		<0.5%	1%
Habitat		16%	
Recreation	<0.5%	1%	4%
Social		<0.5%	2%
Soil fertility		23%	
Soil retention		<0.5%	1%
Waste treatment		11%	
Water supply	5%	2%	

5. DISCUSSION

In January of 2023, the White House published a strategy to incorporate the value of natural capital into official federal economic statistics (Office of Science and Technology Policy, 2023). The strategy acknowledges,

> "People depend on nature to supply important services and economic opportunities. For example, families escape their daily grinds to recreate in nature and travel to experience majestic mountains and tranquil beaches; soils, water, and bees work with America's farmers to grow food; and trees, grasses, and other plants are the original carbon capture and storage system and also filter other pollutants, complementing the efforts of nurses and doctors to make Americans healthier and more productive. With every passing year, scientists, innovators, and economists discover more evidence about how the economy relies on nature and how economic activities change nature's ability to provide services." (p. vi)

The acknowledgement that ecosystem services have both social and economic value, combined with the strategy goal of creating a more inclusive national accounting system to help secure "nature-dependent economic activities," provides an opportunity for NRCS and BLM to explore the use of ecosystem service valuation in the planning and programmatic processes.

The framework outlined in this report illustrates that conservation assistance on grazing lands has ecosystem benefits that extend beyond fence lines (e.g. air and water quality improvements, disaster risk reduction), allowing us to associate conservation practices on both private and non-private lands with benefits to taxpayers.

In the spirit of the constantly-evolving literature surrounding ecosystem services, which has grown exponentially in the past few decades, we intended to improve upon the initial framework set forth in Fletcher et al. (2020)—including more (and more detailed) data on conservation effects to rangeland health, basing those relationships on published science, including additional conservation practices in the analysis, and improving benefit-transfer estimates through the use of function transfer. Furthermore, we show how the framework can be expanded to rangelands managed by other federal agencies aside from the NRCS, as well as demonstrate its replicability by implementing it in a second LRR.

For many years, the public has sought a more comprehensive understanding of the environmental and economic effects of conservation programs and practices. The estimates we present in this report reveal the breadth and magnitude of economic benefits that conservation practices can generate. Despite data constraints that limited the granularity and precision of the analysis, these results provide a broad sense of the economic importance of these select rangeland conservation actions. It represents an initial step toward understanding the benefits of conserving and improving ecosystem health through sustainable management for the economic well-being of communities throughout the region.

Expanding public awareness of the value of goods and services provided by natural capital strengthens our shared understanding about the synergy between our environment, our communities, social well-being, and our economy. We believe such understanding will increase support for public financing of land conservation and stewardship. This report can be used to make the connection between conservation actions and the multiple benefits they provide to nearby and downstream communities, which may inform decision-making, prioritize the most effective practices, design incentive programs to reward land managers for voluntary conservation efforts, or even lead producers to choose practices they may not otherwise adopt.

5.1 RECOMMENDATIONS

Given the limitations of data and relevant literature, these findings establish a starting point for ongoing discussion and research that could be facilitated through academic publications or federal funding for rangeland research to help fill gaps. This study should not be taken as a conclusive analysis of the value provided by ecosystems within the study region or the practices implemented on those lands, as the process of identifying and monetizing benefits provided by conservation practices in the study area has revealed a number of data gaps and next steps to improve study resolution and comprehensiveness:

FILL GAPS IN ECOSYSTEM SERVICE ASSESSMENTS

Primary ecosystem valuation research on natural ecosystems dominated by grasses or shrubs (i.e. rangelands) is quite limited. This means we have only been able to value a fragment of ecosystem goods and services provided by rangelands. Furthermore, valuation literature for wetlands in the southwestern United States is also limited. A number of landcover-ecosystem service combinations could not be valued due to these limitations. Expanding primary valuation research of ecosystem services provided by rangeland landcover types would help address these gaps. As such, the values presented in this report likely underestimate the true value of ecosystem services provided throughout the region, and the influence of NRCS conservation practices on those benefits.

FILL GAPS IN RESEARCH ON HOW CONSERVATION PRACTICES AFFECT LAND HEALTH

Only three of sixty-three conservation practices applied during 2011-2020 in LRR D were valued in this analysis. Expanded primary research on the quantitative effects of conservation practices on ecosystem health is also needed, including effects of implementing multiple practices simultaneously. We also acknowledge that several practices do not appear to directly impact resource concerns. Fences and water-related practices (known as facilitating practices within NRCS) are often needed for Prescribed Grazing to function as designed. Many NRCS contracts include facilitating practices and costs to improve grazing management impacts, but in this study, we evaluated only practices with direct impacts. Additional research is needed on the relationships between facilitating practices and management practices, as well as the virtue of applying facilitating practices in isolation.

5.2 NEXT STEPS

This work is intended to be an evolving framework that can be updated as additional literature and methodologies around ecosystem valuation are published. Aside from our recommendations for addressing gaps in the literature and data (Section 5.1), there are several additional research avenues that could be pursued to further improve the analytical framework.

MARGINAL VS. AVERAGE VALUES

This study uses average per-acre values for ecosystem services, but literature also exists that examines the marginal change in dollar value for ecosystem services due to changes in biophysical metrics, such as how changes in water quality may impact recreational or aesthetic values. Marginal value research could be applied directly to outputs from CORE, avoiding assumptions around the use of the rangeland health indices.

VALUE OF ADDITIONAL PRACTICES

Funding allocation may be better informed by comparison of the marginal value of an additional practice implemented rather than improvements to rangeland health. This would be aided by including additional practices into the analytical framework (see section 5.1) and expanding the CORE database with additional literature.

CONDUCT ANALYSIS ON OTHER LAND USES

Establishing health metrics for other land uses (e.g. cropland, forestland) could broaden the scope and scale of subsequent analyses. This work shows how the framework could be applied to rangelands, but the framework could be applied to other land uses, provided the relevant ecosystem health data are available.

