ACCOUNTING FOR NATURE'S VALUE

WITH USDA-NRCS CONSERVATION PRACTICES IN THE CENTRAL GREAT PLAINS

EARTH ECONOMICS

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1. EXECUTIVE SUMMARY

The term "nature's value" refers to the reality 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 communities. Integrating the economic value of such services into land use planning and resource management could result in more informed decisions about resource allocation and the balance of strategies needed to achieve a range of desired objectives, including those related to agricultural productivity and ecosystem health. Yet today, consideration of a full range of ecosystem service values in conservation planning and policy decision-making is often limited by the lack of comprehensive, rigorous empirical information regarding the economic value of the services provided.

This study focuses on establishing a potential framework for identifying and valuing the ecosystem services derived from conservation actions on rangelands in the Central Great Plains. Due to data limitations, we focused on two conservation practices—Brush Management and Prescribed Grazing—and on a subset of potential ecosystem services, including biological control, soil retention, air quality, and others. This methodology integrates consideration of a broad range of potential benefits of conservation on local communities and economies. It highlights the range of data types, assumptions, and linkages required to produce rigorous ecosystem services valuation estimates in a comprehensive manner.

This study revealed important data gaps and challenges to linking conservation practices on the landscape with improved ecosystem function and increased ecosystem service value. While limitations in data, data granularity, and critical assumptions about the relationships between elements of the framework constrain its precision, the framework and estimates provide a broad sense of the economic importance of NRCS conservation actions.

We developed this potential framework to explore plausible links between ecosystem services and NRCS conservation practices, and to offer NRCS an economic approach to quantify the effects of those practices on the value of non-market ecosystem services. The need to quantify the value of non-market benefits has been recognized in several key pieces of legislation, departmental memos and agency handbooks. The 2019 H.R. 2748, Safeguarding America's Future and Environment Act,[1] the 2015 M-16-01, Incorporating Ecosystem Services into Federal Decision Making,[2] 2014 CEQ Final Interagency Guidelines,[3] and guidance in the 2012 NRCS National Resource Economics Handbook[4] all support the valuation and support of ecosystem services throughout the nation.

Tying practices to ecosystem services to estimate the economic value of conservation practices offers NRCS a more relevant way to communicate conservation successes and accomplishments to the American public, as well as those farmers and ranchers who voluntarily implement conservation practices. NRCS currently reports conservation success in terms of acres-treated or numbers of practices applied, but such metrics rarely show how ecosystem services produce off-site public benefits that are of interest to the public. In addition to reporting "NRCS treated x-acres of invasive plants," this framework—and associated value estimates—allows NRCS to add to reports, "this resulted in improved (or maintained) habitat, water quality, water storage, and other ecosystem services that benefit downstream residents. Voluntary conservation actions by ranchers increased the per acre value provided by nature between $X and $Y dollars." We used peer-reviewed literature and NRCS technical metrics on land health and economic value to develop and test a standardized approach that could be applied to other ecological regions, throughout the country as a means of generating more robust estimates of the benefits supported by NRCS conservation practices, both on and off the ranch.

Ecosystem service valuations could be integrated into conservation planning and policy decision-making in several important ways:

- Improving field-level conservation planning through with more-comprehensive assessments of the potential practice benefits.
- Informing resource allocation into 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 better understanding the 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.

This analysis relies upon available NRCS data, published academic literature, and multiple assumptions about complex functional relationships to bridge gaps in existing research on ecosystem valuation, the impacts of conservation practices, and ecosystem health. Nevertheless, these estimates suggest that rangeland conservation practices—specifically Brush Management and Prescribed Grazing—may significantly improve the ability of rangelands to provide a range 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, as well as those living downstream and in nearby communities. This can lead to better-informed decision making, and support innovative funding mechanisms that ensure that both producers and their neighbors benefit from conservation practices.

Implementing this framework, we estimated that between 2008 and 2016, Brush Management and Prescribed Grazing on private rangelands in Land Resource Region H (LRR H, the Central Great Plains Winter Wheat and Range Region) increased the value of selected ecosystem services by a total between $15 million and $33 million, averaging $1.7 to $3.5 million per year. **That represents an average increase of $2.28 to $4.93 per acre per year of ecosystem services from baseline estimates prior to when those practices were applied.**

The ecosystem services that contributed most to the total value include: air quality (35%); water quality (19%); climate stability (12%); disaster risk reduction (10%); recreation and tourism (7%); water capture, conveyance and supply (7%); soil retention (4%); habitat (3%); and aesthetics (3%).

2. INTRODUCTION

Nature provides a wide range of goods and services that are of value to individuals and communities at the local, regional, and even global scale, including (but not limited to): air and water filtration; food production; natural disaster risk reduction; climate stability and resiliency; cultural and recreational experiences.[5] Collectively, we refer to these outcomes as "ecosystem services," or "nature's value."

In this report, we have categorized ecosystem goods and services following a typology based on the Millennium Ecosystem Assessment (MEA), a common framework in the field of ecological economics:

- Provisioning goods and services provide materials and energy for society, with different ecosystems producing different goods and services. Forests produce lumber, agricultural lands produce food, and rivers supply water for drinking and irrigation.
- Regulating services are benefits produced by biological and chemical processes that moderate natural phenomena. Intact ecosystems limit the spread of disease organisms, maintain water quality, limit soil erosion, and regulate climate.
- Supporting services include the habitat and refugia of living organisms—plants, animal, microorganisms (e.g., rhizobia, actinomycetes), and fungi (e.g., mycorrhizae). These services provide milieus for all life on the planet.
- Information services support meaningful human-nature interactions. This includes spiritually and aesthetically significant natural features, places for outdoor recreation, and opportunities for scientific research and education.

Table 1 provides examples of the 21 ecosystem services identified in this report, showing the array of critical services and benefits provided by nature. Some are traded in markets, and for these, there are mechanisms to assign or impute prices (i.e., economic value). These include traded foods and natural fibers, but also goods and services whose values are implicitly captured within the prices of other traded goods, such as fertile soils, the value of which may be at least partially captured in land prices. However, there are few market mechanisms to communicate the economic value of many other ecosystem benefits—these are known as "non-market" goods and services. We often lack sufficient information to account for the contribution of non-market goods and services; as a result, the benefits of maintaining healthy, functional natural systems is often underrepresented in policy and planning decision-making.

Without functional natural systems, many of the benefits provided by natural systems may need to be replaced by built infrastructure, often at potentially greater cost, due to construction, ongoing maintenance, and eventual replacement costs. Because ecosystems are living, adaptive systems, natural assets may be more resilient and less-costly to maintain than built infrastructure. Acknowledging the economic contribution of natural processes 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.

Agriculture is a unique sector in that it has the potential to degrade natural ecosystems, but also provides many opportunities to design resource management strategies that incorporate ecosystem services into decision-making and land use planning. Since the ravages of the Dust Bowl more than 80 years ago, USDA's Natural Resources Conservation Service (NRCS) has been working with landowners, local and state governments, and other federal agencies to design agricultural land use planning and resource management strategies that address environmental resource concerns and maintain healthy and productive working landscapes. NRCS partners with producers to identify conservation objectives and assess the natural resource opportunities and concerns related to soil, water, animals (including habitat), plants, air, energy and human interaction on private, and in many cases federal, agricultural lands. To address resource concerns, NRCS maintains a suite of conservation practice standards that continue to evolve with research, conservation field trials, and accumulated experience. Such conservation practices are enabled or incentivized by financial and technical assistance through NRCS conservation programs, such as the Environmental Quality Incentives Program (EQIP), to help participants fulfill their conservation plan objectives.

Efficient targeting and implementation of conservation programs, however, requires a comprehensive understanding of the conservation benefits, and ecosystem service improvements, associated with different resource management strategies. However, a full accounting of the ecosystem service impacts of practices and management strategies—and the value of those impacts—is hampered by gaps in applicable research, challenges with data availability and granularity, and an incomplete understanding of biophysical and social interactions at multiple levels. Nevertheless, while it can be challenging to quantify and monetize some factors, their inclusion within the framework 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 service values attributable to conservation practices, this study lays out an estimation framework to link conservation practices, their impacts on resources and ecosystem services, and the assignment of values to those impacts. The precision of these estimates is limited by incomplete data and a lack of applicable valuation studies, but the framework highlights how such data, when available, can be applied, and where improved understanding of critical relationships and interactions is necessary. The estimates derived from this framework can be improved—and made more comprehensive—as such gaps are filled.

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

PROVISIONING SERVICES	
Energy and Raw Materials	Fuel, fiber, fertilizer, minerals, and energy
Food	Livestock, crops, fish, wild game
Medicinal Resources	Traditional medicines, pharmaceuticals, assay organisms
Ornamental Resources	Clothing, jewelry, handicrafts, decoration
Water Storage	Usable surface or ground water, stored reliably
REGULATING SERVICES	
Air Quality	Ability to create and maintain clean, breathable air
Biological Control	Disease, pest and weed control
Climate Stability	Ability to support a stable climate at global and local levels
Disaster Risk Reduction	Ability to prevent or mitigate flood, wildfire, drought, and other natural disasters
Pollination, Seed Dispersal	Dispersal of genetic material via wind, insects, birds, etc.
Soil Formation	Soil creation for agricultural and/or ecosystem integrity
Soil Quality	Soil quality improvement due to decomposition and pollutant removal
Soil Retention	Ability to retain arable land, slope stability, and coastal integrity
Water Quality	Water quality improvement due to decomposition and pollutant removal
Water Supply	Ability to provide natural irrigation, drainage, and other water flows
Navigation	Ability to maintain necessary water depth for recreational and commercial vessels
SUPPORTING SERVICES	
Habitat	Ability to sustain species and maintain genetic and biological diversity
CULTURAL SERVICES	
Aesthetic Information	Sensory enjoyment and appreciation of natural features
Cultural Value	Use of nature in art, symbols, architecture, or for religious or spiritual purposes
Science and Education	Use of natural systems for education and scientific research
Recreation and Tourism	Hiking, boating, travel, camping, and more

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

FIGURE 1. BOUNDARY AND LOCATION OF LRR H WITHIN THE U.S. AND MLRA BOUNDARIES WITHIN LRR H

3. THE FRAMEWORK

This framework was developed to estimate the changes in ecosystem service value associated with grazed rangeland and rangeland management practices in what NRCS identifies as the Central Great Plains, or, more specifically, Land Resource Region H (LRR H, see Figure 1).[i] LRR H is the Central Great Plains Winter Wheat and Range Region, located in the south-central part of the Great Plains.[7] It spans 219,740 square miles (slightly over 140.6 million acres) across Texas, Oklahoma, Nebraska, New Mexico, Kansas, and Colorado, and includes all or part of 302 counties. The terrain is relatively flat, with average annual temperatures ranging from 54 to 60 degrees Fahrenheit, and an average of 20 to 29 inches of annual rainfall. Grasslands and cultivated fields are the most common ecosystem types.

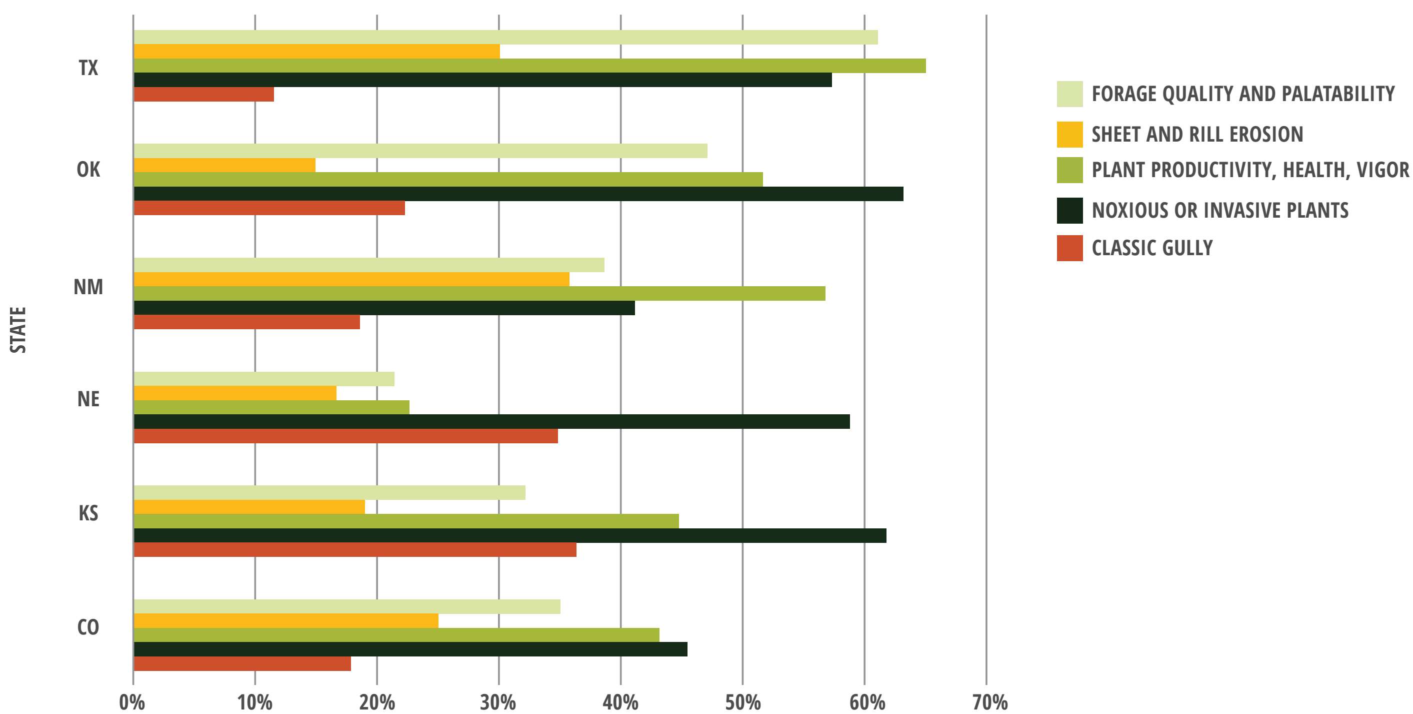
The NRCS, through the National Resources Inventory (NRI), collects and produces scientifically credible information on the status, condition, and trends of land, soil, water, and related resources on the nation's non-federal lands in support of efforts to protect, restore, and enhance the lands and waters of the United States. The 2004-2014 NRI Grazing Land Onsite data study identified the most significant resource concerns on non-federal rangelands in LRR H: noxious or invasive plants; declining plant productivity, health and vigor; forage quality and palatability concerns; sheet and rill erosion; and non-stabilized classic gullies. Figure 2 shows the percent of non-federal rangeland acres affected by the most prevalent rangeland resource concerns in each state that intersects with LRR H.

About 99 percent of the land in LRR H is privately owned, with 92 percent of the land in private cropland or grassland.[8] Agriculture is thus a major source of employment within the region (Table 2). According to the USDA's 2017 Census of Agriculture, the seven states of LRR H have over 500,000 farm and ranch operations.[9] These operations support more than 330,000 full-time farm workers. Beef cattle production is the dominant agricultural enterprise on non-cultivated lands within the region.

[i] 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."8 A MLRA is described as having similar topography, geology, climate, water, soil, biological resources, and land use within its boundary. There are 17 MLRAs within LRR H.

FIGURE 2. PERCENT OF NON-FEDERAL RANGELAND ACRES AFFECTED BY RESOURCE CONCERNS, BY STATE (NOT LIMITED TO LRR H)

Source: NRCS 2004-2014 NRI Grazing Land Onsite Data Study.



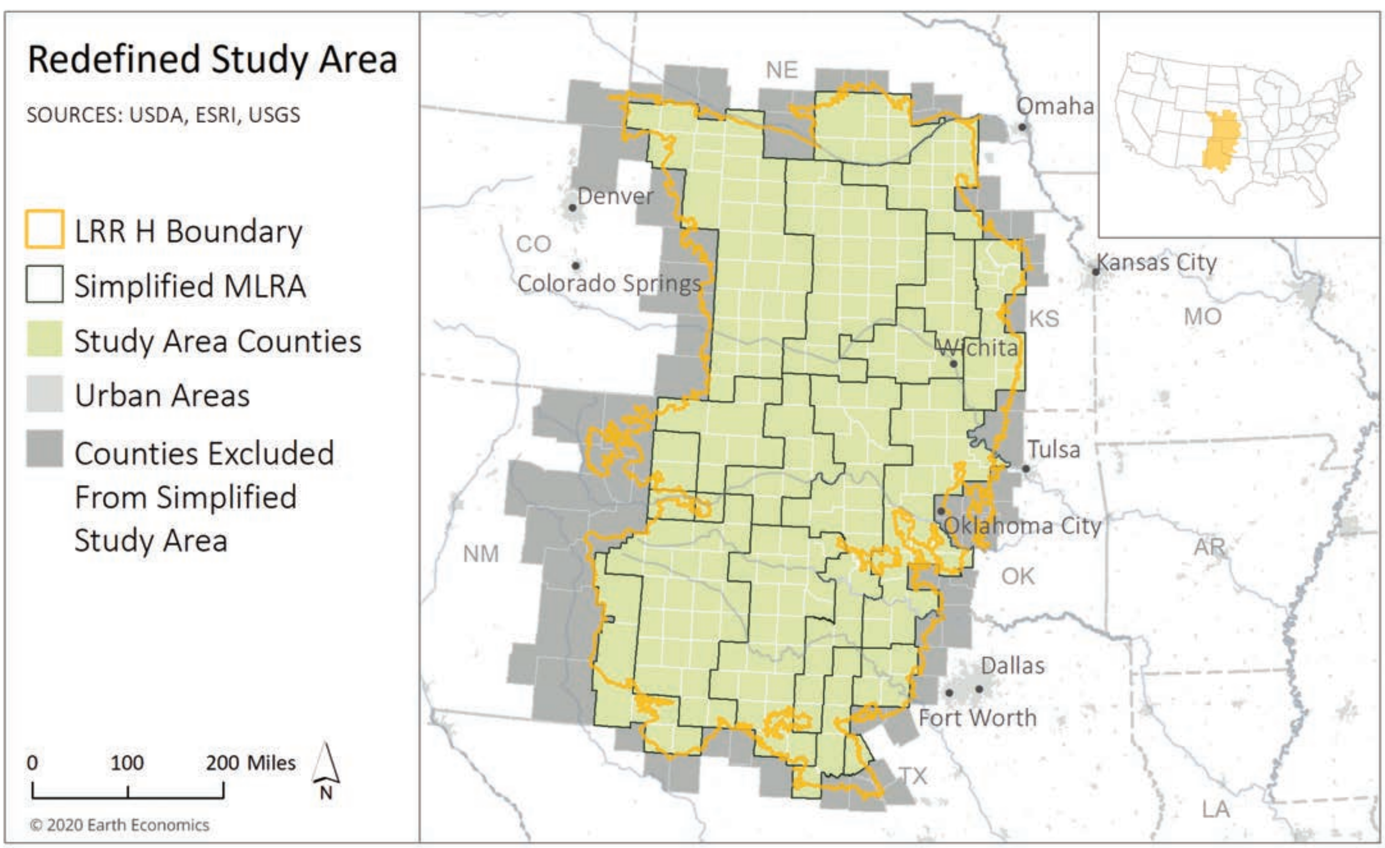
This analysis explores the relationship between NRCS conservation practices applied on rangelands in select counties of LRR H (Figure 3) and measures of ecosystem health and value at the county level. Although NRCS conservation practice data are collected at the field level, such data was aggregated to the county level before being made publicly available for analysis, to ensure landowner confidentiality. The working definition of rangeland and its extent within the redefined study area (Figure 3) is explained in section 3.2.1. Counties with less than half their territory within LRR H have been excluded. Figure 3 shows the original LRR H boundary in yellow, and the study area counties for this report in green. Our use of the term "study area" refers to the redefined boundary of LRR H based on county boundaries.

TABLE 2. PERCENT EMPLOYMENT BY INDUSTRY IN LRR H

INDUSTRY	PERCENT EMPLOYED
Educational services, and health care and social assistance	22.9
Agriculture, forestry, fishing and hunting, and mining	15.5
Retail trade	10.2
Manufacturing	8.4
Construction	7.2
Arts, entertainment, and recreation, and accommodation and foodservices	6.3
Transportation and warehousing, and utilities	5.9
Public administration	5.4
Other services, except public administration	5.0
Professional, scientific, and management, and administrative and waste management services	4.9
Finance and insurance, and real estate and rental and leasing	4.3
Wholesale trade	2.8
Information	1.3

Source: U.S. Census Bureau, 2017 American Community Survey 1-Year Estimates.

FIGURE 3. REDEFINED STUDY AREA



3.1 OVERVIEW

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

1. Determine baseline ecosystem health attributes and baseline value estimates of the ecosystem services provided on rangeland within the study area;
2. Estimate the magnitude of change in ecosystem function associated with the implementation of specific rangeland conservation practices;
3. Quantify any change in nonmarket ecosystem values attributable to the implementation of conservation practices.

This following section provides a brief description of that framework. Subsequent sections elaborate on each step, illustrating how they can be applied to quantify the impacts of NRCS conservation practices on the value of the ecosystem services. The framework adapts work by the U.S. Forest Service and others,[10,11,12] which scales ecosystem service valuation estimates from the literature by local ecosystem health indices to determine the value of ecosystem services provided by natural lands at various stages of degradation, by including elements similar to site data collected by NRCS.

The analysis begins by identifying landcover[ii] characteristics within the study area, and then deriving a baseline measure of the "health" of rangeland ecosystems. Rangeland health has been assumed to be represented by three attributes, as documented in *Interpreting Indicators of Rangeland Health:*[13]

- **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 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 attributes described above.

