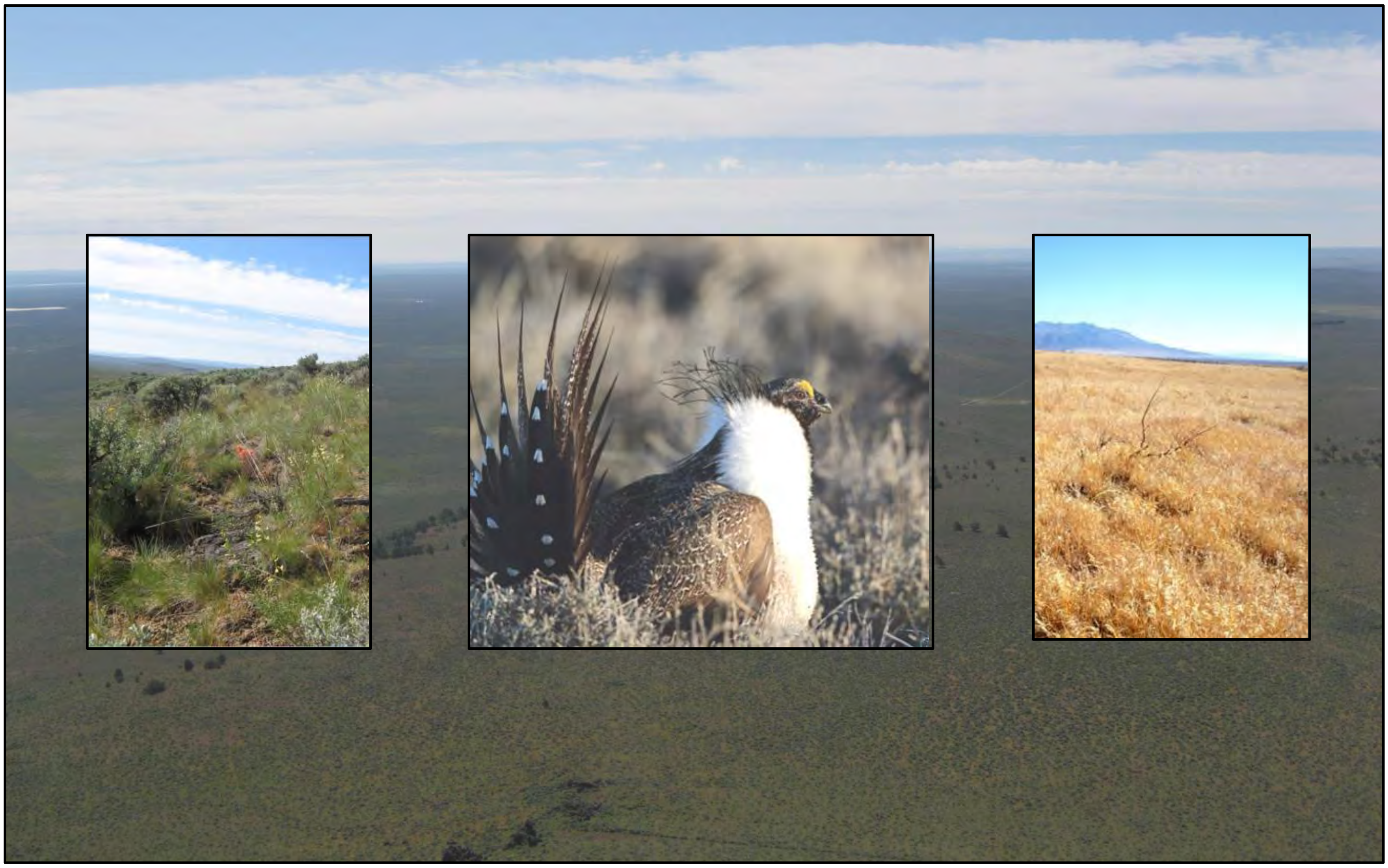

SAGE-GROUSE CONSERVATION

LINKING PRACTICES TO HABITAT METRICS

OUTCOMES & IMPACTS: EXECUTIVE SUMMARIES AND FULL REPORTS

USDA-NATURAL RESOURCES CONSERVATION INNOVATION GRANT

Table of Contents

Project Summary

This document serves as a final report in the fulfillment of USDA Award number 69-3A75-13-212. The NRCS funded program brought together a team of researchers, land managers and ranchers to develop new tools for more effective and efficient conservation and restoration of sagebrush steppe habitat in the western sagebrush steppe.