ESTABLISH FUNCTIONAL FORMS FOR EACH HEALTH ATTRIBUTE AND ECOSYSTEM SERVICE PAIRING

Due to a paucity of appropriate modeling, research, and other data relevant to the study area, the framework assumes 1) linear relationships between health indices and ecosystem services, and 2) the three rangeland health attributes equally affect each ecosystem service. We recognize that these relationships could take on any number of forms in actuality, and that some attributes may not affect every service or that some services may not be affected at all by a health attribute. Through published research or workshops with experts, a more true-to-life representation of these relationships would improve estimates of changes in ecosystem service value attributable to conservation practices.

6. APPENDIX A
ECOSYSTEM SERVICE VALUE RESULTS BY MLRA

TABLE 15. BASELINE ANNUAL MONETIZED RANGELAND ECOSYSTEM SERVICE VALUE BY MLRA (2021 USD)

	Non-Federal Rangelands				BLM-Managed Rangelands			
	$/Year (Millions)		$/Acre/Year		$/Year (Millions)		$/Acre/Year	
MLRA	Low	High	Low	High	Low	High	Low	High
21	126	397	114	359	110	213	113	219
23	210	360	92	158	532	1,046	70	138
24	106	169	131	209	422	742	110	194
25	449	738	118	194	1,559	2,743	126	221
26	19	41	62	135	36	80	64	142
27	119	200	87	146	326	658	71	143
29	21	41	42	81	560	1,333	46	110
30	100	175	59	104	479	1,000	63	133
31	26	46	59	104	94	192	58	118
32	25	42	86	143	100	183	64	116
35	1,382	2,982	63	135	343	951	40	111
36	167	435	55	143	63	209	37	123
38	305	790	74	193	85	178	75	157
39	133	281	119	252	*	*	*	*
40	435	756	59	102	229	491	56	119
41	500	908	90	164	227	412	123	223
42	1,679	2,887	82	142	530	937	91	160
22A	104	524	140	704	*	*	*	*
22B	*	*	*	*	2	8	142	492
28A	243	433	71	127	435	884	66	133
28B	120	200	114	190	428	992	62	143
34A	478	853	70	124	408	934	41	94
34B	58	119	50	104	167	405	51	123

*No data

TABLE 16. AVERAGE ANNUAL IMPROVEMENT IN MONETIZED ECOSYSTEM SERVICE VALUE DUE TO CONSERVATION PRACTICES APPLIED BY MLRA

	Non-Federal Rangelands				BLM-Managed Rangelands			
	$/Year (Millions)		$/Acre/Year		$/Year (Millions)		$/Acre/Year	
MLRA	Low	High	Low	High	Low	High	Low	High
21	1,263	3,453	3	9	3,554	8,822	7	18
23	1,551	4,381	4	11	17,951	30,249	5	9
24	3,823	10,384	5	12	3,438	6,168	4	7
25	1,167	3,309	2	7	22,410	37,181	7	12
26	2,190	6,364	8	24	26,335	42,837	6	10
27	492	1,413	2	6	4,997	9,472	8	12
29	817	1,816	3	8	5	12	1	2
30	*	*	*	*	*	*	*	*
31	847	2,279	5	10	*	*	*	*
32	8,151	27,264	6	18	*	*	*	*
35	598	1,596	4	12	*	*	*	*
36	10,111	29,018	3	8	91	554	2	14
38	373	1,072	6	16	*	*	*	*
39	11,029	30,279	4	10	*	*	*	*
40	10,111	27,913	6	16	*	*	*	*
41	7,514	16,915	5	12	*	*	*	*
42	147	423	3	8	*	*	*	*
22A	*	*	*	*	*	*	*	*
22B	4,482	12,442	8	22	*	*	*	*
28A	1,240	3,572	5	13	*	*	*	*
28B	10,215	28,584	4	11	*	*	*	*
34A	1,306	3,445	5	15	*	*	*	*
34B	1,263	3,453	3	9	3,554	8,822	7	18

*No data

7. APPENDIX B
ECOSYSTEM SERVICE VALUATION ANNOTATED BIBLIOGRAPHY

The following references were used to quantify the economic valuation of ecosystem services in the framework. Specific information (bulleted items) is provided, showing how/where the valuation reference data was applied. We describe the procedure taken to derive the values used from each study, however, the raw and adjusted dollar values are not included, as those are part of a proprietary database owned and managed by Earth Economics.

Bastian, C. T., McLeod, D. M., Germino, M. J., Reiners, W. A., & Blasko, B. J. (2002). Environmental amenities and agricultural land values: a hedonic model using geographic information systems data. *Ecological Economics*, 40(3), 337-349.

- Site: Wyoming agricultural lands
- Rangeland Landcover Types Applied: Forest, Grassland, Shrubland
- Ecosystem Service(s): Aesthetics
- Valuation Methodology: Hedonic pricing
- Sample Size: 138
- Study Site Annual Household Income: $63,049
- Percent of Population Attaining High School Level Education or Greater: 93.2%

This study applies hedonic pricing to estimate the impact of environmental amenities—scenic view, elk habitat, carrying capacity—on Wyoming agricultural lands. The authors determined that "landcover diversity" (which they interpret as a proxy for scenic views) contributed an average of $45 per acre (with a range of $3 to $62) to agricultural parcels. The values are inflated to 2021 USD using 1995 (the last year of sales data) as the base year and annualized using a rental rate of 12%.

Gopalakrishnan, V., Hirabayashi, S., Ziv, G., & Bakshi, B. R. (2018). Air quality and human health impacts of grasslands and shrublands in the United States. *Atmospheric Environment*, 182, 193-199.

- Site: United States
- Rangeland Landcover Types Applied: Grassland, Shrubland
- Ecosystem Service(s): Air quality
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

This study models improvements in air quality for grasslands and shrublands at the state and county levels in the coterminous United States. The authors use the i-Tree Eco model to estimate the air pollution removal capacity of these landcover types for NO_2, O_3, $PM_{2.5}$, and SO_2. Monetary air quality benefits are derived from the US EPA's BenMAP program, which calculates avoided costs of adverse health effects such as emergency room visits, hospital admissions from respiratory illness, and more. Values for each pollutant removed are published in terms of 2010 USD per hectare per year, by state, for both urban and rural areas within the United States. We took the range and average in the "all lands" values from Arizona, Nevada, New Mexico, and Utah for this study for both grasslands and shrublands. We converted the published values using a conversion factor from hectares to acres (2.47105 acres per hectare) to arrive at 2010 USD per acre per year values. We then adjusted converted values to 2021 USD using the World Bank GDP deflator data. Overall, pollution removal benefits were higher in urban areas, and the authors conclude that grasslands and shrublands are critical components in improving air quality and human health in urban regions of the United States. Supporting information for this article contains estimates of pollution removal by state. Values for Arizona, Nevada, New Mexico, and Utah were used in this study.