Benefit Transfer Methods (BTM) are then used to estimate the baseline economic value of ecosystem services produced on rangelands (of various types) within the study area, given the calculated level of rangeland health for each of those rangeland areas. BTM is broadly defined as the use of existing data or information in settings other than for which it was originally collected.[14] The process is similar to home appraisals in which the recent sale value and features of comparable, neighboring homes (e.g., two bedrooms, garage, recently remodeled) are used to estimate the value of an off-market home. As a means of indirectly estimating the value of ecological goods or services,[15] 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.[16]

The second step in the framework involves evaluating the impacts of applied conservation practices on the derived indexes of rangeland health. NRCS data on the applied practices are used to identify the area of impact of conservation practices. A literature review then informs estimates of the impacts of specific NRCS practices on rangeland health attributes. Due to limitations on the availability of relevant literature, this analysis was restricted to impacts of two NRCS rangeland management practices: Brush Management and Prescribed Grazing.

Studies are then selected to estimate the proportional change in rangeland health attributes (percent change per year) associated with each practice. These proportional change estimates are then adjusted to reflect two factors that could alter practice effectiveness: the length of time since practice implementation; and the major climatic variables precipitation

FIGURE 4. GENERAL STEPS IN THE INTEGRATED ECONOMIC FRAMEWORK

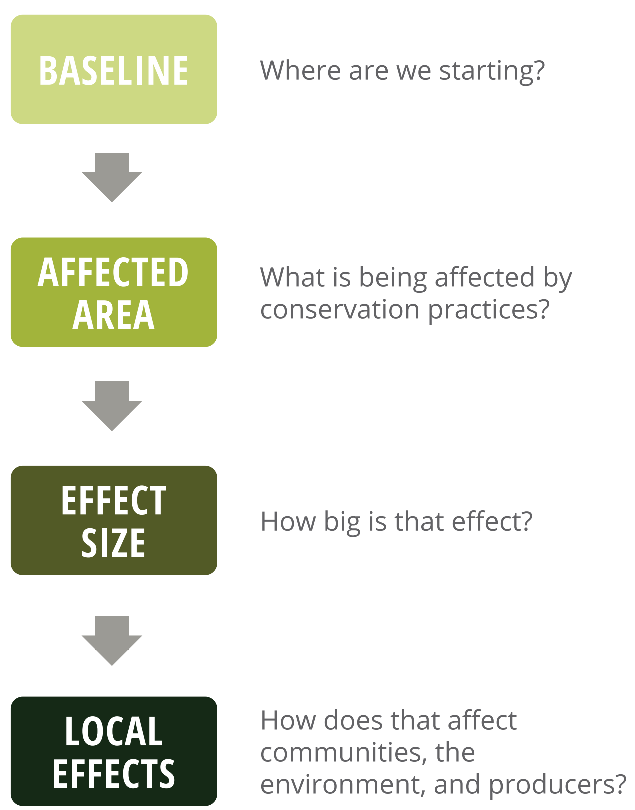
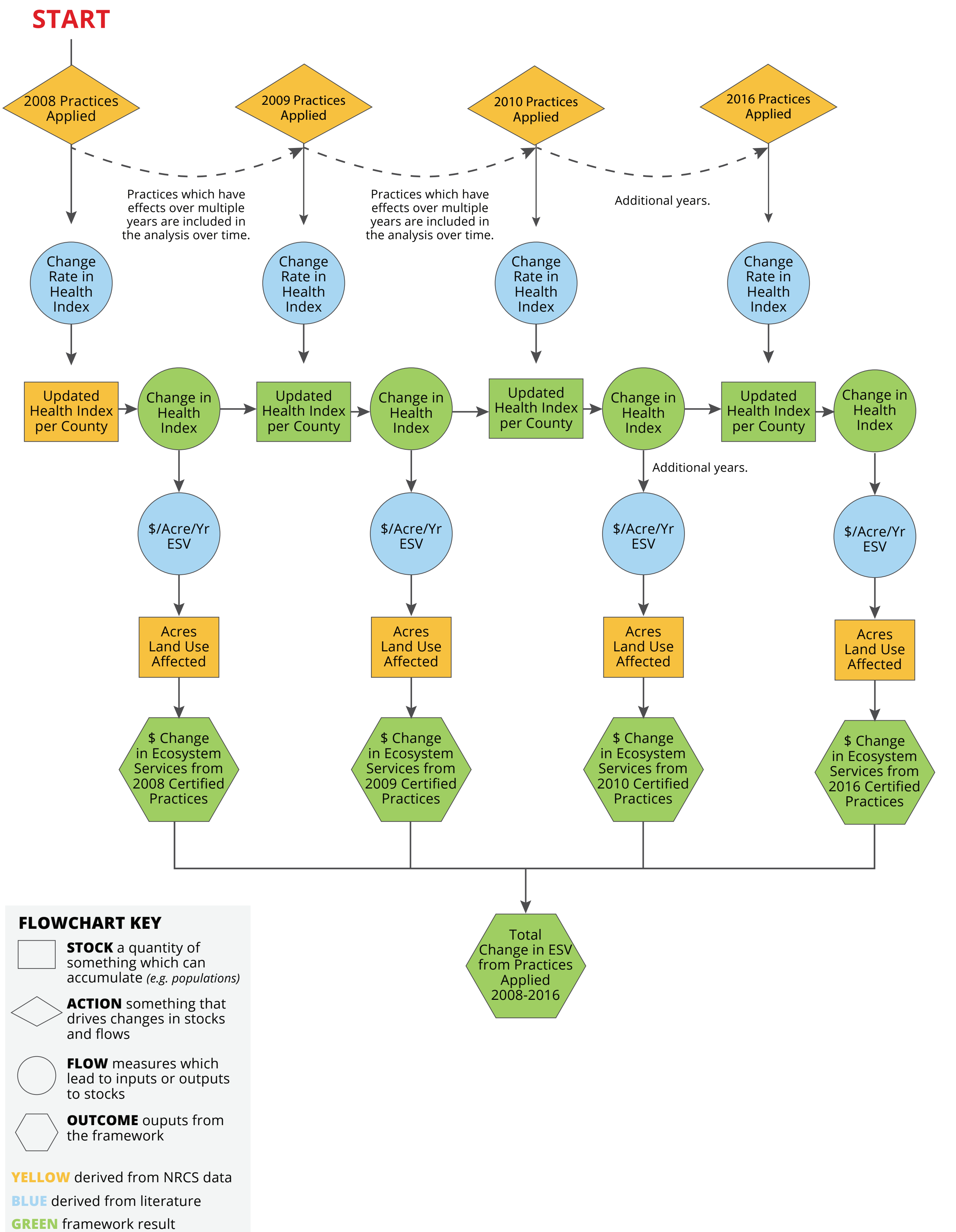


FIGURE 5. EXPANDED INTEGRATED ECONOMIC FRAMEWORK



and temperature. This adjusted proportional change rate is 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 this analysis have been calculated based on 2004-2008 data. Annual impacts of conservation practices on rangeland health and the ecosystem service values associated with those health improvements are then calculated sequentially, as conservation practices were applied each year from 2008 to 2016. Regional totals for the value of changes in ecosystem services were calculated by summing changes in value across each rangeland type within the study area, over the full period of analysis.

The full framework is depicted in Figure 5. The generalized approach outlined in Figure 4 has been applied for each year of NRCS-certified contracts (2008-2016), with the benefits associated with each subsequent year based on conservation practices implemented in prior years. These steps are detailed in the following sections.

As with any attempt to estimate ecosystem service values and land health trajectories, the effectiveness of this approach depends upon sufficient site data and related literature, including—but not limited to—primary studies and related 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. 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.

The following section describes the framework we developed to integrate ecosystem service valuation with the NRCS conservation planning process.

3.2 BASELINE ANALYSIS

The scope of this report is limited to non-federal rangelands in the redefined LRR H, and a subset of NRCS conservation practices applied to those rangelands from 2008-2016. To assess changes in ecosystem service provisioning over time, we first determined the baseline condition of non-federal rangeland in the study area prior to implementation of NRCS conservation practices in the period of analysis. The following sections describe the estimation of baseline levels of rangeland health and the economic value of ecosystem service provisioning. These baselines are the reference conditions to estimate changes in ecosystem function (section 3.2.2) and ecosystem service provisioning (section 3.2.4) associated with implementation of NRCS conservation practices on non-federal rangelands in LRR H between 2008 and 2016.

3.2.1 CHARACTERIZING TYPES OF RANGELAND IN THE STUDY AREA

By using the definition of rangeland, provided by NRCS, it is important to note that for the sake of this study rangeland consists of multiple landcover types. In this analysis, we defined rangelands in terms of landcover types in the National Land Cover Database 2011 (NLCD),[18] which maps landcover at a 30-meter spatial resolution (approximately 0.22 acres per pixel). Table 3 lists the landcover types defined in the NLCD.

The study area covers nearly 140 million acres. More than ninety percent of this area is cropland, grassland, or shrubland, with about four percent categorized as developed. Forest, barren land, pasture, water, and wetlands combined make up less than five percent of the total area. Figure 6 shows a map of where these landcovers occur in the study area, and Table 4 shows the proportion of each landcover type relative to the total area.

Because rangeland encompasses multiple landcover types, we interpreted rangeland as non-urban grassland, shrubland, and wetlands—both in and out of riparian zones—in any climate.[iii]

Landcover categories each provide a different pattern of ecosystem goods and services. Rangeland landcover was further differentiated by spatial attributes relevant to the provision of ecosystem services: location within a riparian area (i.e., freshwater floodplain); proximity to urban areas, as defined by the 2010 US Census Urban Areas dataset;[19] and prevailing temperatures and precipitation patterns, per Köppen-Geiger climate classifications.[22] Such attributes reflect site characteristics that can improve the accuracy of Benefit Transfer of ecosystem service value estimates.[14] For instance, because riparian areas provide distinct ecosystem services, landcover within the study area were differentiated accordingly.[iv, 20, 21]

Climate zones strongly determine regional ecologies and climate-related stresses. Certain ecosystem services may be rarer—and potentially more valuable—in some climates than others. For example, water may be valued more in arid regions than in wet, humid climates. The Köppen-Geiger framework defines five macro-climate groups by seasonal precipitation,

TABLE 3. LAND COVER DEFINITIONS, AS USED IN THE NATIONAL LAND COVER DATABASE (2011).

LANDCOVER	DESCRIPTION
Open Water	Areas of open water, generally with less than 25% cover of vegetation or soil.
Developed	Highly developed areas or areas with any mixture of constructed materials and vegetation.
Barren Land	Areas of bedrock, desert pavement, scarps, talus, slides, volcanic material, glacial debris, sand dunes, strip mines, gravel pits and other accumulations of earthen material. Generally, vegetation accounts for less than 15% of total cover.
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.
Pasture/Hay	Areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops, typically on a perennial cycle. Pasture/hay vegetation accounts for greater than 20% of total vegetation.
Cultivated Crops	Areas used for the production of annual crops, such as corn, soybeans, vegetables, tobacco, and cotton, and also perennial woody crops such as orchards and vineyards. Crop vegetation accounts for greater than 20% of total vegetation. This class also includes all land being actively tilled.
Wetlands	Vegetated areas where the soil or substrate is periodically saturated with or covered with water.

Source: U.S. Geological Survey. 2014. NLCD 2011 Land Cover (2011 Edition, amended 2014) - National Geospatial Data Asset (NGDA) Land Use Land Cover. Sioux Falls, SD

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."[17]

temperature patterns, and vegetation type.[v] Three of these are currently present in the study area: B (Arid/Dry), C (Temperate), and D (Continental/Cold) (Figure 7).[22]

3.2.1.1 CHARACTERIZING RANGELAND BY COUNTY

While we estimated the baseline economic value provided by rangeland ecosystems at the pixel level, matching the resolution of NRCS conservation practice data (aggregated to the county level to ensure landowner confidentiality), required us to characterize average rangeland types at the county level.

TABLE 4. LANDCOVER EXTENTS IN THE STUDY AREA

LANDCOVER	PERCENT OF STUDY AREA
Cropland	41.90%
Grassland	36.20%
Shrubland	13.10%
Developed	4.30%
Forest	2.30%
Pasture	0.70%
Water	0.60%
Wetland	0.60%
Barren Land	0.20%

Thus, we interpreted each acre of rangeland as the proportion of landcover type and attribute combinations present in each county. For example (see Table 5), one acre of rangeland in Andrews County is assumed to include 0.458 acres of arid, non-riparian, non-urban grassland, 0.538 acres of arid, non-riparian, non-urban shrubland, etc.

TABLE 5. EXAMPLE CONVERSION FACTORS FROM LAND USE TO LANDCOVER FOR ANDREWS COUNTY, TEXAS

LAND USE: RANGELAND				
LANDCOVER	CLIMATE TYPE	RIPARIAN ZONE	URBAN ZONE	CONVERSION FACTOR
Grassland	B	No	No	45.8%
Shrubland	B	No	No	53.8%
Shrubland	B	Yes	No	0.1%
Wetland	B	No	No	0.2%

iii Working with NRCS rangeland specialists, we associated rangeland land use with the grassland, shrubland, and wetland landcover types defined in NLCD and restricted rangeland to pixels in non-urban areas.

iv Riparian areas are delimited by the floodplains of surface streams.[20] This study uses the USGS National Hydrography Dataset (NHD)21 to define these riparian areas as a 100-ft buffer around NHD Area and Waterbody features.

v The classification used is the Köppen-Geiger Climate Classification.[22]

FIGURE 6. LANDCOVER TYPES IN THE STUDY AREA

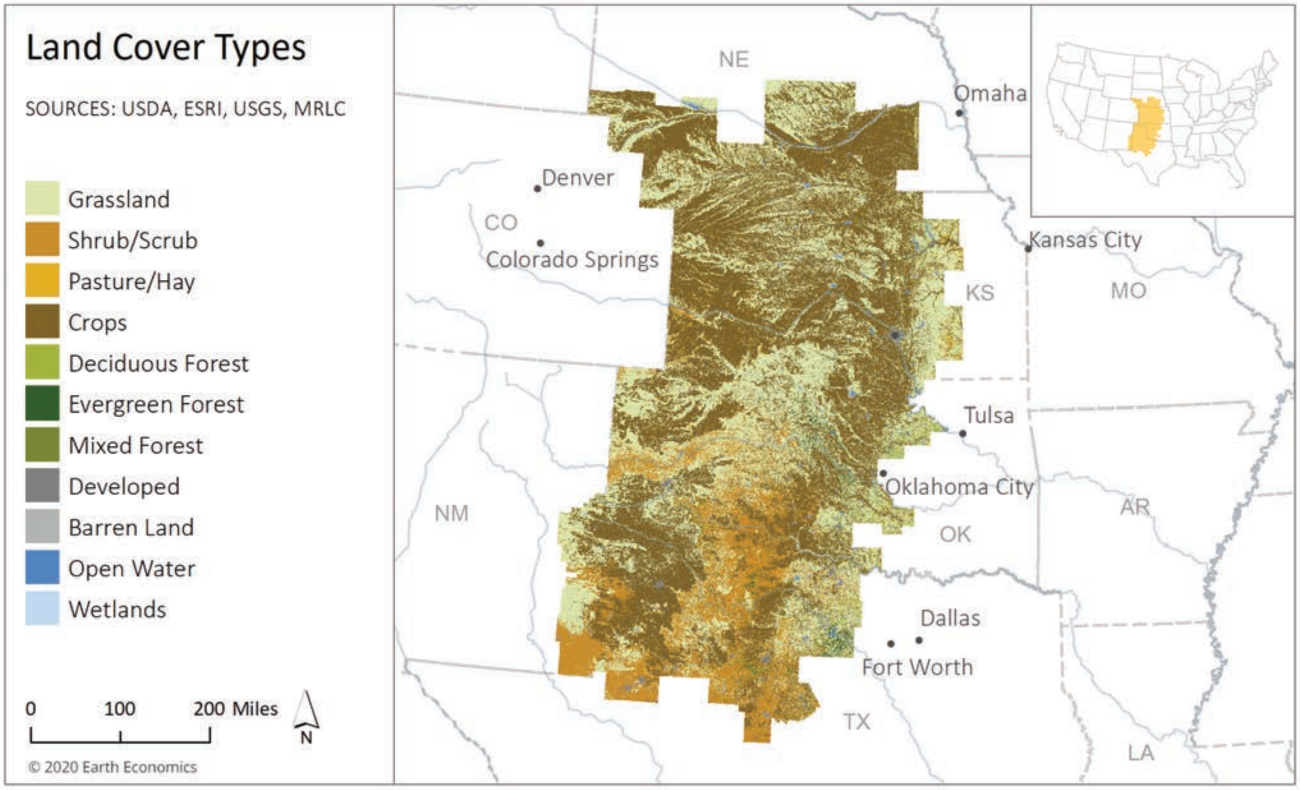
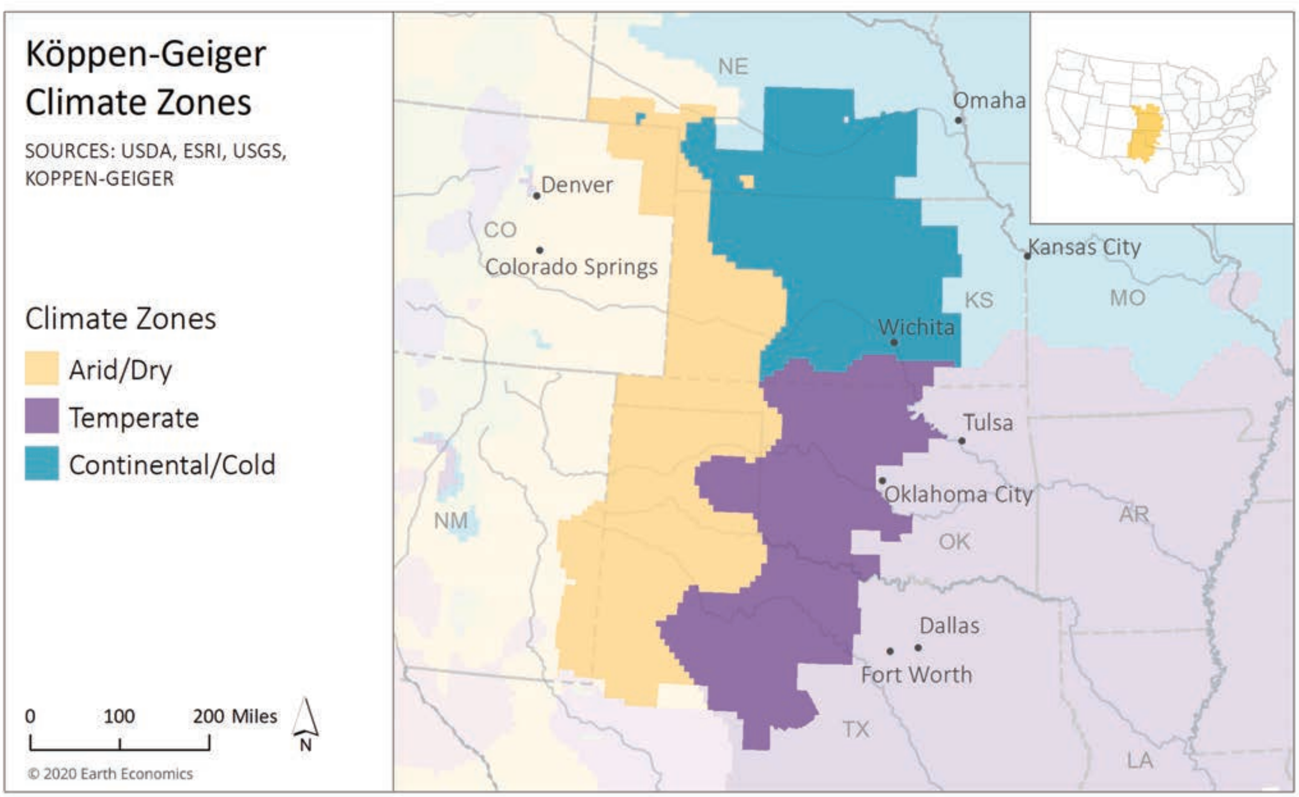


FIGURE 7. KÖPPEN-GEIGER CLIMATE ZONES IN THE STUDY AREA



3.2.1.1 CHARACTERIZING RANGELAND BY COUNTY

While we estimated the baseline economic value provided by rangeland ecosystems at the pixel level, matching the resolution of NRCS conservation practice data (aggregated to the county level to ensure landowner confidentiality), required us to characterize average rangeland types at the county level. Thus, we interpreted each acre of rangeland as the proportion of landcover type and attribute combinations present in each county. For example (see Table 5), one acre of rangeland in Andrews County is assumed to include 0.458 acres of arid, non-riparian, non-urban grassland, 0.538 acres of arid, non-riparian, non-urban shrubland, etc.

3.2.2 CALCULATING A MEASURE OF BASELINE RANGELAND HEALTH

It is common for primary valuation studies to not record detailed site conditions, be associated (or assumed to be associated) with ecosystems at or near fully functional, "pristine" conditions, or measure values at hypothetical pristine conditions. Following previous work, we make the assumption that monetary estimates in the research literature (see section 3.2.3) should be discounted to reflect baseline conditions using a proxy index of ecosystem health.[10,11,12] In the event that ecosystem health is degraded, we expect ecosystem function to decline. This approach avoids overestimating the contribution of ecosystem services from degraded ecosystems. 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 2004-2008, the period immediately prior to conservation practice implementation (2008-2016).