This project developed land manager decision support products to provide guidance for implementing management practices based on threats. Land managers seeking to improve sage-grouse habitat can benefit from implementing science-based management practices in sage-steppe ecosystems.

However, implementation can be challenging as plant community responses to management practices are dependent on a complex combination of factors, including soils, microclimates, invasive species, fire regimes, current habitat state, historical impacts, and more. These complexities make it difficult to interpret and apply scientific research for appropriate management practices.

We formed the Sage-Steppe Habitat Response (Sage-SHARE) working group to create work products for land managers which address these complex issues.

Resulting products include:

- An exploration of the relationship between ecological site descriptions (ESDs) and environmental gradients utilizing a case study of Harney County, Oregon to determine potential logical groupings based on environment,
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- A searchable practices database with manual,
- A manager's guide to simple threat-based models in the western sagebrush steppe,
- A comparison of current vegetation mapping tools and metrics, and
- A manager's guide to the practices database, which includes an analysis of the distribution of scientific articles by practice, elevation band, and precipitation zone. This guide also includes summaries for each individual practice (fire, grazing, seeding, mechanical, and herbicide).

Project Partners & Roles

The efforts which led to this document were funded in part by a Conservation Innovation Grant (CIG) from USDA-Natural Resources Conservation Service to The Nature Conservancy of Oregon and the Eastern Oregon Agricultural Research Center which is jointly operated by USDA-Agricultural Research Service and Oregon State University.

Many institutions and individuals contributed to the development of this document. Authors and groups of authors wrote individual sections and many individuals reviewed the document and provided comments.

This guide was developed to provide support for more effective and efficient conservation and restoration of western sagebrush steppe. With increasing regulatory emphasis on sage-grouse conservation it is critical for land managers to apply best management practices based on sound research.

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Introduction

This project was developed out of a critical need for ecosystem-based decision tools to assist land managers working in sage-grouse habitat in the western sagebrush steppe. We faced a challenge in providing a general and flexible framework that will assist in making management decisions and communicating the logic and science behind best management practices.

There are several issues that make Greater Sage-Grouse (GRSG) conservation unique:

- The broad range of the species across the western U.S.,
- The need for large expanses of intact habitat, and
- The number and diversity of stakeholders involved in conservation planning and implementation.

Relative to other ecosystems, research in the sagebrush steppe has been limited. There is a clear need for more ecological information on the sagebrush steppe, but there is also a need to assemble what we have learned as this region is now under intense scrutiny because of the U.S. Fish and Wildlife Service (USFWS) reviewing the status of GRSG for protection under the Endangered Species Act (ESA). This has resulted in the extensive commitment by public and private stakeholders to maintain and improve habitat.

Sage-grouse population declines are strongly tied to complex ecological problems at landscape scales. Furthermore, the western portion of the sagebrush steppe contains complicated land ownership patterns and many entities involved in land management. This combination means there is no single answer for every problem. It is our goal that the products of this report help land managers make decisions to improve complex sagebrush steppe habitat with a more simplified approach.

This report includes the work completed for the CIG grant. Pages 1-12 provide an overview of these efforts, followed by detailed reports of each deliverable. The products resulting from this project are meant to be complimentary to existing land management tools.

Overall Project Objectives

- Merge and refine threat-based models according to similarity in response to climate, management actions, and threats for the major ecological sites within the western sagebrush steppe;

- Synthesize current management practice literature in sagebrush steppe ecosystems regarding the effectiveness and benefits of key conservation practices for sagebrush steppe ecosystems;

- Evaluate metrics and mapping tools associated with threat-based models to measure plant community response to management practices that improve GRSG habitat; and

- Develop products for land managers who will be applying management practices to sagebrush steppe ecosystems.