Hill, B. H., Kolka, R. K., McCormick, F. H., & Starry, M. A. (2014). A synoptic survey of ecosystem services from headwater catchments in the United States. *Ecosystem Services*, 7, 106-115.

- Site: United States
- Rangeland Landcover Types Applied: Forest, Grassland
- Ecosystem Service(s): Water supply
- Valuation Methodology: Avoided cost
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

This study estimates water supply, climate regulation, and water purification for over 500 headwater stream catchments, using data derived from the National Hydrography Dataset for the lower 48 states. Production functions were created for water supply, climate regulation, and water purification and results were reported for nine ecoregions. The combined ecosystem services—valued at up to $30 million per year overall—were presented in 2013 USD per hectare per year. This study uses water supply value estimates for the Xeric Catchments region, which covers the majority of LRR D, and proportioned the value to forest and grassland based on the proportion of the catchment in each of those landcovers. We converted the published values using a conversion factor from hectares to acres (2.47105 acres per hectare) to arrive at 2013 USD per acre per year values. We then adjusted converted values to 2021 USD using the World Bank GDP deflator data. The authors estimate value by ecoregion. In this study, we used water supply values for the Xeric catchment region, which covers the southwest US.

Liu, H., Hou, L., Kang, N., Nan, Z., & Huang, J. (2022). The economic value of grassland ecosystem services: A global meta-analysis. *Grassland Research*, 1(1), 63-74.

- Site: Global
- Rangeland Landcover Types Applied: Grassland
- Ecosystem Service(s): Habitat, Soil fertility, Waste treatment, Water supply
- Valuation Methodology: Meta Analysis
- Sample Size: 702
- Study Site Annual Household Income: NA
- Percent of Population Attaining High School Level Education or Greater: NA

This study provides a comprehensive assessment of the value of ecosystem services provided by grasslands. The authors construct a global database of grassland ecosystem service values containing 702 observations from 134 primary studies. A linear meta-regression of this database reveals the total value of ecosystem services provided by grasslands ranges from $3955 to $5466 per hectare and that regulating services have the highest value.

Monte Carlo simulation and function transfer was used with the study results to estimate more site-specific values for LRR D. Table 4 in the paper presents the model coefficients estimated in the study, and Table 1 presents descriptions and statistics for each variable in the model. Dollar-per-hectare-per-year values are calculated with a linear regression model:

$$Y = \beta + \sum_{j=1}^{J} a_j X_j \times eH_j$$

where β is the constant, a_j is a vector of coefficients for the independent variables, X_j is a vector of inputs for each independent variable, subscript j is the number of variables, and e is the normally distributed error term. We convert model results to 2021 $/acre/year using GDP deflators and the conversion factor from hectares to acres.

The table below shows the inputs used in the function transfer simulations. A distribution was defined for the GDP per capita variable from data specific to LRR D, specifically, county-level estimates for all counties with at least 50% of their area within LRR D from the Bureau of Economic Analysis Regional Economic Accounts. We used the lognormal distribution as it provided the best fit according to the Anderson-Darling, Chi Square, and K-S tests. The semidesert grasslands variable was set to 1 as it corresponds best to the climate within LRR D. "Year of research" was set to the sample mean. Latitude and longitude were set to the midpoint of LRR D. "Asian studies" was set to 0, since our site is not in Asia and this also creates a more conservative estimate as the coefficient is positive. We averaged outputs using avoided costs and replacement costs for regulating services (soil fertility, waste treatment, water supply), set "contingent valuation" to 1 for "Genetic diversity" as these are common methodologies used to value these services. For this report, we are considering "genetic diversity" to describe habitat values as Liu et al. define this category as "biodiversity protection." We did not use results for some ecosystem services because either a) the values were significantly different than local values already existing in our dataset (recreation, climate regulation, raw materials), or, b) we felt they may double-count other services (water flows). Finally, values were calculated over 10,000 trials. *See table on following page.*

VARIABLE	VARIABLE DESCRIPTION	COEFFICIENT	DISTRIBUTION /INPUT	MEAN	STANDARD DEVIATION
Mediterranean grasslands	Category dummy variable (1 = yes, 0 = no)	-86.372	0	0.06	0.24
Temperate grasslands	Category dummy variable (1 = yes, 0 = no)	-121.102	0	0.38	0.49
Semidesert grasslands	Category dummy variable (1 = yes, 0 = no)	-151.034	1	0.18	0.38
Grasslands (unspecified)	Category dummy variable (1 = yes, 0 = no)	-102.943	0	0.04	0.19
Protected grassland	1 = protected, 0 = unprotected	66.531	0	0.12	0.32
Raw materials	Category dummy variable (1 = yes, 0 = no)	-229.407	0	0.1	0.3
Water supply	Category dummy variable (1 = yes, 0 = no)	20.32	0/1	0.05	0.21
Climate regulation	Category dummy variable (1 = yes, 0 = no)	595.358	0	0.14	0.35
Soil fertility maintenance	Category dummy variable (1 = yes, 0 = no)	401.655	0/1	0.13	0.34
Waste treatment	Category dummy variable (1 = yes, 0 = no)	163.802	0/1	0.09	0.28
Water flow regulation	Category dummy variable (1 = yes, 0 = no)	106.661	0	0.07	0.26
Genetic diversity	Category dummy variable (1 = yes, 0 = no)	343.905	0/1	0.1	0.3
Recreation	Category dummy variable (1 = yes, 0 = no)	-73.399	0	0.16	0.36
Other services	Category dummy variable (1 = yes, 0 = no)	112.718	0	0.06	0.23
Avoided cost method	Category dummy variable (1 = yes, 0 = no)	-290.201	0/1	0.02	0.14
Replacement cost method	Category dummy variable (1 = yes, 0 = no)	301.408	0/1	0.06	0.24
Travel cost method	Category dummy variable (1 = yes, 0 = no)	4.389	0/1	0.02	0.12
Stated preference method	Category dummy variable (1 = yes, 0 = no)	-62.186	0/1	0.07	0.26
Benefit transfer method	Category dummy variable (1 = yes, 0 = no)	-193.047	0	0.74	0.44
SCI	Category dummy variable (1 = yes, 0 = no)	-233.79	0	0.59	0.49
SCI & SSCI	Category dummy variable (1 = yes, 0 = no)	-263.726	0	0.18	0.38
Non-SCI/SSCI	Category dummy variable (1 = yes, 0 = no)	-143.232	0	0.12	0.32
Year of research	The year when the research was conducted	-3.807	2008.83	2008.83	6.06
GDP per capita	GDP per capita in US dollars at the constant price in 2017	-0.0003	Lognormal	15,763	15,026
Latitude	Continuous, latitude of the study area	5.206	38.7	81.94	58.55
Longitude	Continuous, longitude of the study area	2.189	-113.7	32.51	20.02
Asian studies	1 = if the study is in an Asian country; 0 = otherwise	178.44	0	0.73	0.44
Constant	Constant	7904.185	NA	NA	NA