Direct measures of the three attributes of rangeland health—**soil and site stability, hydrologic function,** and **biotic integrity**—are difficult to determine directly, due to the complexity of the underlying processes. Instead, biological and physical characteristics were used as indicators of overall functionality. The baseline status of the three attributes of rangeland health was based on seventeen indicators from NRCS' National Resources Inventory (NRI) *Grazing Land Onsite Data Study, Rangeland Health Assessment Protocol.*[23,24] The NRI is a statistical survey of natural resource conditions and trends on non-federal land within the United States.[vi] Rangeland health was based on NRI's assessment of the degree of departure from reference conditions, as determined by the Ecological Site Descriptions developed by the NRCS for all seventeen indicators:[vii]

- rills
- water flow patterns
- pedestals and/or terracettes
- bare ground
- gullies
- wind-scoured, blowout, and/or depositional areas
- litter movement
- soil surface resistance to erosion
- soil surface loss or degradation
- compaction layer
- plant community composition and distribution relative to infiltration and runoff
- litter amount
- functional/structural groups
- plant mortality/decadence
- annual production
- invasive plants
- reproductive capability of perennial plants.

The NRI data characterize indicators by departure categories, which describe the degree of departure from the reference conditions of each ecological site.[vi] The degree to which indicators depart from expected conditions are characterized as: 1 (none-to-slight); 2 (slight-to-moderate); 3 (moderate); 4 (moderate-to-extreme); or 5 (extreme-to-total). Thus, as departure increases, the 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.

We then combined baseline NRI data for the indicators into rangeland health indices as follows:[24]

1. Because the NRI indicator data were collected at multiple sites over a four-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 county.

2. To normalize the departure scores for each attribute at each point, the median attribute value of the point 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 upper and lower bounds for each health index were subsequently reviewed by NRCS rangeland specialists.

These indices reflect the relative position of each ecological site along a continuum of departure from reference conditions, allowing us to make general statements like, "An index value of 0.80 is closer to the full potential of the ecological site than a value of 0.20," or, "The higher the index value, the healthier the site."

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vi Non-federal land includes privately-owned lands, tribal and trust lands, and lands controlled by state and local governments.

vii Reference conditions are determined by Ecological Site Descriptions developed by the NRCS. 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 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.

Because the conservation practice analysis was conducted at the county scale, attribute indices for each county need to be similarly aggregated to produce average county scores, weighted by the acres represented by each NRI measurement.[viii] We then took the average index score across all rangeland health attributes to represent the overall baseline health of rangelands in each county, meaning each attribute contributes and 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,[13] none describe a site's 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 county scale as a proxy for average rangeland health across broader areas. Moreover, the index score values themselves are of less interest than the relative change in indices attributable to application of conservation practices.

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

DEPARTURE CATEGORY	NRI SCORE	HEALTH INDEX SCORE*
None to Slight	1	1.00
Slight to Moderate	2	0.80
Moderate	3	0.60
Moderate to Extreme	4	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 actually 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.*

3.2.2.1. BASELINE HEALTH INDEX RESULTS FOR THE STUDY AREA

County-aggregated rangeland health indices tended to score above 0.5, with Biotic Integrity exhibiting the greatest variation across counties. Overall, county-level scores for soil and site stability were high, with many counties scoring near 1 (i.e., a none-to-slight departure from reference conditions). County-level scores for hydrologic function were also high but tended to be lower than those for soil and site stability. Figure 8 shows the county-level rangeland health index score for each attribute, while Figure 9 shows the distribution of attribute index scores for all counties in the study area.

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

FIGURE 8. MEDIAN RANGELAND HEALTH ATTRIBUTE INDEX SCORES FROM BASELINE (2004-2008) NRI DATA

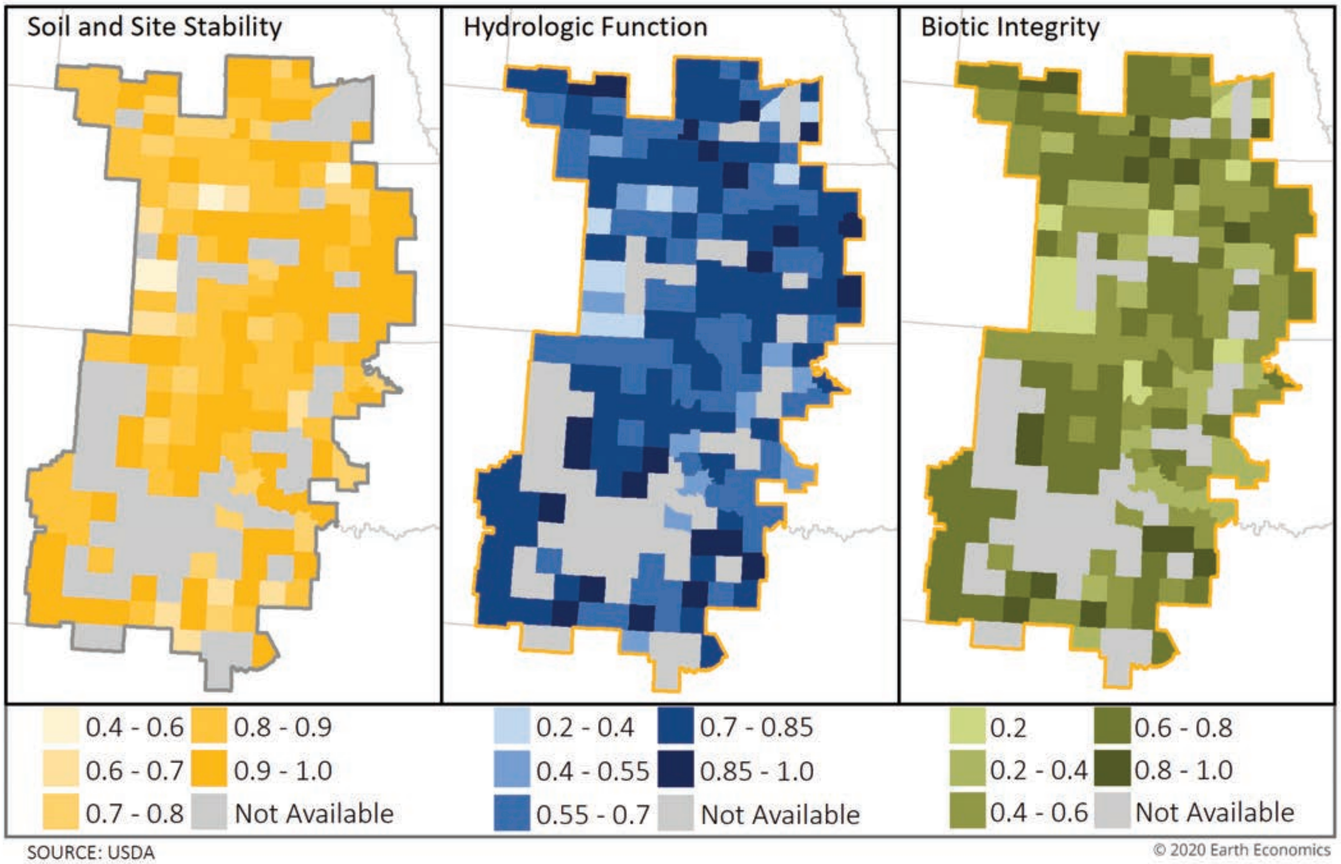
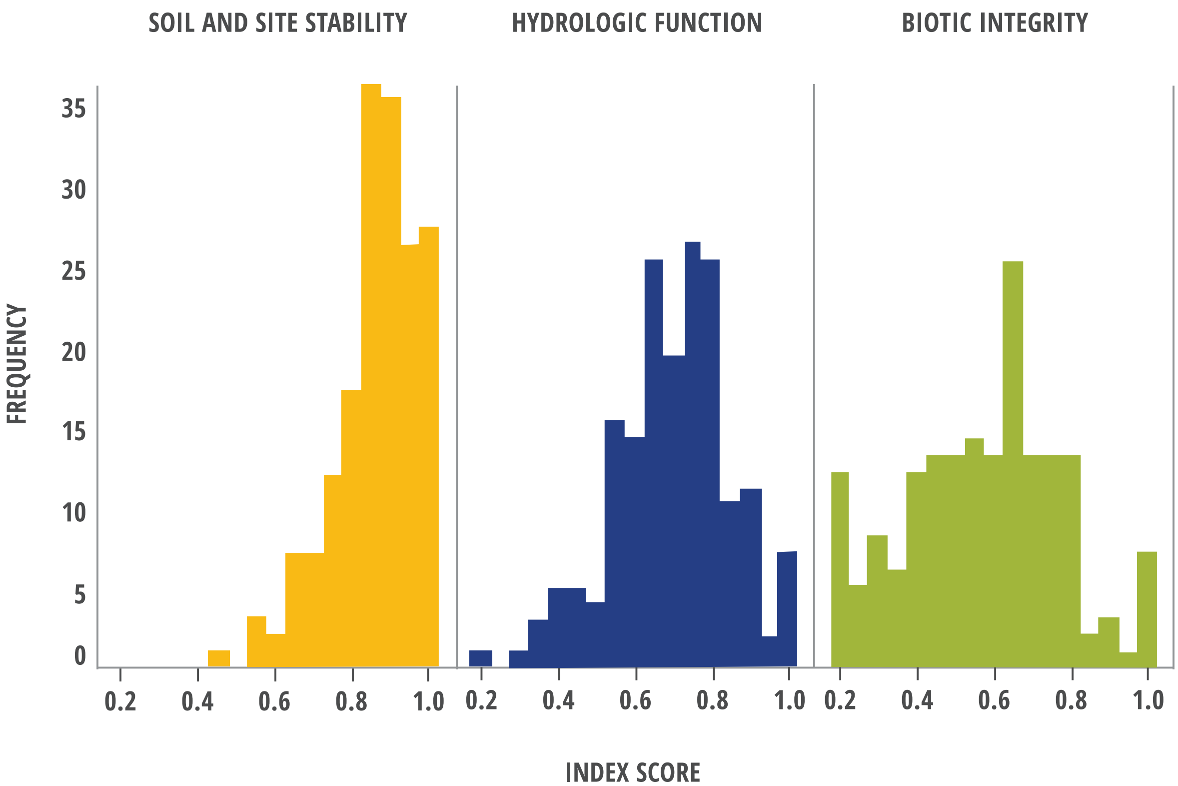


FIGURE 9. VARIATION IN COUNTY BASELINE HEALTH ATTRIBUTE INDEX SCORES FOR RANGELAND IN THE STUDY



3.2.3 NON-MARKET BENEFIT VALUATION METHODOLOGY

We used Benefit Transfer Methods (BTM) to identify published estimates for the economic value of ecosystem services by ecosystem types, transferring these estimates to comparable ecosystems within the study region.[14] As a secondary research method, BTM results can be somewhat imprecise, but when applied transparently, with conservative criteria for selecting eligible primary studies, BTM can generate reasonable estimates that may be sufficient to inform decision making. As with all research, such estimates may be improved over time, as more research and data become available.

3.2.3.1 IDENTIFYING STUDIES FOR USE IN BTM

The BTM process begins by identifying primary studies with similar landcover classifications (e.g., wetland, forest, grassland) and attributes (e.g., climate, proximity to riparian or urban areas) as those found within the study area, as defined in section 3.2.1. 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. These studies were then reviewed again to ensure that values have been selected based on commensurate site attributes[25] and best-available methodologies (Appendix B).

The secondary review used a double-review process—as recommended by best-practices literature—in which two analysts independently identified, reviewed, and coded studies for inclusion into the valuation dataset.[26] Individual primary value estimates were then standardized by units of measure and inflation into 2016 U.S. dollars per acre per year.

The criteria for evaluating values within studies for inclusion in the analysis have been summarized below. Values failing to meet these criteria were excluded from the valuation dataset. Appendix B lists the studies selected for inclusion in the analysis.

SIMILARITY OF ECOSYSTEM GOODS AND SERVICES

At the most basic level, the ecosystem service valued at both study and transfer sites should be similar, as the similarity of uses, goods, and services at both study and transfer sites is critical for valid transfers.[25,27,28,29] During the review process, the ecosystem services in the primary studies were identified. If those services could not plausibly be provided by rangelands within the study area, the value was excluded.

SIMILARITY OF LANDCOVER TYPES

As with ecosystem goods and services , the landcover types at both the study and transfer sites must be similar, as errors diminish as similarities increase.[15,25,30] The ecosystems central to this study are described in the NLCD framework outlined in section 3.2.1. If a reported landcover did not fit into this framework, the study was excluded.

CREDIBLE AND APPROPRIATE METHODOLOGY

Selected studies must be from credible sources, using high-quality data and applying accepted economic valuation methods.[25,29,31,32] 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 preference. Table 7 provides descriptions of the most common valuation techniques.

The economics literature provides guidance on which valuation methods are best-suited to specific ecosystem services. For

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 absence 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.

example, when valuing food provisioning, direct market pricing (revealed preferences) are considered to be more reliable than stated preference approaches. Table 8 lists each ecosystem service and the most appropriate valuation methodologies as identified in, or inferred from, the literature.[33] When available, primary studies have been prioritized for dataset inclusion over those using secondary methods. However, where gaps were found, secondary valuation studies have been used, albeit with a preference for meta-analyses, which provide generalized estimates across multiple study sites.

Furthermore, the double-review process included an assessment of study methodology. Studies were reviewed for indicators such as pre-tested survey language, survey question formats, adequate sample sizes and response rates, data collection methods, sample representation, application of statistical tests to models and data, treatment of outlying information, explanatory power of models, and other factors. If any of these showed weakness in the reported methods, such studies have not been included in the dataset.

STUDY LOCATION

We limited the selection of valuation studies to those conducted in the United States. Studies conducted within the study area were evaluated first. Where these were deemed acceptable, other studies valuing the same ecosystem services but conducted elsewhere in the United States were excluded. Studies based elsewhere in the continental United States were included on a case-by-case basis, depending on transferability (Table 8) and other key criteria. 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 transferable (Table 8). Again, value estimates of ecosystem services conducted within the study were prioritized. Estimates for services of similar U.S. ecosystems located outside of the study area were assessed for relevance on a case-by-case basis, provided those services were considered to have at least medium transferability.

STUDY SITE DEMOGRAPHICS

Benefit transfers tend to be more accurate when demographics, social attitudes, and consumer beliefs at the transfer and study sites are similar.[29,34,35] Unfortunately, few such sociocultural characteristics were reported in the valuation literature identified for this study, apart from easily-obtainable data such as income. Limiting study location can help to partly address the effects of cultural attitudes and beliefs. We recorded the median household income and level of education for each primary study site and compared these to the median and average values of the population within study area, as reported in U.S. Census data. We recognize that populations living outside of the study area may value the grazing lands within it, but those levels of interest were not found in the research identified for this study, so we limited analysis to the population living within the study area. However, we chose to not adjust ecosystem service values by population or income for a few reasons. First, we include value estimates from multiple valuation methodologies, some of which did not consider income in their methodologies. Second, small differences in socio-demographic indicators are unlikely to have a significant impact on value estimates, and the significance of these variables is mixed in valuation models.[36,37,38] Third, the primary study sites included in the dataset tended to have incomes within ten percent of the median household income of the study area.

PUBLICATION YEAR

All things being equal, value transfers will be more accurate if the time between the original publication year and the present is small.[38,39] Accordingly, we omitted literature published prior to 1990. We also prioritized more recent studies for inclusion into the dataset. The oldest study selected was in 1991, but the average publication year of all studies included in the data set was 2005.

TABLE 8. TRANSFERABILITY AND VALUATION METHODS FOR ECOSYSTEM SERVICES

ECOSYSTEM SERVICE	MOST APPROPRIATE VALUATION METHOD	TRANSFERABILITY ACROSS SITES
Aesthetic Information	H, CV, TC, CA	Low
Air Quality	CV, AC, RC	High
Biological Control	AC, P	High
Climate Stability	CV, M, AC, RC	High
Cultural Value	CV, CA	Low
Disaster Risk Reduction	AC	Medium
Energy & Raw Materials	M, P	High
Food	M, P	High
Habitat and Nursery	CV, P, AC, TC	Low
Medicinal Resources	M, AC	Low
Navigation	M, CV	High
Ornamental Resources	AC, RC, H	Medium
Pollination and Seed Dispersal	M, P	Medium
Recreation and Tourism	TC, CV, CA	Low-Medium
Science and Education	CA	High
Soil Formation	AC, CV, RC, P	Medium
Soil Quality	RC, AC, CV	Medium
Soil Retention	AC, RC, H	Medium
Water Capture, Conveyance, and Supply	M, AC, RC, H, P, CV, TC	Medium
Water Quality	RC, AC, CV	Medium
Water Storage	M, AC, RC, P, CV	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, S., R. Costanza, D.L. Childers, et al. 2006. Linking ecology and economics for ecosystem management. Bioscience 56: 121–133. Ecosystem service categories not present in Farber et al. (2006) were included and entries assigned based on similarity to the original classification presented.

3.2.3.2 ASSIGNING MONETARY VALUES TO ECOSYSTEM SERVICES

Using these criteria, the most appropriate ecosystem service values for each landcover-ecosystem service were selected for the valuation dataset. Both maximum and minimum estimates were used as the final values, as reporting a range of values for each combination underscores the variability in the location, methods, and socioeconomic characteristics of the primary studies. The unit of measure for this analysis is dollars-per-acre-per-year ($/acre/year). All values were adjusted to 2016 U.S. dollars using the World Bank GDP inflation and deflation factors. The unit values for all ecosystem services were then summed for each associated landcover type to calculate the total annual value produced by each acre of each landcover. These totals were then adjusted by the baseline ecosystem health index scores, then scaled by their extent within the study area to estimate the total baseline value of ecosystem services across the study area (section 3.2.4).

A total of 77 value estimates from 28 studies on grassland, shrubland, and wetland ecosystem services were included in the dataset. These studies allowed us to estimate the annual economic value of 12 of 21 ecosystem services. Three services were valued on shrubland, and nine services were valued on both grassland and wetland. Table 11 summarizes the general combinations of landcover and ecosystem services that could be valued based on the literature meeting the inclusion criteria. Highlighted combinations represent combinations valued in the benefit transfer dataset. Gaps vary across spatial attributes for each land-cover type, as well. Again, although this dataset represents the best-available approximation of ecosystem service valuation estimates in the study area, it can be extended and improved as new primary analyses and better data become available.

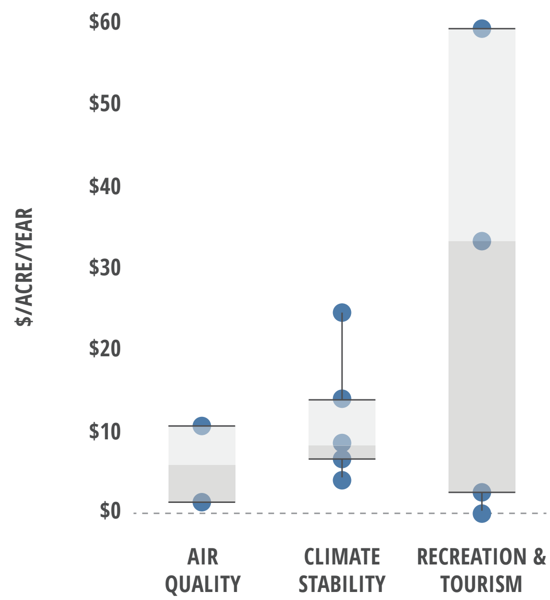
TABLE 9. ECOSYSTEM SERVICE AND GENERAL LANDCOVER COMBINATIONS VALUED IN THE STUDY AREA

ECOSYSTEM SERVICES VALUED IN THIS STUDY	GRASSLAND	SHRUBLAND	WETLAND
Aesthetic Information	•		
Air Quality	•	•	
Biological Control	•		
Climate Stability	•	•	•
Disaster Risk Reduction	•		•
Habitat	•		•
Recreation & Tourism	•	•	•
Soil Retention	•		•
Soil Quality			•
Water Capture, Conveyance, & Supply			•
Water Quality	•		•
Water Storage	•		•

• Value estimates included in the ESV dataset

FIGURE 10. DISTRIBUTION OF ECOSYSTEM SERVICE VALUES (ESV) IN THE DATASET FOR SHRUBLAND

Boxes indicate the middle quartiles of each distribution. Whiskers identify the high and low estimates. Dots represent each value estimate included. Variation is displayed across all spatial attributes and climates for a given landcover type.



That a specific combination of landcover and associated attributes and ecosystem service value has not been included here does not necessarily mean such ecosystems do not produce a given service—or that the service is not valuable—but rather reflects a lack of peer-reviewed data relevant to that combination. For example, shrubland is known to provide valuable services (e.g., recreation, habitat, carbon sequestration), yet there are few valuation studies of this landcover type. Thus, caution should be exercised when comparing total ecosystem service values across landcover types, as differences in values may reflect information gaps, rather than real differences in ecosystem productivity or the value of such services. Ongoing investment in primary valuations is 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.

The following three figures (Figure 10, Figure 11, Figure 12) plot the distribution of ecosystem service value (ESV) estimates included in the dataset, across the relevant landcover types. Boxes indicate the middle quartiles of each distribution (25-50 percent), while whiskers signify high and low estimates. Dots represent each value estimate included. These data reflect variation across all spatial attributes and climates for a given landcover type.

Figure 10 shows the ESV estimates for shrubland. Only three services were valued on shrubland, all less than $60/acre/year. The values for recreation showed the greatest variability, mostly due to the valuation methodologies used in each estimate.

Figure 11 shows the ESV distribution for grasslands. Almost all of these estimates were below $300/acre/year, with the exception of disaster risk reduction, soil retention, and water quality. The high values in these categories were estimated

FIGURE 11. DISTRIBUTION OF ECOSYSTEM SERVICE VALUES (ESV) IN DATASET FOR GRASSLAND

Boxes indicate the middle quartiles of each distribution. Whiskers identify the high and low estimates. Dots represent each value estimate included. Variation is displayed across all spatial attributes and climates for a given landcover type.