Click here to watch a brief video summarizing the Sage-SHARE project!

Overall Impacts

One key finding of the project was the evidence that adaptive management is necessary pertaining to western sagebrush steppe conservation and restoration at the ecosystem level. More research is always better for decision making, but we have to realize that this ecosystem is extremely complicated and there is not a universal answer to most questions. This project—and more specifically the database—point to good starting platforms for making decisions. However, they need to be expanded to better fit into the realities facing rangeland managers today—science provides a starting point, but not an end point.

Threat-based Models

The revisions of the threat-based models over time and their incorporation into a mental model and structured decision making (SDM) system allows for a simple and clear strategy for GRSG habitat conservation while looking at the sagebrush steppe ecosystem from a broader perspective.

Workshops

A main theme that emerged from the field workshop was the need to conceptually incorporate the threat-based models and database approaches into other tools already in use. A second workshop focused on demonstrating the utility of the database in a land management context.

Knowledge Gaps

The database was continuously updated as applicable citations were identified. Greater attention was focused on the sagebrush conservation practices that appeared to lack references, and will continue to be updated as new scientific literature and projects are completed. In general, a lack of lower elevation (<4000 ft) studies was identified, particularly for mechanical and grazing treatments. Priorities and recommendations for future work are noted throughout the manager's guide to rangeland practices.

Lessons Learned

There are many different habitat management visions and concepts used by federal, state, and private land managers. Simple mental models and conceptual frameworks are needed as a "front-end" to the discussion of broad-scale landscape management. A common vision of vegetation dynamics is a critical starting point. There are often gaps in the cycle of conceptualizing, planning, implementing, and adapting for conservation purposes. Involvement of stakeholders is essential when creating and testing new planning and implementation tools—input and transparency are key elements to their success.

Report Structure

The first section is an executive summary containing brief descriptions of objectives, methods, outcomes, and impacts for each deliverable. Full reports follow the executive summary. Links to each full report can be found both in the Table of Contents and each respective summary. Reports begin with an analysis of ecological site descriptions and the need for simple threat-based models in Harney County, Oregon. A description of the literature database follows along with a manager's guide on applying threat-based models to the western sagebrush steppe, an accuracy assessment comparing remotely sensed data to field sampling on the ground, and a second manager's guide to practices based on a review of the literature collected in the database.

.Project Deliverables Summaries

1) Relationship between Ecological Site Descriptions and Western Sage-Steppe Threat-Based Models

(Full ESD report on pp. 13-15)

Objective

Test the hypothesis that ecological site descriptions (ESDs) can be grouped using available environmental data that are predictive of key habitat condition threats (e.g., soil texture, elevation, precipitation zone) using Harney County, Oregon as a case study.

Methods

First, data pertaining to environmental conditions covering Harney County were georeferenced and assembled from various sources (predominantly ILAP—Integrated Landscape Assessment Project and LEMMA—Landscape Ecology, Modeling, Mapping, and Analysis) and represented environmental conditions with respect to soil type, topographic position, temperature, moisture, elevation, and variability in temperature and moisture regime. Data were sampled based on points within Map Unit Keys (MUKEYs), which were determined to represent a geographic area within an ESD.

Within each ESD there is a description of physiographic, climatic, water, soil, and plant community features. Each ESD contains a state-and-transition model (STM) intended to represent potential plant communities and causes of transition from one community to another. Within the NRCS land classification system, Major Land Resource Areas (MLRAs) describe larger areas and ESDs are assigned within each MLRA with areas ranging from less than one thousand to over two million acres. In the western sagebrush steppe, the MLRAs range from eight to eighteen million acres in size.