Liu, S., Liu, J., Young, C.J., Werner, J.M., Wu, Y., Li, Z., Dahal, D., Oeding, J., Schmidt, G., Sohl, T.L., Hawbaker, T.J., Sleeter, B.M. (2012). "Chapter 5: Baseline carbon storage, carbon sequestration, and greenhouse-gas fluxes in terrestrial ecosystems of the western United States". In: *Baseline and Projected Future Carbon Storage and Greenhouse-Gas Fluxes in Ecosystems of the Western United States*. Zhu, Z. and Reed, B.C., eds. USGS Professional Paper 1797.

- Site: Western United States
- Rangeland Landcover Types Applied: Shrubland
- Ecosystem Service(s): Carbon sequestration
- Valuation Methodology: Social Cost
- Sample Size: N/A
- Study Site Annual Household Income: $63,278
- Percent of Population Attaining High School Level Education or Greater: 87%

This study estimated baseline carbon sequestration rates on various ecosystems throughout the western United States. Units published are in Tg of C per square kilometer per year. Values used in this study were the range and average from the Cold and Warm Deserts regions. Units were converted to metric tons of carbon per acre per year and were monetized using the 2020 value for the Social Cost of Carbon (SCC), as developed in the 2021 U.S. Interagency Working Group on Social Cost of Greenhouse Gasses under Executive Order 13990. We applied estimates for shrublands and wetlands to the corresponding landcover types in LRR D across all spatial attributes.

Losey, J. E., & Vaughan, M. (2006). The economic value of ecological services provided by insects. *Bioscience*, 56(4), 311-323.

- Site: United States
- Rangeland Landcover Types Applied: Forest, Shrubland, Grassland
- Ecosystem Service(s): Biological control
- Valuation Methodology: Avoided cost
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

This article focuses on services provided by wild insects inhabiting croplands and rangelands in the United States. The authors estimate avoided cost benefits for several insect-mediated services. In this report, we use the values for livestock pest reduction—such as parasites and pest flies—due to livestock feces decomposition by native insects, which in turn reduces pest habitat. The authors estimate the total losses averted from parasitism and pest flies from this service is 200 million USD each year. This value is divided by the acreage of land in the United States used for cattle ranching and farming to arrive and a dollar per acre per year value.

Lu, X., Kicklighter, D. W., Melillo, J. M., Reilly, J. M., & Xu, L. (2015). Land carbon sequestration within the conterminous United States: Regional- and state-level analyses. *Journal of Geophysical Research: Biogeosciences*, 120(2), 379-398.

- Site: United States
- Rangeland Landcover Types Applied: Grassland
- Ecosystem Service(s): Carbon sequestration
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

This study develops a historical land use and landcover change dataset and combines it with a process-based ecosystem model to estimate carbon sequestration benefits for inland ecosystems in the coterminous United States. Carbon flux is estimated by site around the United States, in grams C per square meter per year. The value for Arizona grasslands is converted to metric tons of C per acre per year and monetized value using the 2020 value for the Social Cost of Carbon (SCC), as developed in the 2021 U.S. Interagency Working Group on Social Cost of Greenhouse Gasses under Executive Order 13990. This value is applied on all grasslands in the study area.

Maczko, K. (2006). *USDA Forest Service rangeland recreation: Site identification, visitor characteristics and activities, and a travel cost model*. Colorado State University.

- Site: Western United States
- Rangeland Landcover Types Applied: Forest, Grassland, Shrubland
- Ecosystem Service(s): Recreation
- Valuation Methodology: Travel Cost
- Sample Size: 1,603
- Study Site Annual Household Income: $63,278
- Percent of Population Attaining High School Level Education or Greater: 87%

This thesis sought to fill gaps about the recreational use and value of rangelands in the western United States by assessing the visitation, demographics, and non-market value of recreation occurring on rangeland recreation sites in the USDA Forest Service (USFS) National Visitor Use Monitoring (NVUM) program. The authors use a travel cost model (R2 of 0.176 and Likelihood Ratio Index of 0.167) to estimate the consumer surplus of recreation, finding an average recreation trip to USDA Forest Service NFS sites is worth $65.68. This value is multiplied by the estimated (using NVUM data) number of trips taken to USFS rangeland recreation sites in a year and divided by the approximate acreage of rangelands to arrive at a dollar per acre per year value.