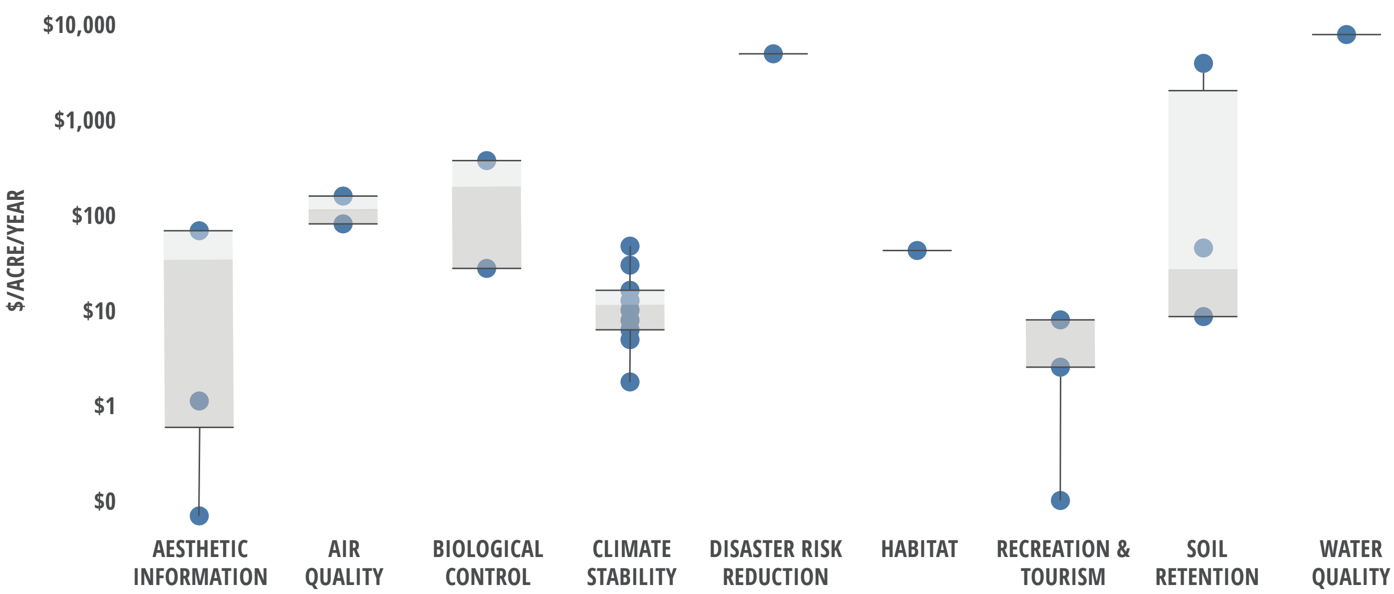
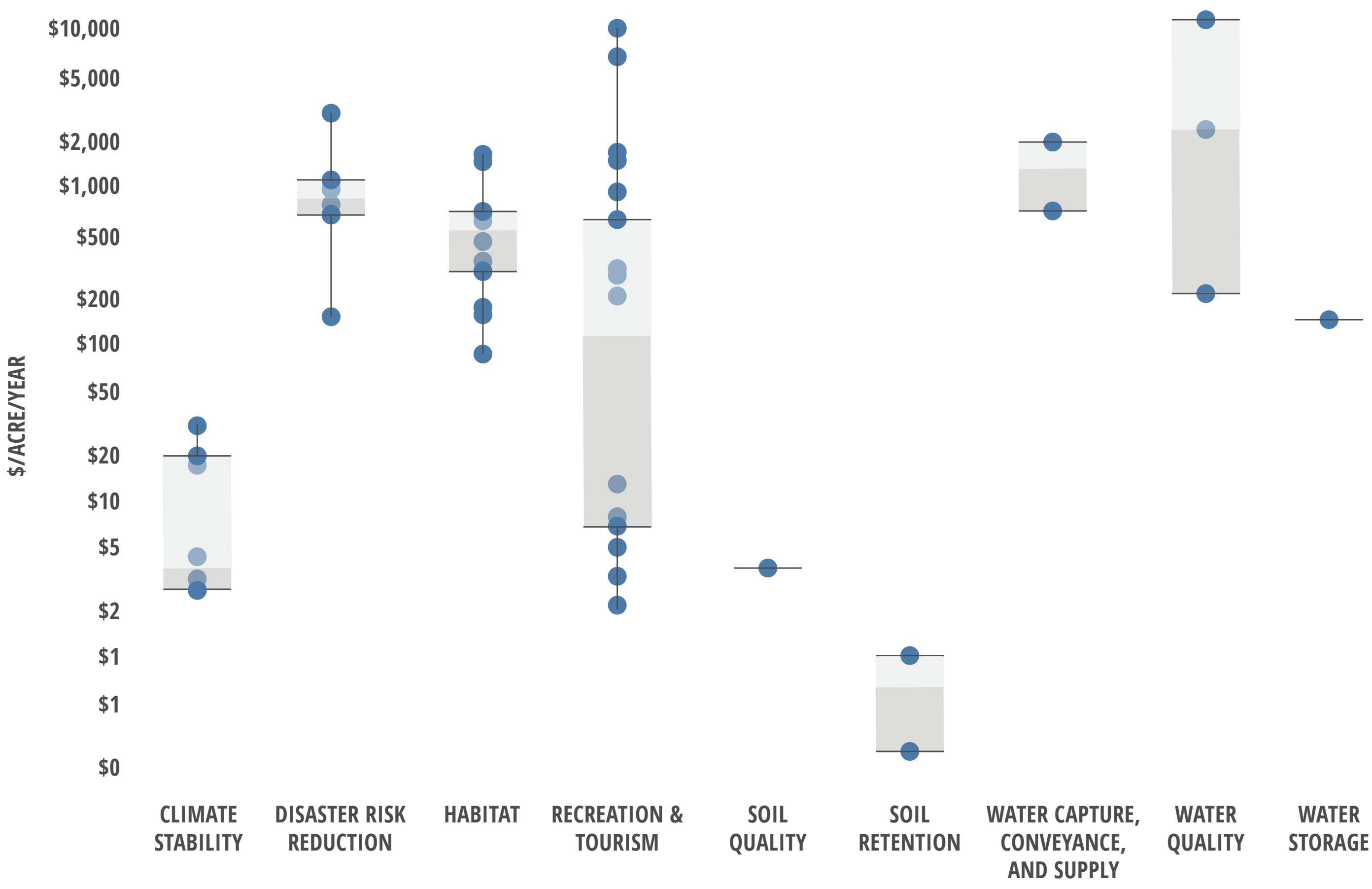


FIGURE 12. DISTRIBUTION OF ECOSYSTEM SERVICE VALUES (ESV) IN DATASET FOR WETLANDS

Boxes indicate the middle quartiles of each distribution. Whiskers identify the high and low estimates. Dots represent each value estimate included. Variation is displayed across all spatial attributes and climates for a given landcover type.



for riparian grasslands, and as such were only applied to rangelands that were identified as riparian.

Figure 12 shows the wetland ESV estimates included in the dataset. These varied more widely across climate and spatial attributes than grassland or shrubland value estimates. Due to lack of primary data, many of the value estimates used for wetlands included meta-analyses that calculated generalized estimates for the value of ecosystem services. Value estimates for riparian wetlands and wetlands in arid climates tended to be much higher than non-riparian wetlands.

3.2.4 CALCULATING BASELINE ECOSYSTEM SERVICE BENEFITS

As we explained in section 3.2.2, we assumed that the valuation literature represents fully-functioning (i.e. at or near reference condition) ecosystems, which should be discounted based on the relevant level of rangeland health. Once ecosystem service unit value estimates have been identified, these should be discounted by baseline rangeland health attribute indices to approximate the baseline ecosystem function and the associated ecosystem service value.[10,11,12]

Following earlier work, 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.[10,11,12] 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 suitable way could be found to introduce genuine precision—selecting other response curves would be equally arbitrary. At a minimum, the linear assumption adopted here provides a straightforward and consistent substitute. This approach also represents a conservative means of adjusting ecosystem service values to site-specific conditions—where primary valuations were based on less than fully-functioning ecosystems, the value of ecosystem services produced has been undervalued.

In this way, we adjust ecosystem service values by the average county-level health attribute score for each landcover type present in each county. For example, a county with an average rangeland health index of 0.5 would be credited with half the ecosystem service value it might have with fully healthy rangeland ecosystems. These dollar-per-acre values are then scaled by the acreage of the associated landcover-attribute combination within each county. The total economic value per landcover-attribute combination (summed from all valued ecosystem services of that combination) are then summed across all landcover types in each county to produce a total ecosystem service value per county.

Total ecosystem service values were calculated as follows in Equation 1 (Table 10 provides an example):

$$(1)\ ESV_j = \Sigma_{m,n}\, Acres_{n,j} \times D_{m,n}$$

Where:

ESV_j	total baseline ecosystem services ($/year) produced in county *j*
$Acres_{n,j}$	the number of acres of landcover-attribute combination *n* in county *j*
BH_j	the weighted average of the median health attribute index scores in county *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 the current level of condition. These values may be viewed as conservative estimates, because it was not possible to value every ecosystem service, nor even all identified ecosystem services on every landcover-attribute combination—the contributions presented here are only partial estimates.

3.2.4.1 RESULTS

The approach used in this study estimates that the baseline ecosystem service value provided by rangelands in the study area, aggregated over all landcover-attribute combinations that we assigned to rangeland (section 3.2.1), ranges from $3 billion to $7 billion each year. Figure 13 shows the distribution of the average baseline ecosystem service values (in millions of $/year) provided by rangeland for each county in the study area. Table 11 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.

TABLE 10. EXAMPLE CALCULATION OF BASELINE ECOSYSTEM SERVICE VALUE FOR LOGAN COUNTY, CO

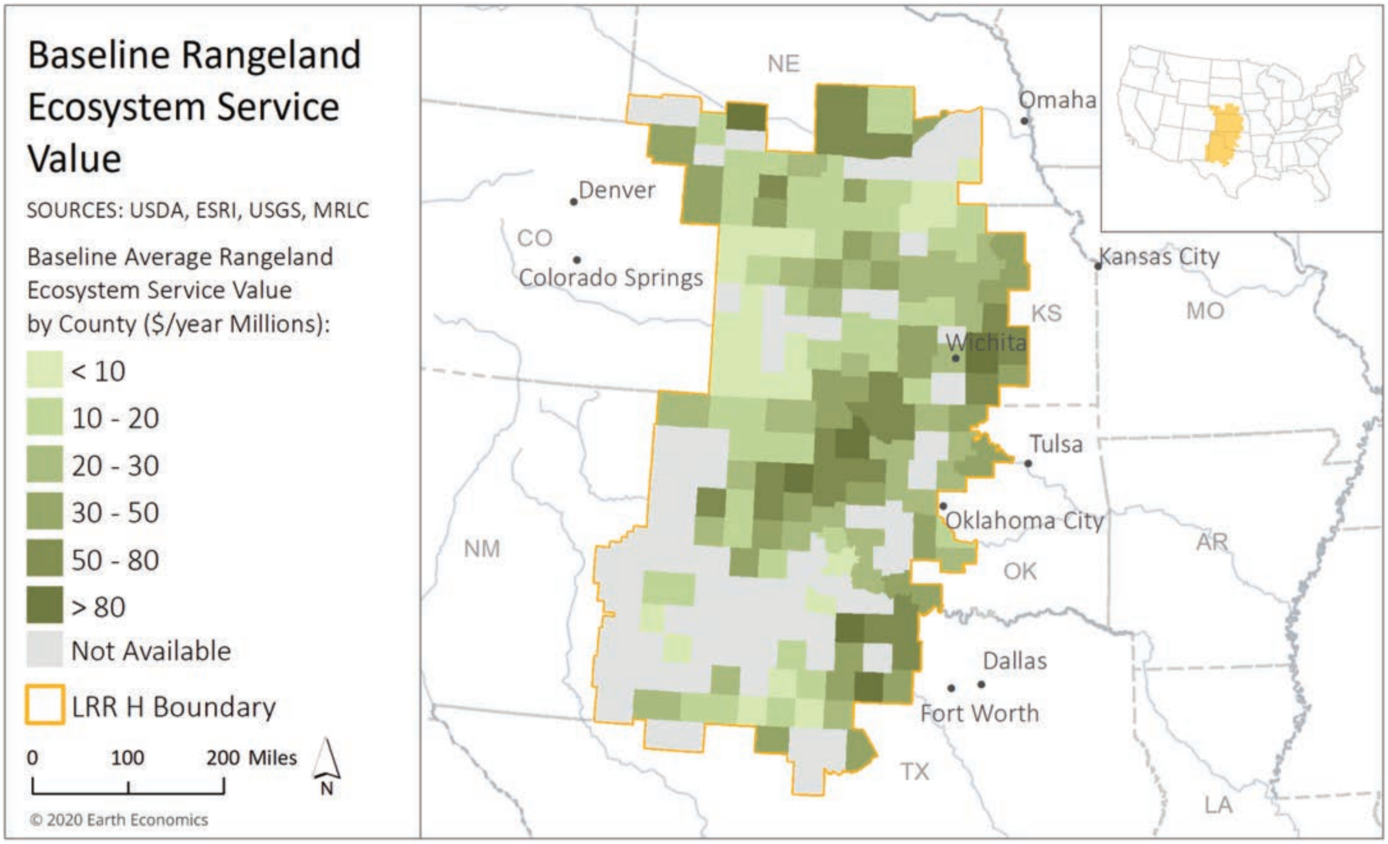
LANDCOVER	ACRES	AVERAGE HEALTH INDEX SCORE	$/ACRE/YEAR		$/YEAR	
			LOW	HIGH	LOW	HIGH
Grassland (Arid, Riparian)	4,680	0.65	$1.74	$69.51	$5,293	$211,449
Grassland (Arid, Non-Riparian)	579,402	0.65	$1.74	$69.51	$655,304	$26,179,251
Wetland (Arid, Non-Riparian)	12,433	0.65	$693.09	$711.00	$5,601,172	$5,745,911
Etc.						
			GRAND TOTAL FOR LOGAN COUNTY*		$10,608,340	$66,128,415

**Columns do not sum as not every landcover combination is shown in the table*

TABLE 11. BASELINE ANNUAL MONETIZED RANGELAND ECOSYSTEM SERVICE VALUE FOR STUDY AREA COUNTIES

LANDCOVER	$/YEAR (MILLIONS)		AVERAGE $/ACRE/YEAR	
	LOW	HIGH	LOW	HIGH
Grassland	$2,260	$4,827	$54	$115
Shrubland	$78	$438	$7	$40
Wetland	$643	$1,665	$925	$2,396
TOTAL	$2,981	$6,930	$55	$129

FIGURE 13. DISTRIBUTION OF AVERAGE BASELINE ECOSYSTEM SERVICE VALUE IN THE STUDY AREA



3.3 CALCULATING THE EFFECTS OF CONSERVATION PRACTICES

NRCS issues payments to producers when a conservation practice in a conservation plan and contract has been applied and meets the specific design requirements. From 2008-2016, NRCS issued 9,500 payments for the Brush Management conservation practice, meaning that Brush Management was applied 9,500 times over nine years, or 1,055 times per year on average. Also from 2008-2016, Prescribed Grazing was applied 4,100 times in 2008-2016 (receiving that many NRCS payments), an average of 456 times each year. Of all practices certified on rangeland in the study area during 2008-2016, Brush Management and Prescribed Grazing were the two most-implemented practices in the study region by both count (applications) and acres treated, representing 14 percent and 6 percent of all practice applications, respectively. Because practices could potentially be applied repeatedly on the same rangelands, we were unable to determine the total *unique* acres where either practice was applied. The NRCS National Planning and Agreement Database (NPAD) does not consistently distinguish whether acres treated in a given contract have been treated previously. Brush Management is a practice that often requires retreatment to achieve reduction targets for undesirable woody plants. The same could be said of Prescribed Grazing—it may need to implemented repeatedly on the same rangelands to achieve desired objective(s), so the same acres may receive NRCS cost-share for multiple years on the same contract. Because of this ambiguity in NRCS NPAD data (and because cost-share data had been aggregated to the county level), it was impossible to distinguish whether NRCS cost-sharing supported retreatment or expanded treatment to other rangelands within the same contract.

Without finer resolution in the NRCS practice application 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 identical change in ecosystem function with each repeat application; there is a law of diminishing returns. To avoid double counting, we counted implementation acres once per practice per contract, recognizing that this approach may under-value the subsequent effects on ecosystem services. Using the average landcover description of land use within each county, we derived the acreage of each landcover affected by a given practice (section 3.2.1, Table 5, for an example), limiting the analysis to those landcover-attribute combinations within the overarching "rangelands" category.

TABLE 12. MEDIAN ANNUAL PERCENT CHANGE RATES IN HEALTH INDEX SCORE, BY PRACTICE

HEALTH ATTRIBUTE	BRUSH MANAGEMENT (314)	PRESCRIBED GRAZING (528)
Soil and Site Stability	2.31	2.46
Hydrologic Function	0.03	3.79
Biotic Integrity	25.63	10.55

We then reviewed the research literature to attempt to quantify the impacts of specific conservation practices on rangeland health attributes. Each study was evaluated by a NRCS CEAP-Grazing Lands sub-team for methodological quality and relevance to the research scope. This resulted in sixteen studies relevant to all three rangeland health attributes for two practices: Brush Management (314), and Prescribed Grazing (528)[ix] (Appendix C). From these, we identified 52 proportional relationships between specific health indicators and specific conservation practices, and translated each into percent changes per year. Such changes could be either positive (e.g., health improvements from reducing bare ground) or negative (worsening health due to increased erosion).

For example, Cassels et al. conducted a study comparing rotational to continuous grazing in tallgrass prairie in Oklahoma (1989-1993), that recorded higher standing crops in the rotational grazing system. We linked these results to Prescribed Grazing practices and the rangeland health biotic integrity attribute. The study showed the standing crop increased by 12.5 percent during the 4-year study, an annualized change of +3.13 percent. We similarly derived relationships from research on the effects of Brush Management practices.

Figure 14 shows a histogram of change rates found in selected literature that were applicable to the study area (additional details can be found in Appendix C). The percent changes reported in the literature are shown on the horizontal axis, and the number of studies reporting those changes on the vertical axis. We adopted the median value for each health attribute for our final change rates (Table 12).

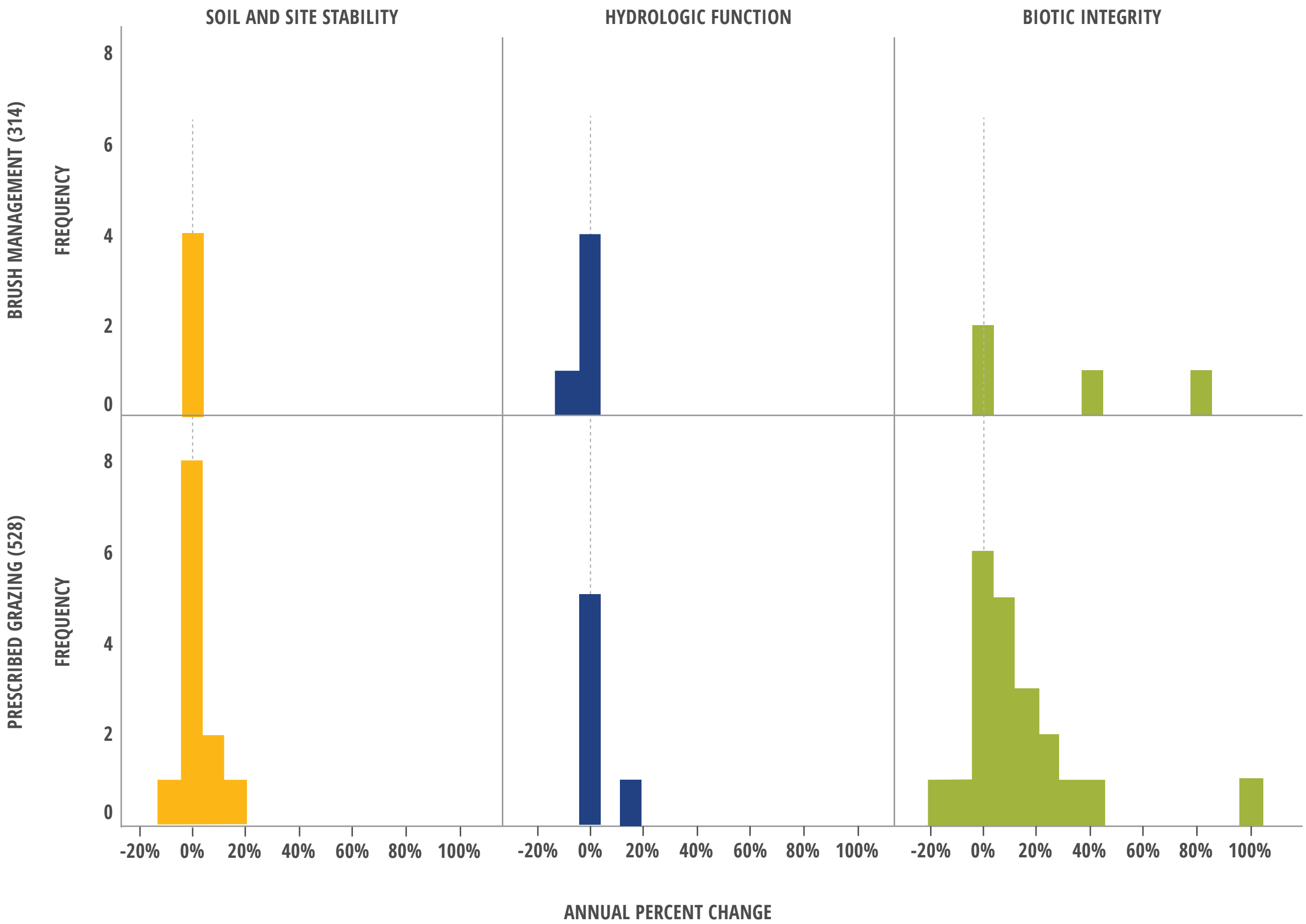
Overall, change rates for soil and site stability and hydrologic function were consistently low across studies, while change rates for biotic integrity showed the greatest variation. Both Brush Management and Prescribed Grazing produced wide variations in biotic integrity.