There were 78 ESDs within the Harney County dataset and ESDs contained between 1 and 19 MUKEYs; of the 78 ESDs, 53 contained more than one MUKEY. This data was then used to run a canonical variate analysis (CVA). In this ordination-based analysis, environmental variables were used to predict membership of samples (MUKEYs) to a particular ESD. Put another way, this analysis indicates the percent variation in ESD membership explained by the sampled environmental variables.

Outcomes

The CVA results suggested that all environmental variables taken together explained about 14.5% of the variation in ESD membership among samples. This is probably an overestimate of the ability of environmental variables to explain ESD membership because 25 of the 78 ESDs had only one sample, which would have resulted in no variability in environmental variable scores for those ESDs.

The five most impactful environmental variables explained about 34% of that variation; those variables included August maximum temperature, the difference between August maximum and December minimum temperatures, slope, mean annual temperature, and percent sand content in the soil. These same variables did a poor job in assigning samples to their correct ESD (i.e., environmental variables explained less than 15% of the variation in ESD membership). We believe this poor fit was reflective of strong within ESD variation in environmental properties. In other words, multiple samples of the same ESD did not display similar values for environmental variables. This analysis suggests that assembling threat-based models based on ESD membership would be somewhat at odds with dominant environmental gradients, and further supports the idea of using dominant gradients of temperature and moisture, as encapsulated by elevation, to determine appropriate (broader) threat-based models.

Conclusions

While ESDs can be utilized successfully at finer scales, it was determined that habitat conditions modeled based on threat could be useful on a larger MLRA scale (see Figure 1 for project area).

Using the STMs and professional experience, the major ESDs within each MLRA can be grouped into the following threat categories: 1) annual grass only, 2) annual grass and conifer, or 3) conifer only (Figure 2). Most of the ESDs in (Figure 2) are dominated by either Wyoming or Mountain big sagebrush, the exception being the two annual grass threat ESDs in MLRA 25. This approach provides the more detailed information contained in an ESD, but retains the simplicity of a threat-based mental model and enables land management decisions to occur on a broader level. It is also consistent with resistance and resilience concepts used in western sagebrush steppe to reduce threats to habitat.