Maher, A. T., Quintana Ashwell, N. E., Maczko, K. A., Taylor, D. T., Tanaka, J. A., & Reeves, M. C. (2021). *Valuation of beef cattle ecosystem services: An economic valuation of federal and private grazing lands ecosystem services supported by beef cattle ranching in the US. Translational Animal Science.*

- Site: United States
- Rangeland Landcover Types Applied: Grassland,

Shrubland
- Ecosystem Service(s): Recreation, Forage Production
- Valuation Methodology: Market Price
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

The authors use publicly available data to estimate the economic value of ecosystem service associated with beef cattle production in the United States. Values are estimated for federal rangelands, private rangelands, at the state level, and nationally. Ecosystem service values used in this report are those for wildlife-related recreation and forage production for federal rangelands in Nevada, Arizona, New Mexico, and Utah—the states which have the largest area within LRR D. The study uses the U.S. Fish and Wildlife Service estimates of total annual non-market values of recreation divided by the area of non-metro and non-urban land as a per-hectare value. Forage production on federal rangelands is estimated using National Agricultural Statistics Service grazing fees by state. All values were converted to dollar per acre per year for use in this report.

Nowak, D. J., Hirabayashi, S., Bodine, A., & Greenfield, E. (2014). Tree and forest effects on air quality and human health in the United States. *Environmental Pollution*, 193, 119-129.
- Site: United States
- Rangeland Landcover Types Applied: Forest
- Ecosystem Service(s): Air Quality
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

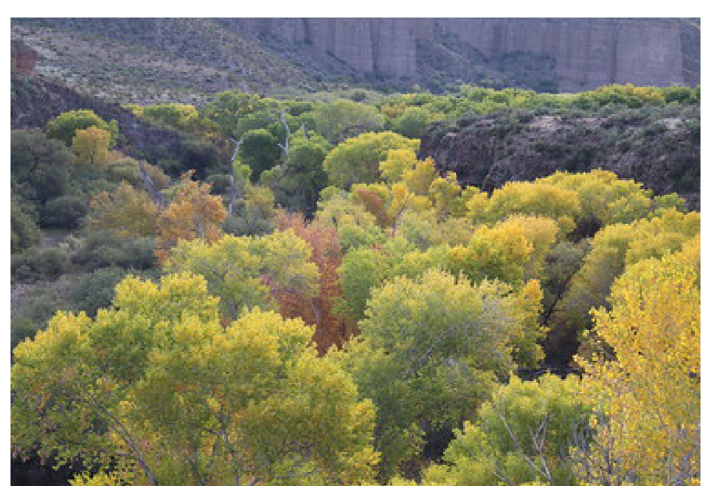
This study models improvements in air quality for forests at the state and county levels in the coterminous United States. Air pollution removal capacity of forests is estimated for NO_2, O_3, $PM_{2.5}$, and SO_2. Monetary air quality benefits are derived from the US EPA's BenMAP program, which calculates avoided costs of adverse health effects such as emergency room visits, hospital admissions from respiratory illness, and more. Values for each pollutant removed are published in terms of 2010 USD per hectare per year, by state, for both urban and rural areas within the United States. We took the range and average in the "all lands" values from Arizona, Nevada, New Mexico, and Utah for this study. We converted the published values using a conversion factor from hectares to acres (2.47105 acres per hectare) to arrive at 2010 USD per acre per year values. We then adjusted converted values to 2021 USD using the World Bank GDP deflator data.

Petrie, M. D., Collins, S. L., Swann, A. M., Ford, P. L., & Litvak, M. E. (2015). Grassland to shrubland state transitions enhance carbon sequestration in the northern Chihuahuan Desert. *Global Change Biology*, 21(3), 1226-1235.
- Site: New Mexico
- Rangeland Landcover Types Applied: Shrubland
- Ecosystem Service(s): Carbon sequestration
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Study Site Annual Household Income: $49,754
- Percent of Population Attaining High School Level Education or Greater: 85.6%

The authors measure surface carbon dioxide flux over 5 years at grassland and shrubland sites in the Sevilleta National Wildlife Refuge in New Mexico. Results were highly variable in desert grassland sequestration and was highly influenced by water availability. Shrubland results were less variable than grassland and less dependent on water availability. The unit of analysis was grams of carbon per square meter per year. Units were converted to metric tons of carbon per acre per year and were monetized using the 2020 value for the Social Cost of Carbon (SCC), as developed in the 2021 U.S. Interagency Working Group on Social Cost of Greenhouse Gasses under Executive Order 13990. We applied the estimates for shrublands from the study to the corresponding landcover types in LRR D across all spatial attributes.

Podolak, K., Edelson, D., Kruse, S., Aylward, B., Zimring, M., & Wobbrock, N. (2015). *Estimating the water supply benefits from forest restoration in the Northern Sierra Nevada. An unpublished report of the nature conservancy prepared with ecosystem economics*. San Francisco, CA.
- Site: California
- Rangeland Landcover Types Applied: Forest
- Ecosystem Service(s): Water supply
- Valuation Methodology: Market Value
- Sample Size: N/A
- Study Site Annual Household Income: $75,235
- Percent of Population Attaining High School Level Education or Greater: 83.3%

This study explored whether increased investment in forest and meadow restoration in the Sierra Nevada mountains could increase and enhance California's water supply. The analysis synthesizes potential water yield impacts from forest thinning from over 150 studies, finding that a three-fold increase in forest restoration could yield up to 6 percent more in mean annual stream flows. Market rates are used to value these benefits. Depending on the watershed, benefits of increased water yield could be as much as $415 million (in 2015 USD). We converted published values (High, Low) by dividing by total published acres to arrive at final 2015 USD per acre per year values. We then adjusted converted values to 2021 USD using the World Bank GDP deflator data.



Rollins, K., Evans, M. D. R., Kobayashi, M., & Castledine, A. (2010). *Willingness to pay estimation when protest beliefs are not separable from the public good definition*. University of Nevada Reno Joint Economics Working Paper No. 10-002.
- Site: Nevada rangelands
- Rangeland Landcover Types Applied: Grassland, Shrubland
- Ecosystem Services: Social
- Valuation Methodology: Contingent valuation
- Sample Size: 2,281
- Study Site Annual Household Income: $60,365
- Percent of Population Attaining High School Level Education or Greater: 86.7

This study sought to determine if protest beliefs affect stated preference willingness-to-pay estimates to prevent ecological losses on Nevada rangelands from wildfire and invasive species (with a response rate of 37%). We interpreted this as an existence value, which we have classified for the purposes of this report under the "social" category (see Table 1). Estimates are constructed for pooled, protest-only, and non-protest samples (model R2 of 0.116). Non-protest respondents tended to have higher willingness to pay than protest respondents. We used estimates from the pooled model, as we cannot determine the population of LRR D that would hold protest or non-protest beliefs.