3.3.1 CAUSAL PATHWAYS

Some conservation practices may not affect the production of certain ecosystem services. We consulted with NRCS subject matter experts to map connections from Brush Management and Prescribed Grazing to rangeland health attributes and to the 12 of 21 ecosystem services valued in this study (see 3.2.3). As described in the previous section, we found evidence to link both practices to each of the three rangeland health attributes (Soil and Site Stability, Hydrological Function, and Biotic Integrity). These health attributes were each connected

[ix] Definitions for these practices can be found in the NRCS National Conservation Practice Standards, www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/technical/cp/ncps/?cid=nrcs143_026849.

FIGURE 14. HISTOGRAM OF RATES OF CHANGE BASED ON REFERENCE LITERATURE IN HEALTH ATTRIBUTES SELECTED



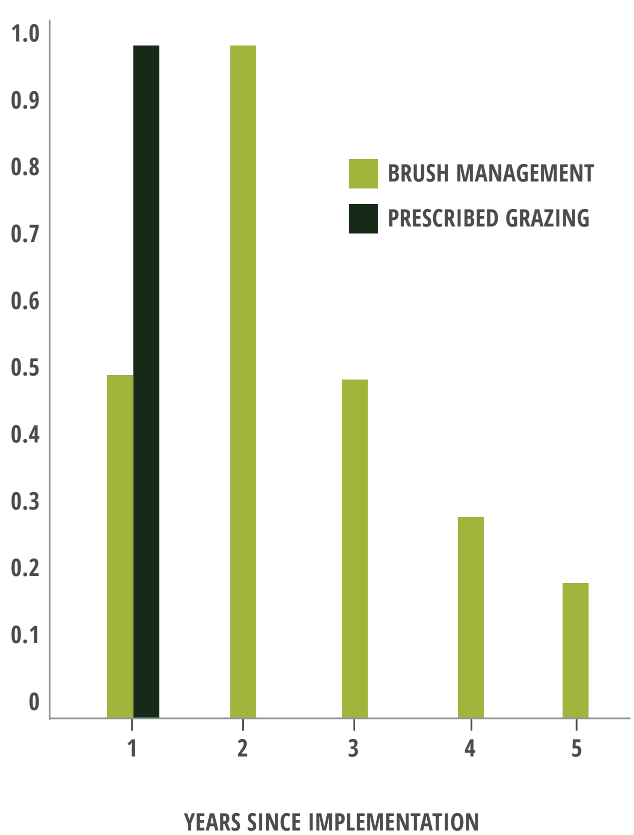
to each of the remaining ecosystem services in turn, except for one relationship—we found no evidence linking Hydrological Function to Water Quality. This is because hydrology focuses on the movement of water, irrespective of its quality.

3.3.2 EFFECTIVENESS OVER TIME

NRCS acknowledges that conservation practices may have limited duration, and that individual practices may be variably effective over time.[40] For example, a fence may be equally effective throughout its lifespan, but the effects of practices such as Brush Management may decline over time. To address such concerns, we referenced NRCS documentation on practice lifespan and conservation practice physical effects to determine the "lag time" for non-structural practices.

To combine the median change rates with practice lifespan and effectiveness over time, we again consulted with NRCS subject matter experts to develop annual effectiveness scores (on a 0-1 scale) relevant to LRR H for each practice. Higher scores signify practices that are more effective at providing the intended benefits to the resource concerns they were designed to address (Figure 15). For each effective year, we multiplied annual improvement to ecosystem health from that practice by the practice effectiveness for that year. For example, if the literature estimated a practice would improve rangeland soil and site stability by 7 percent each year, but that practice is only 50 percent effective in year one, it would provide a benefit to soil and site stability of 3.5 percent in the first year. If that practice were 100 percent effective in year two, it would provide the entire 7 percent improvement in the second year.

FIGURE 15. PRACTICE EFFECTIVENESS OVER 10 YEARS



Because the effects of Brush Management were determined to persist for five years, we estimated the effects of conservation practices for the period 2008-2021. This allows us to account for the full benefits provided by conservation practices within the study area, including those implemented in the final year of the contract period.

While this approach likely oversimplifies relationship between measures of practice effectiveness and ecosystem health, it is important that such dynamics be incorporated—additional research is needed to refine our understanding of the relationships between conservation practices and ecosystem health over time.

3.3.3 PRECIPITATION, TEMPERATURE, AND EVAPOTRANSPIRATION

Weather and climate patterns, particularly in non-irrigated settings, are also significant to the success or failure of most agricultural practices. NRCS rangeland experts expected climate to be a likely factor in the effectiveness of both Brush Management and Prescribed Grazing. Accordingly, we developed county-level "precipitation effectiveness" coefficients that would further adjust practice effectiveness outlined in section 3.3.2. The coefficient was based on monthly Standardized Precipitation Evapotranspiration Index (SPEI-12) data, which incorporates both temperature and rainfall data to monitor drought conditions. We used the SPEI-12 data from December of each year to characterize variations in precipitation and (potential) evapotranspiration over the full calendar year.

We calculated the average value for each pixel in the December SPEI-12[41] images from 1988 to 2017. We then calculated the range of average SPEI pixel values within the boundaries of each MLRA.[x] Since SPEI-12 data include both positive or negative values, we applied a differencing equation to rescale values within each MLRA from 0 to 100, creating an SPEI index value (rSPEI) contextualized to conditions in each MRLA following Equation 2:

$$(2)\ rSPEI_q = \frac{SPEI_q - SPEI_{min}}{SPEI_{max} - SPEI_{min}} \times 100$$

Where:

$rSPEI_q$	is the re-scaled SPEI value for pixel q,
$SPEI_q$	is the SPEI for pixel q,
$SPEI_{max}$ and $SPEI_{min}$	represent the maximum and minimum SPEI value, respectively, within a given MLRA boundary.

We then calculated the average rSPEI value of all pixels within each county, and used these values to estimate the likelihood of implementation success for those practices strongly affected by precipitation (Figure 16).

The darker areas in Figure 16 indicate that the precipitation is "more effective" because evapotranspiration was less than precipitation. In such cases, the lag in attaining full practice effectiveness is likely to be brief. Lighter areas (i.e., lower rSPEI) reflect longer lags before attaining full practice effectiveness, because evapotranspiration is closer to precipitation levels, resulting in “less effective” precipitation. Care should still be taken when interpreting rSPEI values—since rSPEI values reflect variations within the local MLRA, values from one MLRA cannot be meaningfully compared to others, much less the full LRR.

As the rSPEI mapping approach evolved, there was considerable discussion about whether or not to calculate rSPEI values relative to all lower-48 states. However, since mapping a continuous rSPEI layer for the U.S. would result in Arizona having a lower rSPEI than New York state, it became apparent that indices needed to developed for each MLRA independently.

3.3.4 COMBINING EXPECTED EFFECTIVENESS FACTORS

In conclusion, the degree to which ecosystem health is expected to be affected by a practice implemented in a given year was calculated as the product of: (1) the average rSPEI of the county where that practice was implemented; (2) the median rate at which that practice affects that ecosystem health attribute; (3) the effectiveness of that practice over time; and (4) the complement of the health index score from the previous year. The latter is intended to (partially) reflect expectations of diminishing returns—health improvements in ecosystems nearer to their reference conditions are likely to be less dramatic than more degraded ecosystems. In other words, conservation practices can be expected to have relatively greater impacts in counties with highly degraded ecosystems, and magnitude of change driven by conservation practices will decline as ecosystems recover to their reference states.

The resulting expected health attribute changes are unique to each combination of year, county, practice, contract, and baseline health attribute (Equation 3). Table 16 provides the results of carrying through the calculation on one implementation of Brush Management (314) in Logan County, Colorado. The health score in each individual year was used to calculate ecosystem service change due to the practices implemented on the acres affected by that practice (section 3.4).

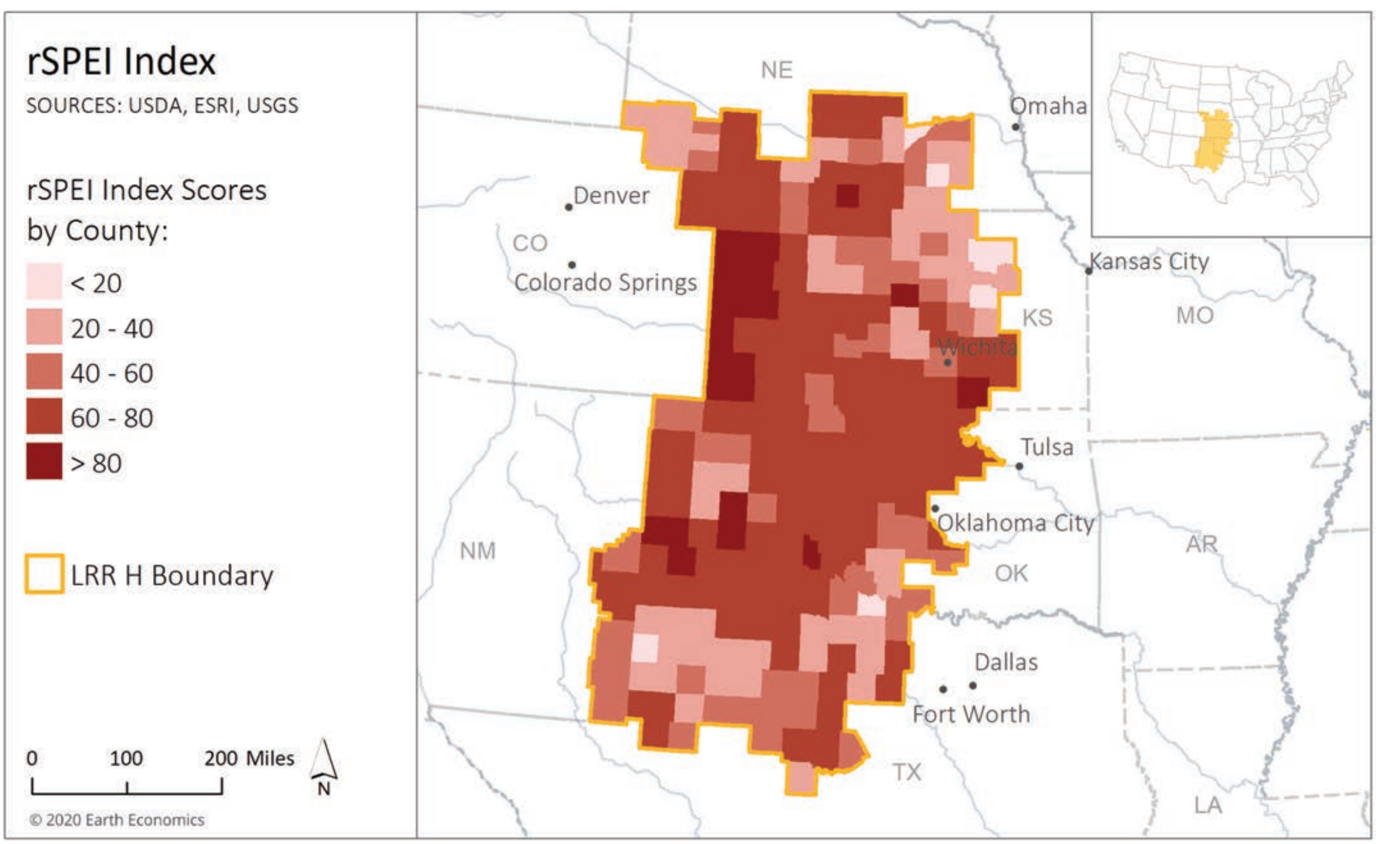
$$(3)\ HC_{ijklp} = rSPEI_j \times cr_{ikl} \times pe_{ilp} \times (1-H_{k,i-1})$$

Where:

HC_{ijklp}	is the change in health attribute *k* by practice *l* on contract p for county *j* in year *i*
$rSPEI_j$	is the SPEI factor for county *j*
cr_{ikl}	is the annual percent change in health attribute *k* for practice *l*
pe_{ilp}	is the practice effectiveness factor for year *i* for practice *l* on contract *p*
$H_{k,\ i-1}$	is the health index for attribute *k* in year *i-1*

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

FIGURE 16. DISTRIBUTION OF AVERAGE BASELINE ECOSYSTEM SERVICE VALUE IN THE STUDY AREA



3.3.5 HEALTH INDEX CHANGE RESULTS DUE TO CONSERVATION PRACTICES

Table 14 shows the average annual change in health attribute index scores due to the implementation of conservation practices (Brush Management and Prescribed Grazing) in 2008–2016, across all contracts and counties in the study area. Figure 17 shows the distribution of these results across counties. For example, Table 14 shows that contracts which implemented Prescribed Grazing in 2008–2016 improved biotic integrity on average by 5.3 percent each year on rangelands in the study region. The degree of improvement by attribute is based on research described in this section.

The values in Figure 17 are influenced by a combination of the following factors: 1) total contracted acres for each practice (Prescribed Grazing and/or Brush Management) in each county; 2) practice effectiveness, and; 3) effective precipitation (rSPEI value) for the county. These data indicate a strong potential for greater improvement of land health, and the rate of that health improvement by implementing suites of conservation practices, rather than single practices in isolation. As section 3.4 explains, the value of ecosystem services associated with each practice showed greater improvement where multiple conservation practices were applied.

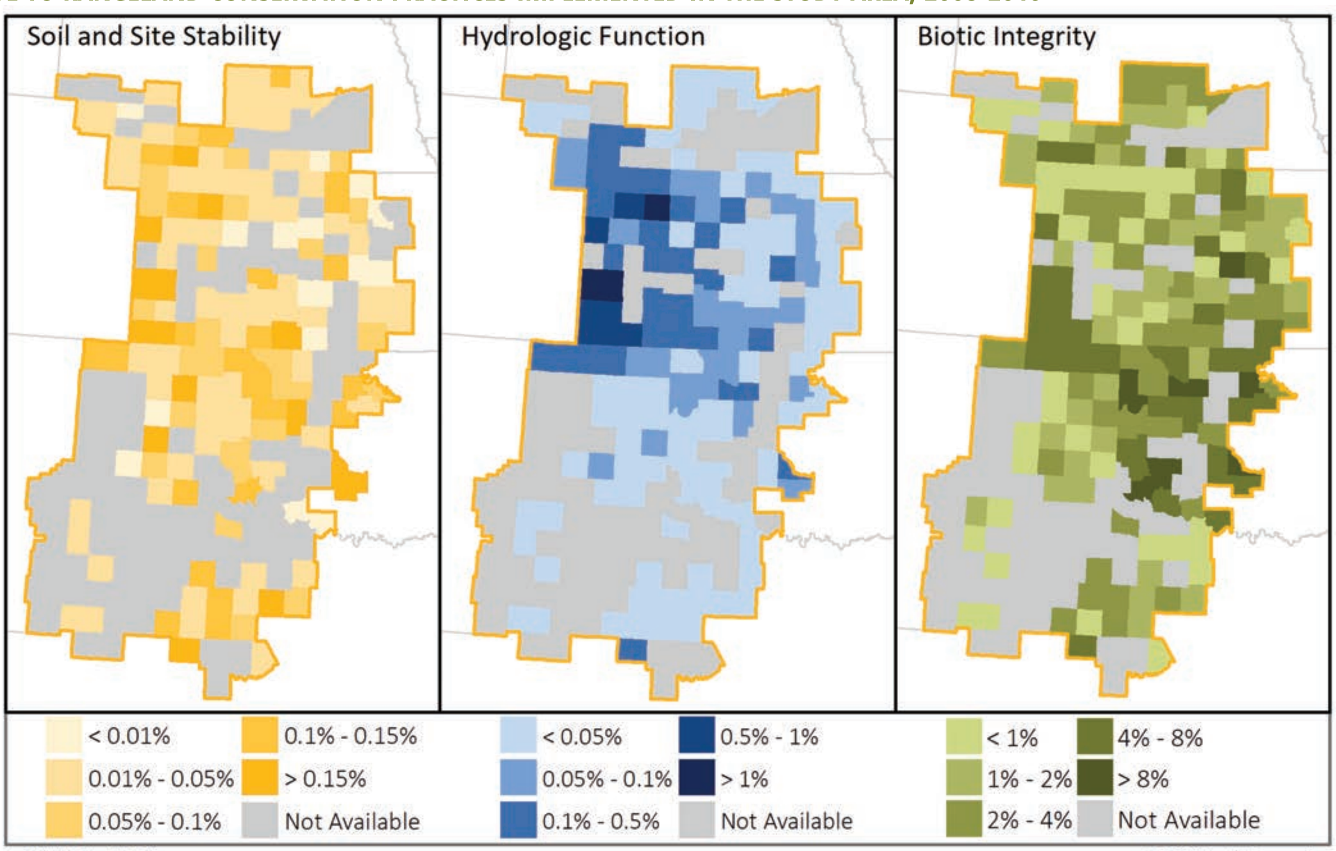
TABLE 14. AVERAGE ANNUAL PERCENT CHANGE IN HEALTH ATTRIBUTES ON CONTRACTS IMPLEMENTED BY PRACTICE

HEALTH ATTRIBUTE	BRUSH MANAGEMENT (314)	PRESCRIBED GRAZING (528)
Soil and Site Stability	0.1%	0.2%
Hydrologic Function	<0.1%	0.4%
Biotic Integrity	0.24	25.63%

TABLE 13. EXAMPLE OF ECOSYSTEM HEALTH CHANGE RESULTS USING BRUSH MANAGEMENT AND BIOTIC INTEGRITY

IMPLEMENTATION YEAR	RSPEII	CHANGE RATE	EFFECTIVENESS	HEALTH SCORE	CHANGE IN
2008	0.24	25.63%	50%	0.520	0.015
2009	0.24	25.63%	100%	0.535	0.029
2010	0.24	25.63%	50%	0.564	0.013
2011	0.24	25.63%	30%	0.577	0.008
2012	0.24	25.63%	20%	0.585	0.005
				TOTAL CHANGE	0.065
				AVERAGE ANNUAL CHANGE	0.013

FIGURE 17. DISTRIBUTION OF AVERAGE PERCENT CHANGE IN HEALTH ATTRIBUTES DUE TO RANGELAND CONSERVATION PRACTICES IMPLEMENTED IN THE STUDY AREA, 2008-2016



SOURCE: USDA

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3.4 QUANTIFYING ECOSYSTEM SERVICE CHANGES DUE TO CONSERVATION PRACTICES

Finally, we identified changes in ecosystem service production and valuation due to NRCS conservation practices applied between 2008 and 2016. First, we identified the specific conservation practices applied—and total acreage to which they were applied—as recorded in each annual NRCS contract. To avoid double-counting, the year a practice was certified by NRCS was considered the first year that a given conservation practice was applied to that landcover, and we estimated the associated changes in ecosystem health as described in section 3.3. We assumed that implementation of more than one practice has additive ecosystem health benefits—a contract certifying two practices 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 certified practice that year.

It is possible that in some circumstances, assuming that impacts are additive might overestimate such impacts. Conversely, conservation practices may have synergistic effects in the provisioning of some ecosystem services and associated resource concerns. In the context of the two practices included in this analysis, we felt that an additive assumption was appropriate, because:

1. We could not determine whether both practices were applied to the same lands;
2. If they had 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. Diminishing returns and the determination of additive benefits would need to be determined on a field-scale basis.

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 then calculated changes in ecosystem service benefits by multiplying the total change in ecosystem health averaged across all health attributes by the spatial extent of each landcover type and the ecosystem service value per acre associated with each landcover type. Summing these values across all landcover types in each county yields the total change in ecosystem service value for each county associated with NRCS contracts for that year (Equation 4):

$$(4)\ ESVC_{ijl} = \Sigma_{kmnp} \frac{HC_{ijklp}}{3} \times D_{mn} \times Ik_{klm} \times A_{np}$$

Where:

$ESVC_{ijl}$	$ESVC_{ijl}$ is the change in ecosystem service value in year i in county j by practice l
H_{ijklp}	H_{ijklp} is the percent change in health attribute k by practice l on contract p for county j in year i
A_{np}	A_{np} is the land cover acres n affected on the contract p
D_{mn}	D_{mn} is the dollar-per-acre-per-year ecosystem service value for service m and landcover n
Ik_{klm}	Ik_{klm} is a binary variable set to 1 if the practice l has been determined to affect the particular health index k and service m, and 0 otherwise

Because applied practices were aggregated to the county level to preserve producer confidentiality, we were unable to determine whether any acres received repeated treatments from 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 county by adding the ecosystem health change for that year to the previous annual health score, weighted by county-level landcover distribution. These new index values were then used as the baseline for the subsequent year. This process was repeated each year. Because practices certified in a given year are likely to provide residual effects (see section 3.3.2), the effects of both newly-certified practices and the residual effects from previously-certified practices were combined until the effective lifespan of a practice was reached.