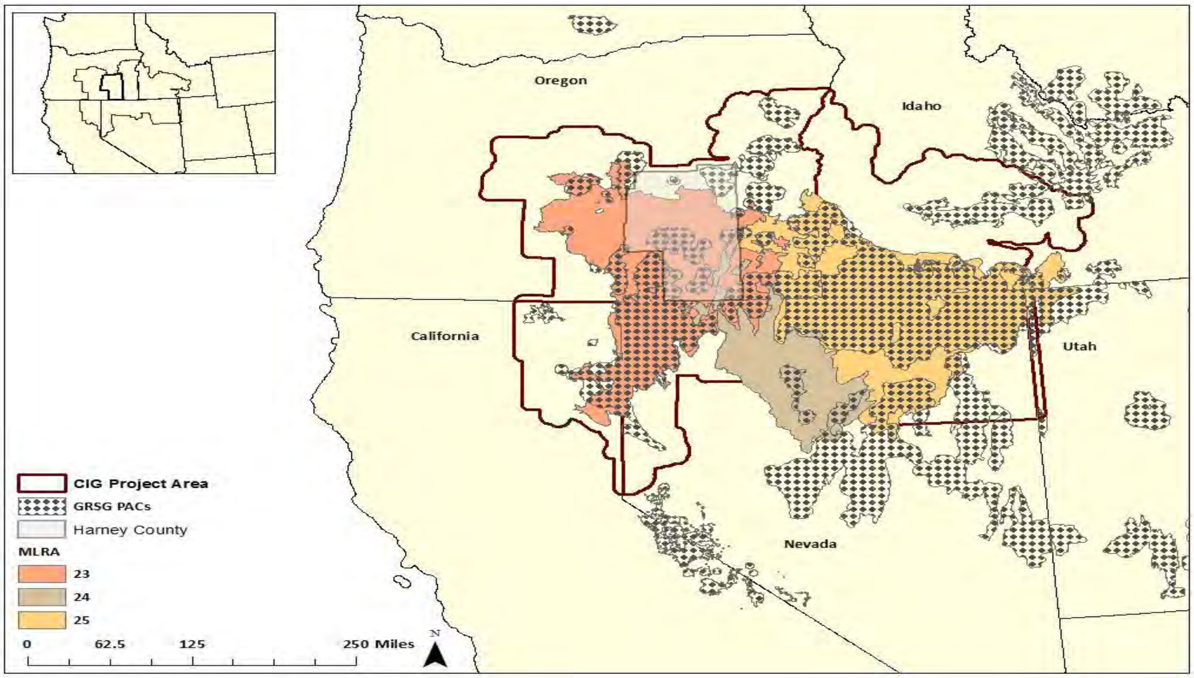

Figure 1. Project area showing GRSG Priority Areas of Conservation (PACs), and MLRAs 23, 24, and 25 with Harney County (OR) outlined in red.

Relavent Ecological Threats			
	Annual Grass Threat		
		Juniper Encroachment Threat	
Major Land Resource Areas (MLRAs) and Example Ecological Site Descriptions (ESDs)			
MLRA 23	**Loamy 10-12** (023xy212 OR) 1,235,938 acres	**Loamy 12-16** (023xy318 OR) 140,739 acres	**Shallow Loam 16-2** (023xy501 OR) 43,636 acres
MLRA 24	**Loamy 8-10** (024xy005 NV) 1,013,062 acres	**Loamy Slope 12-14** (024xy021 NV) 219,193 acres	**No conifer-only ESDs** *Dr. Tamzen Stringham, pers. comm.*
MLRA 25	**Claypan 10-12** (025xy018 NV) 404,724 acres	**Loamy 8-10** (025xy019 NV) 2,554,757 acres	**Loamy Slope 16+** (025xy004 NV) 133,319 acres
	Increasing productivity and site potential		

Figure 2. A comparison between the ecological threat models and examples of Ecological Site Descriptions (ESD) from selected Major Land Resource Areas within the project area (see Figure 1 above). ESD descriptions include name, unique identifier, and acreage. Example ESDs are arranged in order of increasing productivity and site potential.

2) Database for Sagebrush Steppe Conservation Practices

(User's Guide on pp. 16-22)

Objectives

Summarize existing literature on the effectiveness and benefits of key management practices in the western sagebrush steppe. Plant community responses to different practices depend on numerous factors including soil type, microclimate, invasive species, fire or other disturbance regimes, and current habitat state. To evaluate the effectiveness of key rangeland management practices on sites in varying ecological states, existing literature on sagebrush steppe and sage-grouse habitat in and around the western sagebrush steppe was collected and entered into a relational Microsoft Access database called Sage-Steppe Habitat Response (Sage-SHARE). The results taken from this extensive literature review can be used to inform management decisions at different spatial scales while identifying knowledge gaps where further research is needed.

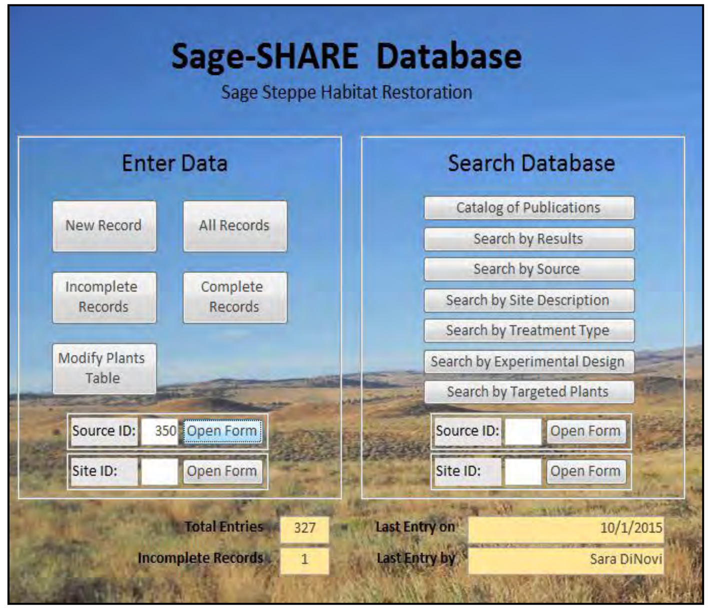

Figure 3. Sage-SHARE database main screen.