Monte Carlo simulation and function transfer was used with the study results to estimate more site-specific values for LRR D. Table 6 in the paper presents the model coefficients estimated in the study, and Table 2 presents descriptions for each variable in the model. Mean willingness to pay per household per year is calculated from the pooled Random Effects Probit model using the following formula:

$$-1 \times (\alpha + \sum(\beta_1 X) \div \beta_2)$$

where α is the constant, β_1 is a vector of coefficients for the independent variables, X is a vector of inputs for each independent variable, and β_2 is the coefficient on the bid variable. Willingness to pay is then scaled by the number of households and the total acres of rangeland in LRR D to arrive at a dollar per acre per year value. Dollar values are inflated to 2021 USD using 2005 (the survey year) as the base year.

The table below shows the inputs used in the function transfer simulations. Distributions were defined for the Income and Age variables from data specific to LRR D. Both data are obtained from the U.S. Census Bureau. Lognormal distributions were used for these two variables as it provided the best fit according to the Anderson-Darling and K-S tests. According to Census data, the population of LRR D tended to be younger and slightly wealthier than the sample. The variable OG was set to 0 and cons set to 1. All other variable inputs used normal distributions defined with the mean and standard deviation for each variable provided in Table 2 as well as limits of 0 and 1 (being dummy variables or defined on a scale of 0 to 1). Value was calculated over 10,000 trials. *See table on following page.*

VARIABLE	VARIABLE DESCRIPTION	COEFFICIENT	DISTRIBUTION /INPUT	MEAN	STANDARD DEVIATION
Bid	Dollar amount presented to respondent	-0.104	N/A	65.612	27.153
Income	Household annual income in $1000's	0.03	Lognormal	74.15	41.24
Age	Age of respondent	0.181	Lognormal	40.53	7.63
Age2		-0.002	Age2	N/A	N/A
Yrs_NV	Number of years lived in Nevada	0.034	20.971	20.971	12.001
Job_ag	1 = ranching or agriculture; else = 0	1.007	0.075	0.075	0.264
Job_lndscp	1 = landscaping; else = 0	-4.665	0.02	0.02	0.139
Job_mine	1 = mining; else = 0	-0.956	0.152	0.152	0.36
Job_constr/mfn	1 = construction or manufacturing; else = 0	-0.059	0.097	0.097	0.296
Job_trade	1 = wholesale or retail trade; else = 0	2.722	0.075	0.075	0.264
Job_wtrmgnt	1 = water resources management; else = 0	0.622	0.023	0.023	0.151
Job_othutil	1 = utilities (other than water); else = 0	-2.839	0.034	0.034	0.182
Job_health	1 = healthcare; else = 0	-0.131	0.088	0.088	0.283
Job_nrsci	1 = natural resource / environmental sciences; else = 0	-1.993	0.036	0.036	0.186
Job_ed	1 = education/academia; else = 0	-0.165	0.1	0.1	0.301
Job_ent	1 = arts, entertainment, hotel, food services; else = 0	1.013	0.048	0.048	0.215
Job_recr	1 = outdoor recreation & tourism; else = 0	3.171	0.043	0.043	0.203
Job_publnds	1 = public land management; else = 0	2.325	0.027	0.027	0.162
Job_admin	1 = public admin (not land & water resources); else = 0	0.13	0.016	0.016	0.126
Job_fire	1 = firefighting; else = 0	-0.331	0.02	0.02	0.139
Lrt	1= lives in large rural town; else = 0	-1.229	0.14	0.14	0.347
dmthd_prsgrz	1 = Prescribed grazing not appropriate; else = 0	1.387	0.061	0.061	0.24
Vs	1 = single bid questionnaire version; 0 otherwise	-0.601	0.195	0.195	0.397
Vd	1 = double bid questionnaire version; 0 otherwise	-1.016	0.151	0.151	0.358
OG	Scenario: 1 = restoration (obtain gain); 0 = prevent loss	-0.995	0	0.52	0.5
Info	1 = information sheet provided; 0 = no information sheet	0.024	0.667	0.667	0.472
cons	Constant	-1.074	1	Not reported	Not reported

Schuman, G. E., Janzen, H. H., & Herrick, J. E. (2002). Soil carbon dynamics and potential carbon sequestration by rangelands. *Environmental Pollution*, 116(3), 391-396.

- Site: United States
- Rangeland Landcover Types Applied: Grassland
- Ecosystem Service(s): Carbon sequestration
- Valuation Methodology: Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

The study analyzed carbon sequestration on (grazed) rangelands in the United States. Estimates are provided in Mg C per hectare per year, which were adjusted to metric tons C per acre per year and monetized value using the 2020 value for the Social Cost of Carbon (SCC), as developed in the 2021 U.S. Interagency Working Group on Social Cost of Greenhouse Gasses under Executive Order 13990. We applied this value on all grasslands in the study area.

Smith, J. E. (2006). *Methods for calculating forest ecosystem and harvested carbon with standard estimates for forest types of the United States* (No. 343). United States Department of Agriculture, Forest Service, Northeastern Research Station.

- Site: United States
- Rangeland Landcover Types Applied: Forest
- Ecosystem Service(s): Carbon sequestration
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Average Study Site Annual Household Income: $62,843
- Percent of Population Attaining High School Level Education or Greater: 88.0%

This study seeks to fully account for all carbon stored throughout the lifetime of forests and forest products in the US. The authors identified 10 regions, 51 forest types, and 6 forest ecosystem carbon pools. Two separate tables were developed for afforestation and reforestation. Values are estimated in metric tons of C per acre per year and were monetized using the 2020 value for the Social Cost of Carbon (SCC), as developed in the 2021 U.S. Interagency

Working Group on Social Cost of Greenhouse Gasses under Executive Order 13990. Values used from this study were from the Southern Rocky Mountain region, which covers Wyoming, Colorado, New Mexico, Utah, Nevada, and Arizona. The value was applied to all forests in the study area.