3.4.1 ESTIMATED VALUE ATTRIBUTABLE TO TWO RANGELAND CONSERVATION PRACTICES

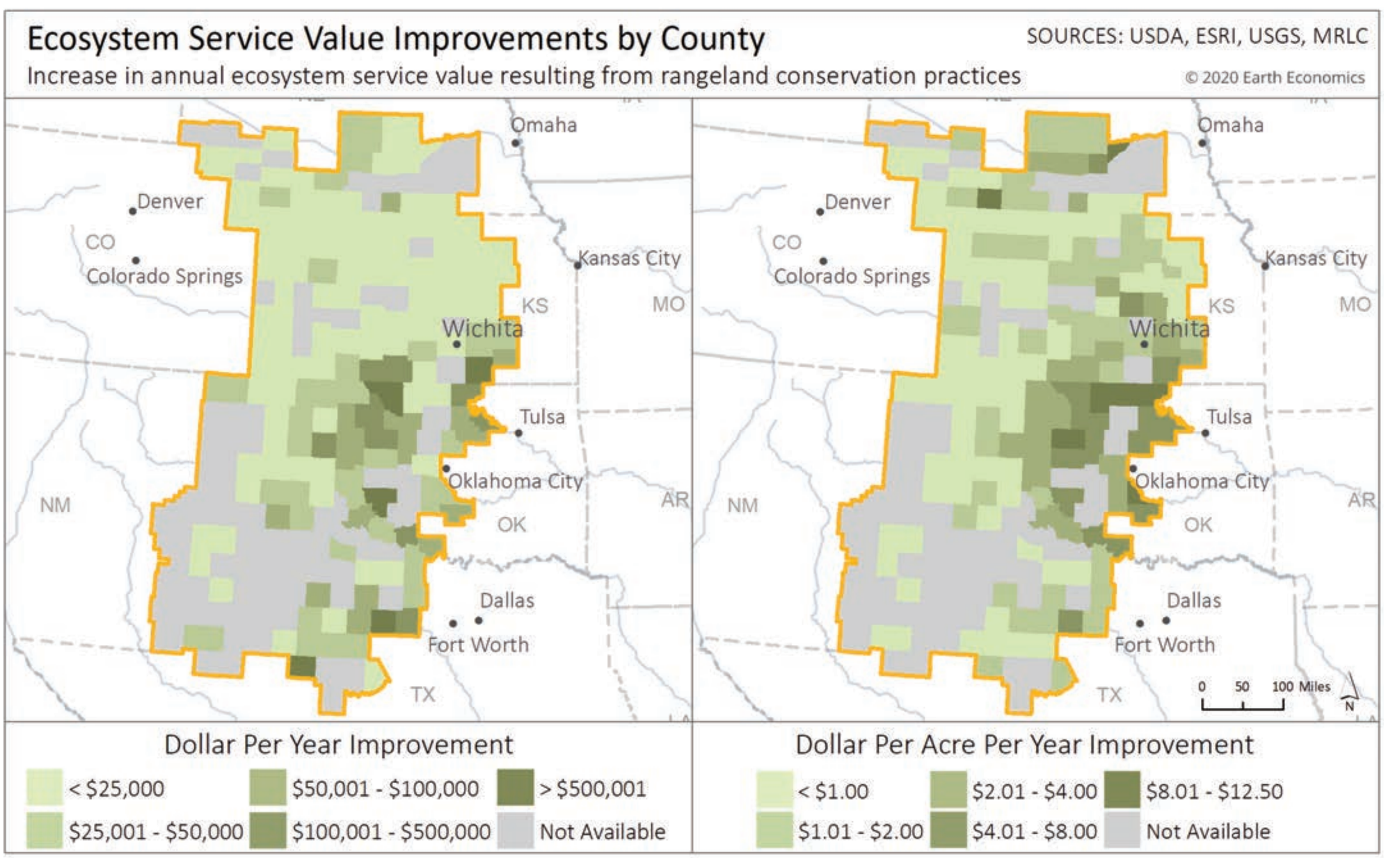
To review, this study estimated the value of ecosystem services produced on rangeland in LRR H that could be attributed to the implementation of two NRCS conservation practices (Brush Management and Prescribed Grazing) between 2008 and 2016. Because the effects of Brush Management persist for five years, these effects were estimated for the period 2008-2021, to ensure that our estimates included benefits provided by practices implemented in the final year of the contract period. These benefits were assessed relative to baseline estimates of the average annual ecosystem services produced throughout the study area between 2004 and 2008, and further limited by the effects of localized MLRA climates, as well as expectations that annual increases would slow as ecosystem health recovered nearer to reference conditions.

We conclude that Brush Management and Prescribed Grazing practices implemented between 2008 and 2016 increased the value provided by ecosystem services across the study area by $15 million to $33 million from 2008-2016. These benefits averaged $1.7 million to $3.6 million per year, or $2.28 to $4.93 annually per affected acre. Figure 19 shows the average annual ecosystem service improvement for each county in the study region. Table 15 shows the average dollar per acre improvement by practice.

TABLE 15. AVERAGE ANNUAL IMPROVEMENT IN MONETIZED ECOSYSTEM SERVICE VALUE DUE TO RANGELAND CONSERVATION PRACTICES APPLIED, FOR THE COUNTIES THAT FALL WITHIN THE STUDY AREA, 2008-2016

CONSERVATION PRACTICE	IMPROVEMENT IN ESV ($/ACRE/YEAR)	
	Low	High
Brush Management (314)	$0.65	$1.36
Prescribed Grazing (528)	$0.54	$1.40

FIGURE 19. DISTRIBUTION OF AVERAGE ANNUAL CHANGE IN ECOSYSTEM SERVICE VALUE DUE TO RANGELAND CONSERVATION PRACTICES IMPLEMENTED IN THE STUDY AREA



4. LIMITATIONS AND SENSITIVITIES OF THE FRAMEWORK

All estimation methods have strengths and weaknesses. Benefit transfer methods (BTM) estimate the economic value of a given ecosystem (e.g., wetlands) based on studies of similar ecosystems. As with any effort to generalize, the main limitation in applying BTM to value ecosystem services is recognition that each ecosystem is unique and that hydrological, chemical, and biological functions vary even within individual ecosystems. This may limit the validity of assuming that unit values (e.g., $/acre/year) derived in one location are relevant to others. Unit values may be further influenced by scale, as scarce benefits tend to be valued more highly than those produced in surplus. In economics, it is generally understood that as supply declines, the *marginal* unit price generally increases—average values are not the same as a range of marginal values.

There is also the question of *exchange value*, the conventional means of determining value, where goods and services are priced in markets. We cannot conceive of any transaction in which most of a large region's ecosystems would be bought and sold. This emphasizes the point that the value estimates for large areas—as opposed to the unit values per acre—are more comparable to national income account aggregates and not exchange values.[42] These aggregates (e.g., GDP) routinely impute values to public goods for which no conceivable market transaction is possible. The value of ecosystem services produced by large geographic regions is comparable to these kinds of aggregates. Ultimately, the use of average values in ecosystem valuation is no more—or less—justified than the use of averages in other macroeconomic contexts, such as in the development of economic statistics such as Gross Domestic Product.

The absence of even imaginary transactions was a prominent criticism of a 1997 study by Costanza et al. that estimated the value of all ecosystems worldwide.[43] Yet one can conceive of an exchange in which, for example, a large portion of a watershed might be sold for development, so that the basic technical requirement of an economic value reflecting the exchange value could be satisfied. Yes even this hypothetical is unnecessary, if one recognizes the different purpose of valuation at the regional scale—a purpose that is more analogous to national income accounting than to estimating exchange values.[42] Moreover, exchange values are less relevant for decisions that affect nonmarket benefits, whether it is estimating the benefits of a conservation program, or calibrating mechanisms to improve such programs.

Alternatively, one could conduct primary valuation of ecosystem services produced on a site-by-site basis. While this approach would not attempt to extrapolate value from study sites to other locations, the scale and complexity of most ecosystems would make large-scale *in situ* valuations so difficult and costly that they would make regional analysis all but impossible. This is the principal reason why we selected BTM as a valuation method.

The studies on which we based our calculations encompass a range of geographic areas, analytical methods, investigators, and time periods. Many provided a range of value 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." Also, we performed only limited sensitivity analyses. In this sense, the approach we present here is similar to determining the *asking price* for a piece of land based on the prices of comparable parcels (aka "comps"). Although each property is unique, realtors and lenders support this process, publicizing solitary asking prices, rather than price ranges.

Our study presents ranges of estimates of the value of ecosystem services over time (years) and space (counties and MLRAs). These estimates have limitations, as indicated above. We believe this report still improves our understanding of the value of NRCS conservation practices in two ways. First, the framework and methods we describe here offer 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.

With better information about the links between land use, management 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. 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 GENERAL LIMITATIONS

STATIC ANALYSIS

This analysis is a static, partial equilibrium framework that ignores interdependencies and dynamics. The impact of this omission on valuations is difficult to assess.

SCARCITY EFFECTS

Where ecosystems in the study area are less able to provide a given service—due to either ecosystem health or the spatial extent of those ecosystems effects—this valuation likely underestimates shifts in the relevant demand curves. The unit value of many ecosystem services rapidly increases as those services become increasingly scarce.44 To the degree that the services identified here are scarcer than assumed, their value has been underestimated in this study. Further reductions in supply appear likely, as land conversion and development continue.

4.2 GEOSPATIAL LIMITATIONS

GEOSPATIAL DATA

Since this application of BTM associates ecosystem service values to landcover types—and their location relative to riparian and urban areas—one of the most important concerns with GIS quality assurance is reliability of the landcover maps, in terms of both categorical precision and accuracy.

SCALE AND RESOLUTION

Large-scale landcover datasets are usually derived from multiple data types and sources which range in scale from coarse to fine, with widely varying spatial and spectral resolutions. Lower resolution source data may result in inadequate data for high value ecosystem units (i.e., wetland, beach, riparian vegetation).

ECOSYSTEM HEALTH

It is possible that the ecosystems identified from the geospatial analysis are functional to the point that they can deliver higher values than those in the primary studies. This would result in an underestimate of current value. On the other hand, if ecosystems are less healthy than those in primary studies, this valuation overestimates current value.

SPATIAL EFFECTS

This ecosystem service valuation assumes spatial homogeneity of services within ecosystems (i.e. that every acre of forest produces the same ecosystem services). This is clearly not the case. Real-world ecosystem productivity emerges from specific patterns of the biophysical relationships producing those services. Addressing such factors would require spatial dynamic analyses, which have shown that including interdependencies and dynamics can lead to significantly higher values, as changes in ecosystem function cascade throughout ecosystems and the benefiting communities.[44]

4.3 BENEFIT TRANSFER/ DATABASE LIMITATIONS

INCOMPLETE COVERAGE

That not all ecosystem services have been well-researched is perhaps the most serious issue, as it often results in a significant underestimate of the full value produced by any given ecosystem. More complete research coverage would almost certainly change the values estimated in this report, and likely increase the total estimates.

SELECTION BIAS

Bias may be introduced in choosing the valuation studies, as in any appraisal methodology. Secondary reviews for appropriateness and rigor limit this potential, and reporting ranges rather than single value estimates (e.g., averages) partially mitigates any remaining issues.

4.4 PRIMARY STUDY LIMITATIONS AND SENSITIVITIES

PRICE DISTORTIONS

Any distortions in the prices used to estimate ecosystem service values (as influenced by subsidies or taxation) have been carried through the analysis. This may occur when nonmarket values are estimated based, in part, on revealed preferences (i.e., implicit exchange value). However, such prices do not reflect environmental externalities and are therefore again likely to underestimate the true value produced by ecosystems.

NON-LINEAR/THRESHOLD EFFECTS

Lacking clear guidance from the supporting literature, we have assumed linear responses to changes in ecosystem extent, with no thresholds or discontinuities. The presence of thresholds or discontinuities would likely result in higher values for affected services. Further, once critical thresholds are passed, valuation may leave the normal sphere of marginal change, where larger-scale social and ethical considerations dominate, such as endangered species listings.

SUSTAINABLE USE LEVELS

The estimates reported here are not necessarily based on sustainable use levels. Limiting demand to sustainable levels would imply higher values for ecosystem services as the effective supply of such services is reduced. If the limitations described above were addressed, estimate ranges would most likely be narrower, with significantly higher values overall. At this point, however, it is impossible to precisely determine how much low or high values might change.

4.4.1 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*. Studies to quantify error in BTM find that, on average, well-designed benefit transfers can produce errors of up to 42 percent.[46] 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. One of the strengths of BTM is that it is a cost-effective and timely means of producing reasonable estimates.

Sensitivity analysis is an important means of assessing the validity of the study.[47] An alternative to conducting a sensitivity analysis is to assess how much BTM estimates vary based on changes in the supporting data.[46] In general, BTM efforts that are less-sensitive to single data points are considered more robust. Absent corroborating evidence, researchers may choose to omit outlying data points as "recording" errors. However, in some cases, outliers may accurately reflect study site conditions. Here, we assessed the BTM sensitivity as the change in total ecosystem service value per acre as each ecosystem-ecosystem service combination valued was dropped from the dataset.[46]

Table 16 shows the relative contribution of each landcover and ecosystem service value to the overall baseline estimate. The largest total value in the study area comes from air quality benefits provided by grassland, followed by water conveyance and supply from wetlands and climate stability and water quality provided by grasslands. These proportions depend both on the per-acre values in the dataset as well as the extent of the landcover-attribute combination to which those values were scaled.

TABLE 16. PROPORTION OF BASELINE TOTAL ECOSYSTEM SERVICE VALUATION (ESV) IN THE STUDY AREA

Ecosystem Services Valued	Grassland	Shrubland	Wetlands
Aesthetic Information	6%		
Air Quality	32%	1%	
Biological Control	<1%		
Climate Stability	9%	2%	<1%
Disaster Risk Reduction	6%		2%
Habitat	3%		1%
Recreation and Tourism	3%	3%	4%
Soil Quality			<1%
Soil Retention	3%		<1%
Water Capture, Conveyance, and Supply			10%
Water Quality	9%		6%
Water Storage			<1%

4.4.2 SENSITIVITY OF ECOSYSTEM SERVICE VALUE CHANGES

We took the same approach to assess sensitivity of the final modeled output based on inclusion (or exclusion) of different ecosystem services in the valuation process. Table 17 shows the proportional contribution of modeled ecosystem values changes that follow from implementing conservation practices on rangeland landcover types. These largely align with the baseline distributions, with grassland air quality showing the highest contribution to the total estimate. However, water quality benefits on grasslands are slightly more dominant than in the baseline estimates, and the other ecosystem services benefits are distributed more evenly across landcover types. Refer to Appendix B and C for detail on the studies that contributed to these proportional changes."

TABLE 17. PROPORTION OF CHANGE IN ESV DUE TO RANGELAND CONSERVATION PRACTICES, BRUSH MANAGEMENT AND PRESCRIBED GRAZING

Ecosystem Services Valued	Grassland	Shrubland	Wetlands
Aesthetic Information	3%		
Air Quality	35%	<1%	
Biological Control	<1%		
Climate Stability	8%	4%	<1%
Disaster Risk Reduction	9%		1%
Habitat	2%		1%
Recreation and Tourism	3%	2%	2%
Soil Quality			<1%
Soil Retention	4%		<1%
Water Capture, Conveyance, and Supply			7%
Water Quality	14%		5%
Water Storage			<1%

5. DISCUSSION

Nature provides benefits that are fundamental to a functioning economy, many of which are never directly exchanged via markets. Accordingly, economic development plans, conservation efforts, and legislation often fail to account for the full value provided by nature. In the 2019 SAFE Act, Congress found that:

> "[H]ealthy, diverse, and productive communities of fish, wildlife, and plants provide significant benefits to the people and economy of the United States, including … abundant clean water supplies; flood and coastal storm protection; clean air; a source of food, fiber, medicines, and pollination of the crops and other plants of the United States; … outdoor recreation …; hunting and fishing opportunities …; opportunities for scientific research and education…"[1]

The acknowledgment that ecosystem services have both social and economic value, combined with the purpose of the Act to "use all practicable means to protect, manage, and conserve healthy, diverse, and productive fish, wildlife, and plant populations" provides an opportunity for NRCS to explore the use of ecosystem service valuation in the planning and programmatic processes.

Estimating the value of ecosystem services is not new to the Federal government. For example, a 2016 study revealed that U.S. households are willing to pay at least $92 million to avoid the loss of the National Park Service and its programs. Granted, taxpayers do not have access to the private lands where the bulk NRCS funds are provided to address resource concerns and conservation, nor did we directly adopt either a willingness to pay or avoidance cost methodology in this framework—though the dataset includes values estimated by such methods. Yet NRCS provides an average of $170 million annually for conservation treatments on federally-owned lands that are accessible to the public (2016 unpublished internal NRCS report based on 10-years of NRCS data). This framework illustrates that conservation assistance on private grazing lands has ecosystem benefits that extend beyond fence lines (e.g., air and water quality improvements, disaster risk reduction), allowing us to associate NRCS practices on both private and non-private lands with benefits to taxpayers.

NRCS has been a pioneer in conservation for decades, and this integrated ecological-economic framework can be used to provide an indication of the value of the ecosystem services provided from NRCS conservation investments. By accounting for the contributions of natural systems, we can make informed, strategic decisions to support the long-term prosperity and resilience of our economy, landscapes, and communities. Expanding our understanding of economic benefits can also help to prioritize the most effective practices, and design incentive programs to compensate or reward land managers for their voluntary conservation efforts. Including the nonmarket value of ecosystem services in conservation planning may lead producers to choose practices they may not otherwise adopt.

There are numerous examples of ranchers who market "non-market" ecosystem services,[49] but many others have no idea how to begin that conversation—this framework could help by establishing baseline estimated prices for such services. The value provided by conservation practices could also help NRCS demonstrate the added-value of such practices to the general public. The potential inclusion of such ecosystem service values in NRCS payment schedules would directly link NRCS financial and technical assistance to outcomes the general public considers important. While we were able to monetize only two conservation practices implemented in the study region—and their effect on a portion of all ecosystem services known to be produced on those lands—these estimates suggest that conservation practices provide significant non-market benefits to both on-farm and off-farm beneficiaries.

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.

5.1 RECOMMENDATIONS AND NEXT STEPS

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. 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.

Given limitations of data and relevant literature, these findings establish a starting point for ongoing discussion and research. 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 that will improve study resolution and comprehensiveness.

FILL GAPS IN ECOSYSTEM SERVICE ASSESSMENTS

Ecosystem valuation research on natural ecosystems dominated by grasses or shrubs (i.e., rangelands) is quite scarce—we were only able to value a portion of ecosystem goods and services provided by rangelands. A number of landcover and ecosystem service combinations could not be valued due to such limitations. Expanding the published primary valuation research of such services on rangeland landcover types would help address gaps in rangeland ecosystem services analyses. As such, the values presented in this report likely underestimate the true value of ecosystem services provided within the region, and the influence of NRCS conservation practices on those benefits.

FILL GAPS IN RESEARCH ON HOW CONSERVATION PRACTICES AFFECT LAND HEALTH

Only two NRCS conservation practices were valued in this analysis. Moreover, impacts on ecosystem function are better documented for some practices than others. Researchers should continue to produce primary research on the quantitative effects of conservation practices on aspects of ecosystem health. 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 which improve grazing management impacts, but in this study, we evaluated only NRCS contracts explicitly for Prescribed Grazing or Brush Management. Additional research is needed on the relationships between facilitating practices and management practices, as well as the virtue of applying facilitating practices in isolation.

CONDUCT MORE DETAILED ASSESSMENT OF ECOSYSTEM HEALTH ATTRIBUTES

We estimated baseline ecosystem health based on NRI rangelands data, which had been collected between 2004 and 2008. Data was missing for several counties within the study region. More comprehensive data would improve ecosystem health estimates across all counties within the region.

One alternative is to use modeled baseline data (e.g., annual soil loss and water runoff values), rather than NRI site assessment data. We were unable to pursue this approach due to the lack of models for rangelands, in relation to conservation practices. Refer to the discussion below in the “Perform modeling and validation in the region” section below.

CONDUCT ANALYSIS ON OTHER LAND USES

Establishing health metrics for other land uses (e.g., cropland, forest land) 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 COMBINATION

We assumed linear relationships between health indices and ecosystem services due to a paucity of appropriate modeling, research, and other data relevant to the study area. We recognize that these relationships could take on any number of forms in actuality. A more true-to-life representation of these relationships would improve estimates of changes in ecosystem service value attributable to conservation practices.

PERFORM MODELING AND VALIDATION IN THE REGION

Models of landscape changes due to conservation practices were not available when we conducted this study. However, models linking practice implementation to biophysical outputs could support function transfers, which are generally considered more accurate than benefit transfers. Comparisons could then be made between the results in this report and such modeled results.

BRING ECOSYSTEM SERVICE VALUATION INTO STANDARD ACCOUNTING AND DECISION-MAKING TOOLS

There are several ongoing efforts to integrate ecosystem services into accounting frameworks. Accounting rules currently recognize timber and fossil fuel as natural capital assets, but the range of environmental resources allowable in accounting frameworks need to be expanded. Ecosystem service valuation can provide a useful framework for this expansion, offering governments, utilities, businesses, and private landowners a means of calculating rates-of-return on conservation and restoration investments, or to include the value produced by natural capital within benefit-cost analyses. Integrating ecosystem services into these types of accounting frameworks may shift public and private investment towards more productive and resilient projects.

6. APPENDIX A
MLRA-SCALE BASELINE VERSUS RESULTS TABLES

This appendix presents results from Chapter 3 aggregated to the MLRA level. Each county was assigned to a unique MLRA (Section 3). Results for counties assigned to each MLRA were summed or averaged, as appropriate.