Methods

Sage-SHARE is broken up into two set parts or built functions; one for data entry and another for data queries. Each study entered into the database has fields to populate within site description, experimental design, and results. Within the site description, fields include five key rangeland management practices: Prescribed or wild fire, prescribed grazing, rangeland seeding, mechanical treatments, and herbicide application. No data interpretation was made while entering sources. Simple, built-in queries can be run from the main page such as filtering studies by targeted plant species, elevation, or desired result (Figure 3). However, more complex queries were necessary and designed to more efficiently analyze the data. The data entered into Sage-SHARE were first catalogued on "MyEndnoteWeb", which allows for a versatile and license-free mechanism from which to manage the literature library. Microsoft Access® 2007 or newer is required to support opening or editing the database.

Outcomes

The comprehensive literature review was captured in a relational database (Figure 4), which was created in Microsoft Access® 2013. Jessica Lambright, a volunteer with The Nature Conservancy, designed and constructed the database with team input. A complimentary bibliographic catalogue was created via EndNote Web.

Conclusions

The database has the capability for additional relevant research and expert knowledge to be added, and is searchable to determine best management practices by multiple variables including ecological sites. To complete a query in the Sage-SHARE database, one can use the main form or create one's own through the query wizard. From the main form there are several choices to narrow down data by results, target species, and treatment type.