Liu, S., Liu, J., Young, C.J., Werner, J.M., Wu, Y., Li, Z., Dahal, D., Oeding, J., Schmidt, G., Sohl, T.L., Hawbaker, T.J., Sleeter, B.M. 2012. "Chapter 5: Baseline carbon storage, carbon sequestration, and greenhouse-gas fluxes in terrestrial ecosystems of the western United States". In: Baseline and Projected Future Carbon Storage and Greenhouse-Gas Fluxes in Ecosystems of the Western United States. Zhu, Z. and Reed, B.C., eds. USGS Professional Paper 1797.

- Site: Western United States
- Landcover Types: Wetland, Grassland, Shrubland
- Climate Groups: B, C, D
- Spatial Attribute: None
- Ecosystem Services: Climate Stability
- Valuation Methodology: Social Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 87%

Estimated baseline carbon sequestration rates on various ecosystems throughout the western United States. This value was monetized using the 2015 value for the Social Cost of Carbon (SCC), adjusted to 2016$, as developed in Nordhaus 2017 "Revisiting the social cost of carbon" Proceedings of the National Academy of Sciences 201609244.). We applied estimates for grasslands, shrublands, and wetlands to the corresponding landcover types in LRR H across all spatial attributes.

Weltz, M. A., Spaeth, K., Taylor, M. H., Rollins, K., Pierson, F., Jolley, L., ... & Rossi, C. (2014). Cheatgrass invasion and woody species encroachment in the Great Basin: benefits of conservation. *Journal of Soil and Water Conservation*, 69(2), 39A-44A.

- Site: Great Basin (CA, ID, NV, OR, UT)
- Rangeland Landcover Types Applied: Grassland, Shrubland
- Ecosystem Service(s): Fire risk reduction, soil retention
- Valuation Methodology: Avoided Cost
- Sample Size: N/A
- Study Site Annual Household Income: $65,165
- Percent of Population Attaining High School Level Education or Greater: 86.8%

This study compares ecosystem service benefits of natural rangeland vegetation to rangelands containing invasive annual grasses. The alteration of native plant communities can increase the likelihood of damaging natural processes like fires, floods, and erosion. The authors report findings from a study on the wildfire fuel load reduction benefits of native rangelands, finding that wildfire return intervals are longer for native plant communities and wildfire treatments are cost-effective compared to invaded rangelands. In addition, the USDA Rangeland Hydrology and Erosion Model is used to find reduced erosion on natural rangelands compared to invaded rangelands. We value the reduction in tons per acre of soil loss using the weighted average dollar per ton water erosion values for Arizona, Nevada, New Mexico, and Utah from Hansen & Ribaudo (2008)[xvi] to arrive at a dollar per acre value.

[xvi] Hansen, L., & Ribaudo, M. (2008). Economic measures of soil conservation benefits: Regional values for policy assessment. USDA Technical Bulletin, (1922).

8. APPENDIX C

REFERENCES USED TO DETERMINE PRACTICE EFFECTIVENESS

Balliette, J. F., McDaniel, K. C. & Wood, M.K. (1986). Infiltration and sediment production following chemical control of sagebrush in New Mexico. *Journal of Range Management*, 39,160–165.

- Study Location: New Mexico
- MLRA(s): 35, 36
- Health Attribute(s): Hydrologic Function
- Practice(s) Assessed: Brush Management

Bates, J. D., Rhodes, E. C., Davies, K. W., & Sharp, R. (2009). Postfire succession in big sagebrush steppe with livestock grazing. *Rangeland Ecology & Management*, 62(1), 98-110.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Bates, J. D., Miller, R. F., & Svejcar, T. (2007). Long-term vegetation dynamics in a cut western juniper woodland. *Western North American Naturalist*, 67(4), 549-561.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Bates, J. D., Miller, R. F., & Svejcar, T. J. (2000). Understory dynamics in cut and uncut western juniper woodlands. *Rangeland Ecology & Management/Journal of Range Management Archives*, 53(1), 119-126.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity, Hydrologic Function
- Practice(s) Assessed: Brush Management

Bates, J. D., Miller, R. F., & Svejcar, T. (2005). Long-term successional trends following western juniper cutting. *Rangeland Ecology & Management*, 58(5), 533-541.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity, Hydrologic Function
- Practice(s) Assessed: Brush Management

Benz, L. J., Beck, K. G., Whitson, T. D., & Koch, D. W. (1999). Reclaiming Russian knapweed infested rangeland. *Rangeland Ecology & Management/Journal of Range Management Archives*, 52(4), 351-356.

- Study Location: Wyoming
- MLRA(s):32
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Herbaceous Weed Treatment

Bich, B. S., Butler, J. L., & Schmidt, C. A. (1995). Effects of differential livestock use on key plant species and rodent populations within selected Oryzopsis hymenoides/ Hilaria jamesii communities of Glen Canyon National Recreation Area. *The Southwestern Naturalist*, 281-287.

- Study Location: Utah
- MLRA(s): 35
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Prescribed Grazing

Brock, J. H., Blackburn, W. H., & Haas, R. H. (1982). Infiltration and sediment production on a deep hardland range site in North Central Texas. *Rangeland Ecology & Management/Journal of Range Management Archives*, 35(2), 195-198.

- Study Location: Texas
- MLRA(s): 78C
- Health Attribute(s): Hydrologic Function, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Busby, F. E., & Gifford, G. F. (1981). Effects of livestock grazing on infiltration and erosion rates measured on chained and unchained pinyon-juniper sites in southeastern Utah. *Rangeland Ecology & Management/ Journal of Range Management Archives*, 34(5), 400-405.

- Study Location: Utah
- MLRA(s): 35
- Health Attribute(s): Hydrologic Function
- Practice(s) Assessed: Brush Management

Carlson, D. H., Thurow, T. L., Knight, R. W., & Heitschmidt, R. K. (1990). Effect of honey mesquite on the water balance of Texas Rolling Plains rangeland. Rangeland Ecology & Management/Journal of *Range Management Archives*, 43(6), 491-496.