TABLE 18. AVERAGE BASELINE RANGELAND HEALTH ATTRIBUTE SCORES BY MLRA

MLRA	Soil and Site Stability	Hydrologic Function	Biotic Integrity
71	0.90	0.75	0.63
72	0.82	0.65	0.53
73	0.90	0.75	0.62
74	0.88	0.69	0.53
75	0.87	0.74	0.66
76	0.97	0.80	0.65
79	0.96	0.79	0.66
77A	0.81	0.59	0.44
77B	*	*	*
77C	0.90	0.82	0.76
77E	0.90	0.78	0.69
78A	0.93	0.80	0.68
78B	0.90	0.82	0.77
78C	0.85	0.71	0.62
80A	0.86	0.66	0.51
80B	0.91	0.68	0.49

**no data*

TABLE 19. AVERAGE BASELINE RANGELAND HEALTH ATTRIBUTE SCORES BY MLRA

MLRA	$/Year (Millions)		$/Acre/Year	
	Low	High	Low	High
71	294	437	112	167
72	201	761	32	121
73	360	617	71	122
74	110	230	73	152
75	45	58	63	80
76	289	650	84	189
77A	11	141	5	64
77B	*	*	*	*
77C	23	272	8	90
77D	15	273	3	59
77E	220	636	47	136
78A	78	155	57	114
78B	107	354	22	72
78C	509	982	77	148
79	111	211	99	189
80A	440	852	108	210
80B	167	300	108	194
Total	**$2,981**	**$6,930**	**$55**	**$129**

**no data*

TABLE 20. AVERAGE ANNUAL PERCENT CHANGE IN RANGELAND HEALTH ATTRIBUTE VALUES FROM BRUSH MANAGEMENT AND PRESCRIBED GRAZING PRACTICES IMPLEMENTED BY MLRA WITHIN LRR H, 2008-2016

MLRA	Soil and Site Stability	Hydrologic Function	Biotic Integrity
71	0.02	0.01	1.22
72	0.05	0.17	1.25
73	0.02	0.03	1.02
74	0.03	0.03	1.82
75	0.02	0.01	0.9
76	0.02	0.01	1.57
79	0.01	0.03	1.28
77A	0.06	0.16	2.19
77B	*	*	*
77C	0.01	0.01	0.43
77D	0.02	0.03	0.52
77E	0.02	0.04	2.18
78A	0.04	0.01	0.98
78B	0.06	0.03	1.78
78C	0.06	0.03	3.05
80A	0.03	0.02	2.97
80B	0.06	0.02	1.36

*no data

TABLE 21. AVERAGE ANNUAL IMPROVEMENT IN MONETIZED ECOSYSTEM SERVICE VALUE DUE TO CONSERVATION PRACTICES APPLIED, BY MLRA, 2008-2016

MLRA	$/Year (Millions)		$/Acre/Year	
	Low	High	Low	High
71	38	55	3.44	4.89
72	35	131	0.61	2.26
73	78	121	1.79	2.88
74	28	57	2.95	6.09
75	12	15	1.36	1.66
76	119	266	4.34	9.69
77A	7	87	0.28	3.33
77B	*	*	*	*
77C	5	59	0.13	1.63
77D	3	44	0.08	1.1
77E	73	193	1.39	3.58
78A	67	132	1.72	3.4
78B	184	457	1.15	2.91
78C	485	993	4.59	9.42
79	24	46	2.55	4.91
80A	323	627	5.79	11.19
80B	217	366	5.44	9.16
Total	1,698	3,647	2.28	4.93

*no data

FIGURE 20. DISTRIBUTION OF AVERAGE BASELINE ECOSYSTEM SERVICE VALUE IN LRR H, BY MLRA

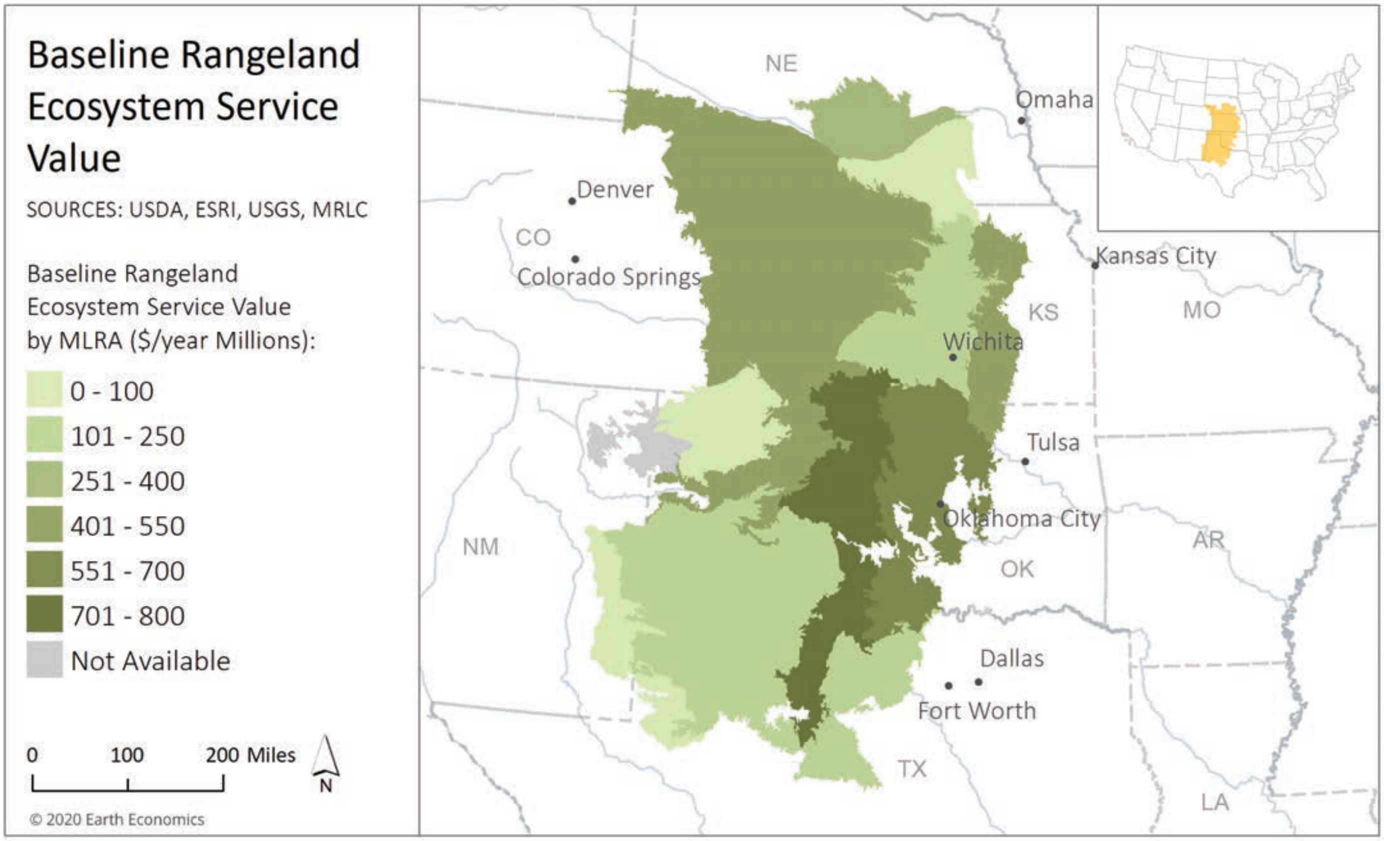


FIGURE 21. DISTRIBUTION OF AVERAGE PERCENT CHANGES IN HEALTH ATTRIBUTES DUE TO RANGELAND CONSERVATION PRACTICES (BRUSH MANAGEMENT AND PRESCRIBED GRAZING) IMPLEMENTED IN LRR H, 2008-2016, BY MLRA

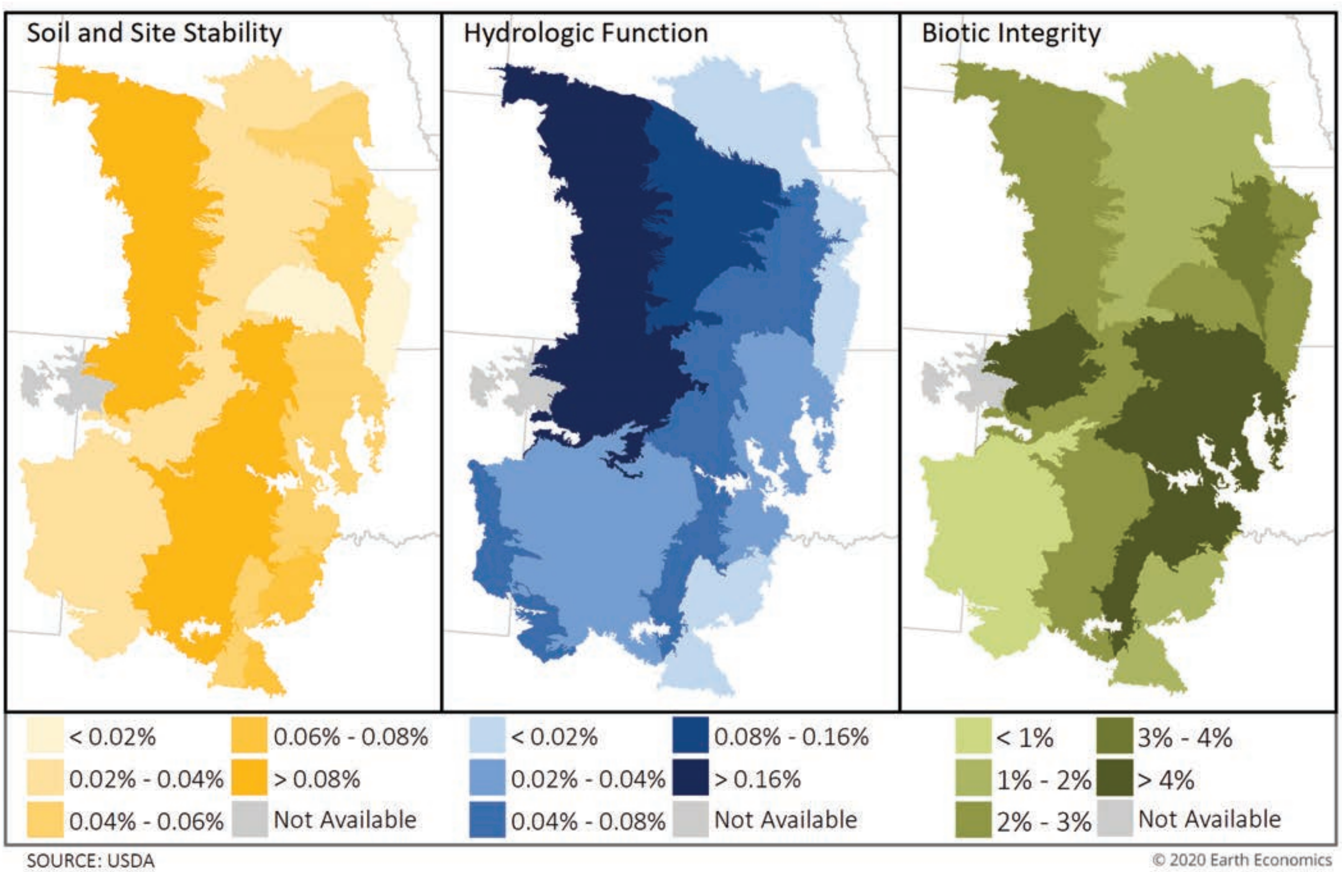
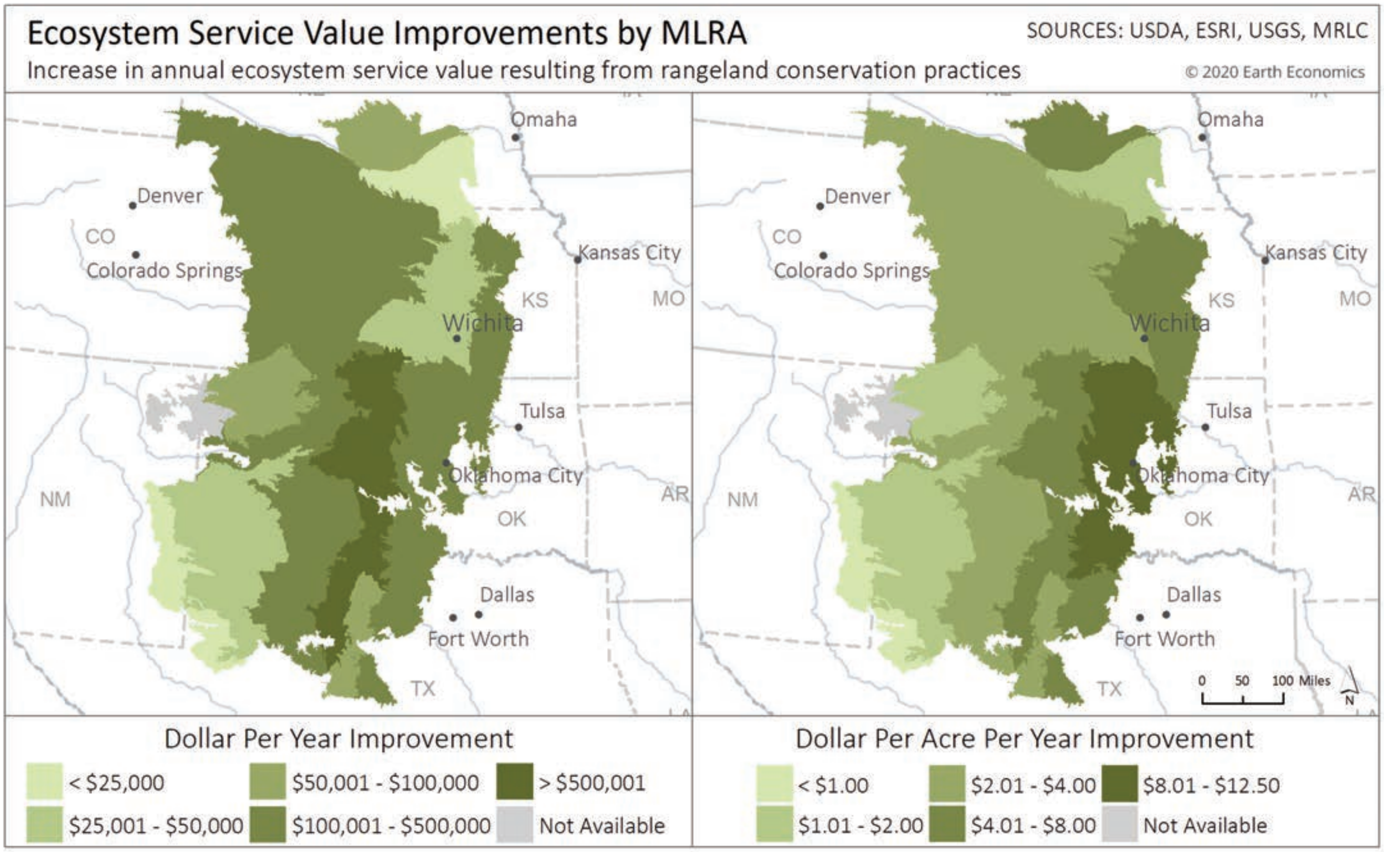


FIGURE 22. DISTRIBUTION OF AVERAGE ANNUAL CHANGE IN ECOSYSTEM SERVICE VALUE DUE TO BRUSH MANAGEMENT AND PRESCRIBED GRAZING IMPLEMENTED IN LRR H, BY MLRA



7. APPENDIX B
ECOSYSTEM SERVICE VALUATION REFERENCES

The following references were used to quantify the economic valuation of ecosystem services in the framework. We provided specific information (bulleted items) showing how/where the valuation reference data was applied, but we did not include the raw or adjusted dollar values, as those are part of a proprietary database owned and managed by Earth Economics.

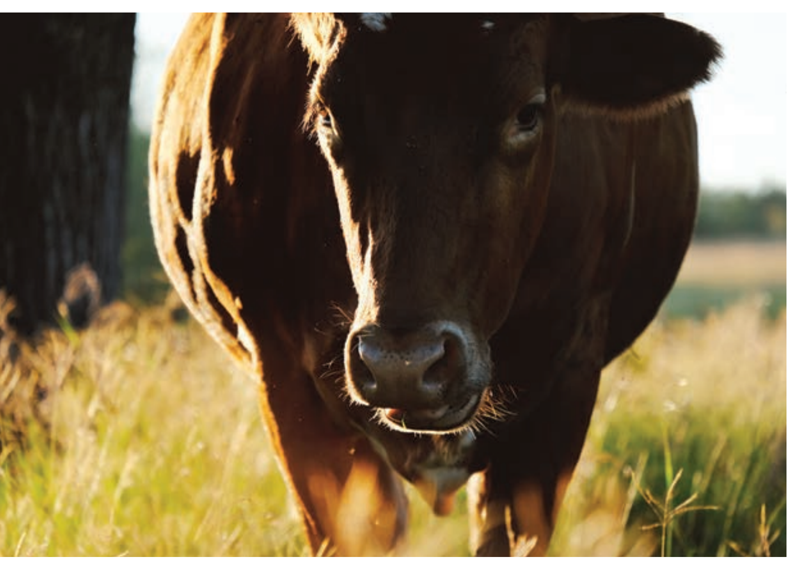
Brander, L. M., Brouwer, R., Wagtendonk, A. 2013. Economic valuation of regulating services provided by wetlands in agricultural landscapes: A meta-analysis. Ecological Engineering 56: 89-96.

- Site: United States
- Landcover Types: Wetland
- Climate Groups: B, C, D
- Spatial Attribute: None
- Ecosystem Services: Water Capture, Conveyance, & Supply
- Valuation Methodology: Meta-Analysis
- Sample Size: 66
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 87%

A meta-analysis of wetland economic valuation literature which estimated regulating services in agricultural landscapes. We applied the water supply value estimated in the meta-regression model for wetlands in the study area, as no other suitable values were found in the literature.

Bridgeham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C. 2006. The carbon balance of North American wetlands. Wetlands 26(4): 889-916.

- Site: United States
- Landcover Types: Wetland
- 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 carbon stored by wetlands in North America. We monetized this value 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 the contiguous United States to wetlands of all types in the study area.

Butler, L. D., Workman, J. P. 1993. Fee hunting in the Texas Trans Pecos area: A descriptive and economic analysis. The Journal of Range Management 46(1): 38-42.

- Site: Texas
- Landcover Types: Grassland, Shrubland
- Climate Groups: B
- Spatial Attribute: None
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Market Price
- Sample Size: 130
- Average Study Site Annual Household Income: $43,000
- Percent of Population Attaining High School Level Education or Greater: 69%

Surveyed fees charged to hunt on private ranches in the Trans-Pecos region of Texas where 90 percent of landcover is rangeland (a mix of grasses and shrubs). While this area is not within the study area boundaries, we include it due to its close proximity and similar landcover types; we applied these values only to arid grasses and shrub lands within the study area. The population in the Trans-Pecos region has a median household income 14 percent lower than in the study area, and generally lower educational attainment rates. There were 130 usable survey responses, with an overall response rate of 45 percent.

Colby, B. G., Smith-Incer, E. 2005. Visitor Values and Local Economic Impacts of Riparian Habitat Preservation: California's Kern River Preserve. Journal of the American Water Resources Association 41(3): 709-717.

- Site: California
- Landcover Types: Wetland
- Climate Groups: B
- Spatial Attribute: Riparian
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Contingent Valuation
- Sample Size: 156
- Average Study Site Annual Household Income: $51,000
- Percent of Population Attaining High School Level

Education or Greater: 74%

Used contingent valuation to estimate annual willingness-to-pay for visits to a wetland preserve in an arid region of California, known for exceptional birding opportunities. We applied the visitation estimates to recreation value on arid riparian wetlands in the study area.

Cooper, J., Loomis, J. B. 1991. Economic value of wildlife resources in the San Joaquin Valley: Hunting and viewing values. Dinar, Ariel, Zilberman, David (eds.) Kluwer Academic Publishers.

- Site: California
- Landcover Types: Wetland
- Climate Groups: B
- Spatial Attribute: Riparian
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Travel Cost
- Sample Size: 1300
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 78%

Quantified recreation values in the San Joaquin Valley of California using travel cost models, estimating values for birdwatching and waterfowl hunting, with a survey response rate of 44 percent. The original study site has a median household income 17 percent higher than in the study area, but generally lower educational attainment rates. We applied these recreation values to arid wetlands within the study area.

Creel, M., Loomis, J. B. 1992. Recreation Value of Water to Wetlands in the San Joaquin Valley: Linked Multinomial Logit and Count Data Trip Frequency Models. Water Resources Research 28(10): 2597-2606.

- Site: California
- Landcover Types: Wetland, Unspecified
- Climate Groups: B
- Spatial Attribute: Riparian
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Travel Cost
- Sample Size: 1141
- Average Study Site Annual Household Income: $51,000
- Percent of Population Attaining High School Level Education or Greater: 74%

Estimated recreational benefits provided by wetlands in the San Joaquin Valley of California, using travel cost models for waterfowl hunting, fishing, and wildlife viewing. There were 1,141 usable survey responses, with an overall response rate of 35 to 51 percent, varying by survey type. The study site population has a median household income similar to the study area, but generally lower educational attainment rates. We applied the recreation values only to arid wetlands within the study area.

Delfino, K., Skuja, M., Albers, D. 2007. Economic Oasis: Revealing the True Value of the Mojave Desert.

- Site: Arizona; California; Nevada; Utah
- Landcover Types: Shrubland
- Climate Groups: B
- Spatial Attribute: None; Riparian
- Ecosystem Services: Air Quality; Water Storage
- Valuation Methodology: Avoided Cost; Market Price
- Sample Size: Not reported
- Average Study Site Annual Household Income: $55,000
- Percent of Population Attaining High School Level Education or Greater: 88%

Broadly defined economic contributions of ecosystem service benefits provided by the Mojave Desert, using benefit transfer to estimate the value of reduced air particulates. We applied that to arid shrubland in the study area, due to the high transferability of air quality values (refer to section 3.2.3.1, Identifying Studies for Use in BTM, and Table 10).

DeLonge, M.S., Ryals, R., Silver, W. 2013. A Lifecycle Model to Evaluate Carbon Sequestration Potential and Greenhouse Gas Dynamics of Managed Grasslands. Ecosystems 16: 962-979.

- Site: California
- Landcover Types: Grassland
- Climate Groups: C
- 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: 83%

Modeled carbon flux over several scenarios in managed grasslands in California. We monetized this value 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 this value to temperate grasslands within the study area.

Feagin, R. A., Martinez, M. L., Mendoza-Gonzalez, G., Costanza, R. 2010. Salt Marsh Zonal Migration and Ecosystem Service Change in Response to Global Sea Level Rise: A Case Study from an Urban Region. Ecology and Society 15(4): 14-32.

- Site: Texas
- Landcover Types: Wetland, Unspecified
- Climate Groups: C
- Spatial Attribute: Riparian
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Contingent Valuation; Travel Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $66,000
- Percent of Population Attaining High School Level Education or Greater: 88%

Estimated recreation benefits in East Texas wetlands, using both contingent valuation and travel cost methods. The original study site median household income is about 30 percent higher than in the study area, and higher educational attainment rates. We applied these values to temperate riparian wetlands within the study area.

Gascoigne, W. R., Hoag, D., Koontz, L., Tangen, B. A., Shaffer, T. L., Gleason, R. A. 2011. Valuing ecosystem and economic services across land use scenarios in the Prairie Pothole Region of the Dakotas, USA. Ecological Economics 70(10): 1715-1725.

- Site: North Dakota; South Dakota
- Landcover Types: Grassland
- Climate Groups: D
- Spatial Attribute: None
- Ecosystem Services: Habitat; Soil Retention
- Valuation Methodology: Contingent Valuation; Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 92%

Used biophysical values derived from the Prairie Pothole Region of the US to assess tradeoffs under different land use scenarios, focusing on croplands and grasslands. We selected two values from this study: waterfowl habitat and soil retention. The authors used a contingent valuation study (previously conducted in the same area) to value waterfowl production, and the RUSLE model and estimates of soil conservation benefits (Economic Research Service) to value the soil retention benefit of grassland. The original study site has a median household income approximately 10 percent higher than the study area, and higher educational attainment rates.

Hansen, L., Feather, P., Shank, D. 1999. Valuation of Agriculture's Multi-site Environmental Impacts: An Application to Pheasant Hunting. Agriculture and Resource Economics Review 28(2): 199-207.

- Site: Midwestern States
- Landcover Types: Grassland
- Climate Groups: D, C
- Spatial Attribute: None
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Travel Cost
- Sample Size: 5834
- Average Study Site Annual Household Income: $55,000
- Percent of Population Attaining High School Level Education or Greater: 91%

Estimated the benefits of pheasant hunting on lands across 13 Midwestern states, using travel cost methods. Surveyed lands were all enrolled in the NRCS CRP program, and included areas within the study area. The original study area included temperate and continental climates of the US; we applied the estimates all grasslands and rangelands in the study area, except those in arid areas. Based on 5,834 survey respondents, and the authors calculated the average consumer surplus throughout the entire study area.

Harrison, G. L. 2014. Economic Impact of Ecosystem Services Provided by Ecologically Sustainable Roadside Right of Way Vegetation Management Practices. Florida Department of Transportation.

- Site: Florida
- Landcover Types: Grassland
- Climate Groups: C
- Spatial Attribute: None
- Ecosystem Services: Air Quality
- Valuation Methodology: Benefit Transfer
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $51,000
- Percent of Population Attaining High School Level Education or Greater: 88%

Estimated the economic value of several ecosystem services for roadside right-of-way ecosystems in Florida, including air quality. The authors estimated the annual pollutant removal of herbaceous vegetation, based on six previously published sources throughout the U.S. We transferred only this value, due to its high transferability. Since Florida is largely a temperate climate, we applied that estimate only to temperate grasslands within the study area.

Hovde, B., Leitch, J. A. 1994. Valuing Prairie Potholes: Five Case Studies. North Dakota State University.

- Site: North Dakota
- Landcover Types: Wetland
- Climate Groups: D
- Spatial Attribute: Riparian
- Ecosystem Services: Disaster Risk Reduction; Recreation & Tourism; Soil Retention
- Valuation Methodology: Benefit Transfer; Market Price; Travel Cost; Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $56,000
- Percent of Population Attaining High School Level Education or Greater: 91%

Estimated the value of multiple ecosystem services provided by prairie wetlands in North Dakota. We selected values for recreation and soil retention. Recreation was estimated using travel costs, while soil retention was estimated based on the avoided cost of soil excavation which would occur in absence of those wetlands. The median household income at the selected study sites is 13 percent higher than LRR H, and the population has a similar educational attainment levels. The original study sites were all in continental climates—we applied these values only to wetlands within the same climate zone within the study area.

Ingraham, M. W., Foster, S. G. 2008. The value of ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S. Elsevier B.V.

- Site: United States
- Landcover Types: Wetland
- Climate Groups: B, C, D
- Spatial Attribute: Riparian
- Ecosystem Services: Soil Quality; Water Capture, Conveyance, & Supply
- Valuation Methodology: Meta-Analysis
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 87%

A broad meta-analysis of the value of ecosystem services provided by ecosystems within National Wildlife Refuges throughout the lower-48 states. We selected values for soil quality, and water capture, conveyance, and supply produced by wetlands. The original study sites span

the entire country—the median US household income is 17 percent higher than the study area and has higher educational attainment rates. We applied these values to all wetlands within the study area.

Ko, J. 2007. The Economic Value of Ecosystem Services Provided by the Galveston Bay/Estuary System. Texas Commission on Environmental Quality.

- Site: Texas
- Landcover Types: Wetland
- Climate Groups: C
- Spatial Attribute: Riparian
- Ecosystem Services: Disaster Risk Reduction, Habitat, Recreation & Tourism
- Valuation Methodology: Replacement Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $66,000
- Percent of Population Attaining High School Level Education or Greater: 88%

Evaluated avoided and replacement costs of several ecosystem services provided by wetlands in east Texas. They estimated the value of habitat as the avoided costs of protecting and restoring wetland habitats. Flood risk reduction values were based the replacement cost of constructing flood control structures, when compared to natural wetland flood mitigation. Wetland recreation value were estimated based on the replacement cost of constructing artificial wetlands providing similar recreational functions. The original study site falls outside of the study area, and values for these services have low-to-medium transferability. The median household income in east Texas is about 30 percent higher than in the study area and has higher educational attainment rates. We applied these values only to riparian, temperate wetlands within 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.

Mast, J. C. 2002. Clarifying Ambiguity: Public Policy, Contingent Valuation and the Consideration of Environmental Aesthetics. ProQuest Information and Learning Company.

- Site: Arizona
- Landcover Types: Grassland
- Climate Groups: B
- Spatial Attribute: None
- Ecosystem Services: Aesthetic Information
- Valuation Methodology: Contingent Valuation
- Sample Size: 154
- Average Study Site Annual Household Income: $54,000
- Percent of Population Attaining High School Level Education or Greater: 87%

Evaluated Arizona residents' willingness-to-pay for aesthetics on public grasslands within that state. We chose this value over others due to Arizona's proximity to LRR H; we applied it to arid grasslands within the study area. Arizona's median household income is 8 percent higher than the study area, and the state has slightly higher educational attainment rates. There were 154 usable survey responses, with a 42.5 percent response rate, overall.

Poor, P. J. 1999. The Value of Additional Central Flyway Wetlands: The Case of Nebraska's Rainwater Basin Wetlands. Journal of Agricultural and Resource Economics 24(1): 253-265.

- Site: Nebraska
- Landcover Types: Wetland
- Climate Groups: D
- Spatial Attribute: Riparian
- Ecosystem Services: Habitat
- Valuation Methodology: Contingent Valuation
- Sample Size: 952
- Average Study Site Annual Household Income: $57,000
- Percent of Population Attaining High School Level Education or Greater: 91%

Determined Nebraskan's willingness-to-pay for wetland habitat, using contingent valuation. Those habitat services have low transferability, the original study site is within LRR H. The population within the study are has a 15 percent higher median household income than LRR H and has higher educational attainment rates. As original study site has a continental climate, we applied this estimated only to

continental-climate wetlands within the study area.

Rein, F. A. 1999. An economic analysis of vegetative buffer strip implementation. Case study: Elkhorn Slough, Monterey Bay, California. Coastal Zone Management Journal 27(4): 377-390.

- Site: California
- Landcover Types: Grassland, Wetland
- Climate Groups: C
- Spatial Attribute: Riparian
- Ecosystem Services: Soil Retention, Disaster Risk Reduction, Biological Control
- Valuation Methodology: Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $63,000
- Percent of Population Attaining High School Level Education or Greater: 71%

Assessed multiple economic service benefits provided by riparian vegetative buffers in Monterey County, California, which has a temperate climate. We selected estimates for biological control, disaster risk reduction, and soil retention— all services with medium or high transferability. The authors estimated these values as avoided costs for herbicide use, flood damage, and erosion reduction, respectively, at a watershed-level scale. We adjusted the values from watershed to per-acre scale with the site acreage given in the study and applied them to riparian grasslands within temperate zones of the study area. The population in the original study area has 28 percent higher median household income than in the study area, but lower educational attainment rates.

Richardson, R. B. 2005. The Economic Benefits of California Desert Wildlands: 10 Years Since the California Desert Protection Act of 1994. The Wilderness Society.

- Site: California
- Landcover Types: Shrubland
- Climate Groups: B

- Spatial Attribute: None
- Ecosystem Services: Air Quality
- Valuation Methodology: Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $54,000
- Percent of Population Attaining High School Level Education or Greater: 84%

Estimated the economic benefits produced by wildlands in California deserts, an arid region. Air quality benefits were based on avoided public health costs. The original study area has a 10 percent higher median income than the study area and has slightly lower educational attainment rates.

Richer, J. 1995. Willingness to Pay for Desert Protection. Contemporary Economic Policy 13(4): 93-104.

- Site: California
- Landcover Types: Shrubland
- Climate Groups: B
- Spatial Attribute: None
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Contingent Valuation
- Sample Size: 264
- Average Study Site Annual Household Income: $54,000
- Percent of Population Attaining High School Level Education or Greater: 84%

Estimated willingness-to-pay to protect deserts in eastern California. There were 264 usable survey responses, with a response rate of 38.6 percent. We applied this value to arid shrubland within the study area. The original study area has a 10 percent higher median household income than the study area, but slightly lower educational attainment rates.

Roberts, L. A., Leitch, J. A. 1997. Economic valuation of some wetland outputs of mud lake, Minnesota-South Dakota. North Dakota State University.

- Site: South Dakota, Minnesota
- Landcover Types: Wetland
- Climate Groups: D
- Spatial Attribute: Riparian
- Ecosystem Services: Disaster Risk Reduction, Recreation & Tourism, Water Storage
- Valuation Methodology: Contingent Valuation, Avoided Cost, Replacement Cost
- Sample Size: 250
- Average Study Site Annual Household Income: $54,000
- Percent of Population Attaining High School Level Education or Greater: 91%

Estimated the value of multiple ecosystem services produced by a wetland along the border of South Dakota and Minnesota, including flood control, water supply, and recreation. These benefits were estimated from avoided cost, replacement cost, and contingent valuation, respectively. We apply these values to wetlands within continental climate zones of the study area. The original study site has a similar median household income to the study area, and higher educational attainment rates.

Ryals, R., Silver, W.L. 2013. Effects of organic matter amendments on net primary productivity and greenhouse gas emissions in annual grasslands. Ecological Applications 23: 46-59.

- Site: California
- Landcover Types: Grassland
- Climate Groups: C
- 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: 83%

A field experiment to track carbon storage on rangelands in California. We monetized this value 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 this value to temperate grasslands in the study area.

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

- Site: United States
- Landcover Types: Grassland
- 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%

Analyzed carbon sequestration on (grazed) rangelands in the United States. We monetized this value 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 this value on all grasslands in the study area.

Sengupta, S., Osgood, D. E. 2003. The Value of Remoteness: a hedonic estimation of ranchette prices. Ecological Economics 44(1): 91-103.

- Site: Arizona
- Landcover Types: Grassland
- Climate Groups: B
- Spatial Attribute: None
- Ecosystem Services: Aesthetic Information
- Valuation Methodology: Hedonic Pricing
- Sample Size: 8751
- Average Study Site Annual Household Income: $48,000
- Percent of Population Attaining High School Level Education or Greater: 90%

Estimated the value of the proximity to grasslands through a hedonic model of the sale of parcels assessed as rangeland in Yavapai County, Arizona (sample = 8,751). We chose this due to Arizona's proximity to LRR H. Because the climate of the original study site area is arid, we applied this value to arid grasslands in the study area. The median household income in the study area is similar to the study area, with slightly higher educational attainment rates.

Weber, M. A. 2007. Riparian Valuation in the Southwestern United States. University of Arizona.

- Site: Arizona
- Landcover Types: Shrubland
- Climate Groups: B
- Spatial Attribute: Riparian
- Ecosystem Services: Recreation & Tourism
- Valuation Methodology: Travel Cost
- Sample Size: 6069
- Average Study Site Annual Household Income: $47,000
- Percent of Population Attaining High School Level Education or Greater: 85%

Valued riparian resources in the Southwestern United States, using travel cost to estimate recreational benefits. The survey sample size was approximately 6,000. The median household income in the study site is similar, but educational attainment rates are lower than the study area.

Woodward, R., Wui, Y. 2001. The economic value of wetland services: a meta-analysis. Ecological Economics 37(2): 257-270.

- Site: Global
- Landcover Types: Wetlands
- Climate Groups: B, D
- Spatial Attribute: Riparian
- Ecosystem Services: Disaster Risk Reduction, Habitat, Recreation & Tourism, Water Quality
- Valuation Methodology: Meta Analysis
- Sample Size: 39
- Average Study Site Annual Household Income: N/A
- Percent of Population Attaining High School Level Education or Greater: N/A

A meta-analysis of multiple ecosystem services produced by wetlands, worldwide. We selected we selected values for habitat, recreation, and water quality from this study where no alternatives estimates were identified. We applied these to arid and continental wetlands—again, only where no other applicable values were found.

Zhongwei, L. 2006. Water Quality Simulation and Economic Valuation of Riparian Land use Changes. University of Cincinnati.

- Site: Ohio
- Landcover Types: Grassland, Wetland
- Climate Groups: C
- Spatial Attribute: Riparian
- Ecosystem Services: Water Quality
- Valuation Methodology: Avoided Cost
- Sample Size: Not Reported
- Average Study Site Annual Household Income: $58,000
- Percent of Population Attaining High School Level Education or Greater: 90%

Determined the value of nitrogen removal by wetlands and riparian grasslands bordering agricultural lands, based on avoided costs. We applied these estimates to temperate riparian grasslands and wetlands within the study area. The study area has slightly higher median household income than the study area, with similar educational attainment rates.

8. APPENDIX C

REFERENCES USED TO DETERMINE PRACTICE EFFECTIVENESS

Afinowicz, J.D., Munster, C.L., & Wilcox, B.P. 2005. Modeling Effects of Brush Management on the Rangeland Water Budget: Edwards Plateau, Texas. JAWRA Journal of the American Water Resources Association, 41(1), 181-193.

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Hydrologic Function
- Practice Assessed: 314

Archer, S.R. and Predick, K.I. 2014. An Ecosystem Services Perspective on Brush Management: Research Priorities for Competing Land use Objectives. Journal of Ecology 102: 1394–1407.

- Study Location: United States
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 314

Briske, D.D., editor. 2011. Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps. United States Department of Agriculture, Natural Resources Conservation Service.

- Study Location: United States
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 314

Derner, J. D., T. W. Boutton, and D. D. Briske. 2006. Grazing and Ecosystem Carbon Storage in the North American Great Plains. Plant and Soil 280:77-90.

- Study Location: Konza Prairie
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 528

Heitschmidt, R. K., S. L. Dowhower, W. E. Pinchak, and S. K. Canon. 1989. Effects of Stocking Rate on Quantity and Quality of Available Forage in a Southern Mixed Prairie. Journal of Range Management 42:468-473.

- Study Location: Texas
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 528

Knight, R.W., Blackburn, W.H., Scifres, C.J. 1983. Infiltration Rates and Sediment Production Following Herbicide/Fire Brush Treatments. Journal of Range Management, 154-157.

- Study Location: Texas
- Land Use: Rangeland
- Health Attribute: Biotic Integrity, Hydrologic Function, Soil and Site Stability
- Practice Assessed: 314

McGinty, W. A., F. E. Smeins, and L. B. Merrill 1979. Influence of Soil, Vegetation, and Grazing Management on Infiltration Rate and Sediment Production of Edwards Plateau Rangeland. Journal of Range Management 32:33-37.

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Hydrologic Function, Soil and Site Stability
- Practice Assessed: 528

McIlvain, E. H., and D. A. Savage. 1951. Eight-year Comparisons of Continuous and Rotational Grazing on the Southern Plains Experimental Range. Journal of Range Management 4:42?47.

- Study Location: Oklahoma
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 528

Reardon, P. O., and L. B. Merrill. 1976. Vegetation Response Under Various Grazing Management Systems in the Edwards Plateau of Texas. Journal of Range Management 29:195-198.

- Study Location: Texas
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 528

Rogler, G. A. 1951. A Twenty-Five Year Comparison of Continuous and Rotation Grazing in the Northern Plains. Journal of Range Management 4:35?41.

- Study Location: North Dakota
- Land Use: Rangeland
- Health Attribute: Biotic Integrity
- Practice Assessed: 528

Teague, W.R., Dowhower, S.L., Baker, S.A., Haile, N., DeLaune, P.B., Conover, D.M. 2011. Grazing Management Impacts on Vegetation, Soil Biota and Soil Chemical, Physical and Hydrological Properties in Tall Grass Prairie. Agriculture, Ecosystems, and Environment 141(3-4): 310-322.

- Study Location: Texas
- Land Use: Rangeland
- Health Attribute: Biotic Integrity, Hydrologic Function, Soil and Site Stability
- Practice Assessed: 528

Tuppad, P., Santhi, C., Wang, X., Williams, J.R., Srinivasan, R., Gowda, P.H. 2010. Simulation of conservation practices using the APEX model. Applied engineering in agriculture 26(5): 779-794.

- Study Location: Mill Creek Watershed, Texas
- Land Use: Rangeland
- Health Attribute: Hydrologic Function, Soil and Site Stability
- Practice Assessed: 314, 528

Warren, S.D., M.B. Nevill, W.H. Blackburn, and N.E. Garza. Soil Response to Trampling Under Intensive Rotation Grazing. Soil Sci. Soc. Am. J. 50:1336-1341

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Soil and Site Stability
- Practice Assessed: 528

Wood, M.K., Blackburn, W.H. 1981. Grazing Systems and Their Influence on Infiltration Rates in the Rolling Plains of Texas. Journal of Range Management 34: 331-335.

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Biotic Integrity, Hydrologic Function, Soil and Site Stability
- Practice Assessed: 528

Wood, M.K., Blackburn, W.H. 1981b. Sediment production as influenced by livestock grazing in the Texas rolling plains. Journal of Range Management 34:228-231.

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Soil and Site Stability
- Practice Assessed: 528

Wood, M.K., Blackburn, W.H. 1984. Vegetation and soil responses to cattle grazing systems in the Texas rolling plains. Journal of Range Management 37:303-308.

- Study Location: Edwards Plateau
- Land Use: Rangeland
- Health Attribute: Biotic Integrity, Soil and Site Stability
- Practice Assessed: 528

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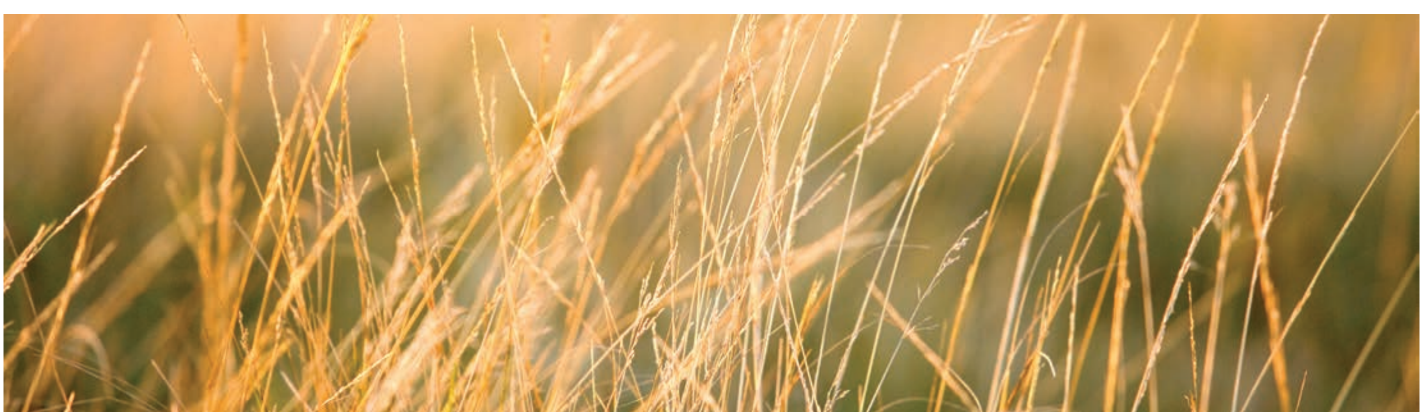

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