- Study Location: Texas
- MLRA(s): 78A
- Health Attribute(s): Hydrologic Function, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Chambers, J. C., Miller, R. F., Board, D. I., Pyke, D. A., Roundy, B. A., Grace, J. B., ... & Tausch, R. J. (2014). Resilience and resistance of sagebrush ecosystems: implications for state and transition models and management treatments. *Rangeland Ecology & Management*, 67(5), 440-454.

- Study Location: California, Idaho, Oregon, Nevada, Utah, Washington
- MLRA(s): 7, 8, 11, 23, 25, 28A
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Cline, N. L., Roundy, B. A., Pierson, F. B., Kormos, P., & Williams, C. J. (2010). Hydrologic response to mechanical shredding in a juniper woodland. *Rangeland Ecology & Management*, 63(4), 467-477.

- Study Location: Utah
- MLRA(s): 28A
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Davies, K. W., Bates, J. D., Johnson, D. D., & Nafus, A. M. (2009). Influence of mowing Artemisia tridentata ssp. wyomingensis on winter habitat for wildlife. *Environmental Management*, 44, 84-92.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Davies, K. W. (2010). Revegetation of medusahead-invaded sagebrush steppe. *Rangeland Ecology & Management*, 63(5), 564-571.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Herbaceous Weed Treatment

Davies, K.W. & Bates, J.D. (2014). Attempting to restore herbaceous understories in Wyoming big sagebrush communities with mowing and seeding. *Restoration Ecology*, 22(5), 608-615.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Gamougoun, N. D., & Smith, R. D. (1984). Soil, Vegetation, and Hydrologic Responses to Grazing Management at Fort Stanton, New Mexico. *Rangeland Ecology & Management/Journal of Range Management Archives*, 37(6), 538-541.

- Study Location: New Mexico
- MLRA(s): 39
- Health Attribute(s): Biotic Integrity, Hydrologic Function, Soil and Site Stability
- Practice(s) Assessed: Prescribed Grazing

Gibbens, R. P., Herbel, C. H., & Lenz, J. M. (1987). Field-scale tebuthiuron application on brush-infested rangeland. *Weed Technology*, 1(4), 323-327.

- Study Location: New Mexico
- MLRA(s): 42
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Hastings, B. K., Smith, F. M., & Jacobs, B. F. (2003). Rapidly eroding piñon–juniper woodlands in New Mexico: response to slash treatment. *Journal of Environmental Quality*, 32(4), 1290-1298.

- Study Location: New Mexico
- MLRA(s): 36
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Herbel, C. H., Gould, W. L., Leifeste, W. F., & Gibbens, R. P. (1983). Herbicide treatment and vegetation response to treatment of mesquites in southern New Mexico. *Rangeland Ecology & Management/Journal of Range Management Archives*, 36(2), 149-151.

- Study Location: New Mexico
- MLRA(s): 42, 70D, 77D
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Jernigan, M. B., McClaran, M. P., Biedenbender, S. H., & Fehmi, J. S. (2016). Uprooted buffelgrass thatch reduces buffelgrass seedling establishment. *Arid Land Research and Management*, 30(3), 320-329.

- Study Location: Arizona
- MLRA(s): 41
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Herbaceous Weed Treatment

Morton, H. L., Ibarra-F, F. A., Martin-R, M. H., & Cox, J. R. (1990). Creosotebush control and forage production in the Chihuahuan and Sonoran deserts. *Rangeland Ecology & Management/Journal of Range Management Archives*, 43(1), 43-48.

- Study Location: Arizona
- MLRA(s): 41
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Olson, R., Hansen, J., Whitson, T., & Johnson, K. (1994). Tebuthiuron to enhance rangeland diversity. *Rangelands Archives*, 16(5), 197-201.

- Study Location: New Mexico
- MLRA(s): 32, 36
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management, Prescribed Grazing

Pierson, F. B., Bates, J. D., Svejcar, T. J., & Hardegree, S. P. (2007). Runoff and erosion after cutting western juniper. Rangeland Ecology & Management, 60(3), 285-292.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity, Hydrologic Function, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Ross, M. R., Castle, S. C., & Barger, N. N. (2012). Effects of fuels reductions on plant communities and soils in a piñon-juniper woodland. *Journal of Arid Environments*, 79, 84-92.

- Study Location: Utah
- MLRA(s): 35
- Health Attribute(s): Biotic Integrity, Soil and Site Stability
- Practice(s) Assessed: Brush Management

Steers, R. J., & Allen, E. B. (2010). Post-fire control of invasive plants promotes native recovery in a burned desert shrubland. *Restoration Ecology*, 18, 334-343.

- Study Location: California
- MLRA(s): 30
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Herbaceous Weed Treatment

Stonecipher, C. A., Thacker, E., Welch, K. D., Ralphs, M. H., & Monaco, T. A. (2019). Long-term persistence of cool-season grasses planted to suppress broom snakeweed, downy brome, and weedy forbs. *Rangeland Ecology & Management*, 72(2), 266-274.

- Study Location: Utah
- MLRA(s): 28A
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Herbaceous Weed Treatment

Thacker, E., Ralphs, M. H., & Monaco, T. A. (2009). Seeding cool-season grasses to suppress broom snakeweed (Gutierrezia sarothrae), downy brome (Bromus tectorum), and weedy forbs. *Invasive Plant Science and Management*, 2(3), 237-246.

- Study Location: Oregon
- MLRA(s): 23
- Health Attribute(s): Biotic Integrity
- Practice(s) Assessed: Brush Management

Wood, M. K., Garcia, E. L., & Tromble, J. M. (1991). Runoff and erosion following mechanical and chemical control of creosotebush (Larrea tridentata). *Weed Technology*, 5(1), 48-53.

- Study Location: New Mexico
- MLRA(s): 42
- Health Attribute(s): Hydrologic Function, Soil and Site Stability
- Practice(s) Assessed: Brush Management

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Law, J. Y., Brendel, C., Long, L. A., Helmers, M., Kaleita, A., & Soupir, M. (2020). Impact of stacked conservation practices on phosphorus and sediment export at the catchment scale. *Journal of Environmental Quality*, 49(6), 1552–1563. https://doi.org/10.1002/jeq2.20140

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