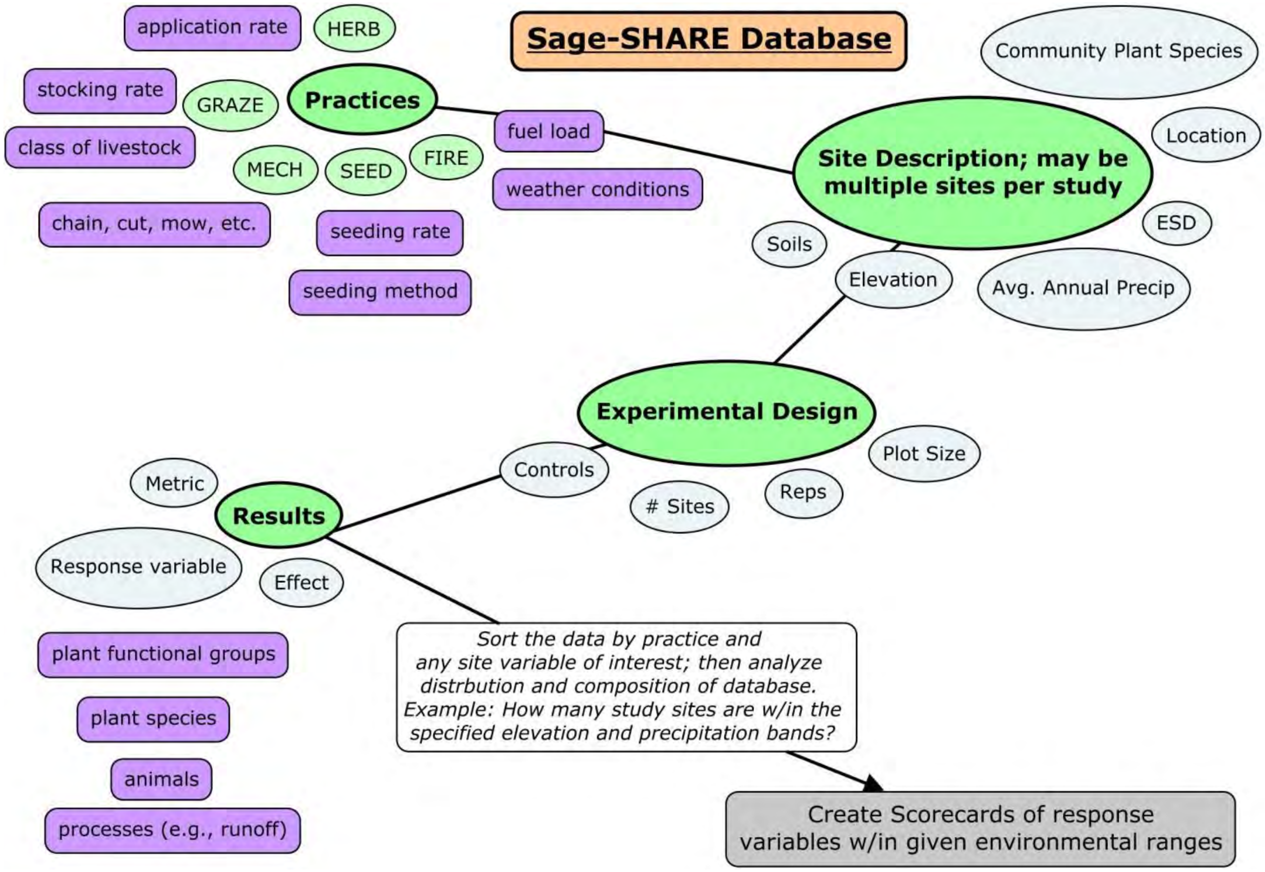

Figure 4. Conceptual map of the Sage-SHARE relational database

3) Manager Guide 1: Applying Threat-based Mental Models to Greater Sage-Grouse Conservation

(Full report on pp. 23-64)

Objectives

Explore and develop simple mental models as a tool that can accommodate many stakeholder values and serve as a bridge between ecological complexity and decision-making to help accomplish large scale conservation. The combination of scale required for success, the resultant involvement of many stakeholders, and complex land ownership patterns makes GRSG conservation perhaps the largest and most complex effort ever attempted in association with the Endangered Species Act (ESA).

Methods

A reasonable place to start in the development of mental models for GRSG conservation is with the major threats to sage-grouse habitat: conifer expansion and exotic annual grass invasion, which influence 33 of the 39 major GRSG populations in the western sagebrush steppe. The approach involves three sequential steps: 1) identify the primary threats, 2) develop simple mental models that incorporate the threats into habitat (or vegetation) dynamics, and 3) apply structured decision making (SDM) for evaluating management alternatives and best use of individual practices to address the threats.

SDM can be a rigorous, transparent, and interactive approach to conservation that involves stakeholders, the basic elements of which are 1) define clear, quantifiable objectives and constraints relative to the problem; 2) identify potential management actions; 3) evaluate the potential effects of management actions as they relate to initial objectives; 4) address uncertainty; and 5) assess trade-offs and select a decision.

Outcomes

The primary threats identified are a shift in plant community dominance to: 1) invasive annual grasses only, 2) invasive annual grasses and conifer, or 3) conifer only. Generally, invasive annual grasses are more of a threat at low to mid elevation sites, and conifers are a threat at mid to high elevation sites. In the case of the threat-based model, the clear objectives should be to reduce threats (annual grasses, conifers, or both). Figure 5 shows the link between threat-based models (purple text) and SDM (steps are red).

Conclusions

There has already been an effort of this sort for GRSG conservation. The process of developing mental models and management practices to address GRSG habitat threats took place during 2011 to 2013 in Harney County, OR and involved stakeholders from a wide variety of perspectives resulting in the Candidate Conservation Agreement with Assurances (CCAA) program. The CCAA effort was a great first step in developing a mental model-based approach to GRSG conservation, but requires additional steps such as the development of a formal decision-making process like SDM, a means of evaluating the effects of management practices, and an adaptive system.

The combination of a threat-based mental model and SDM provides the pertinent detail necessary to develop an overall strategy for GRSG habitat conservation in the western sagebrush steppe while streamlining the process to allow for navigation of complex scenarios.

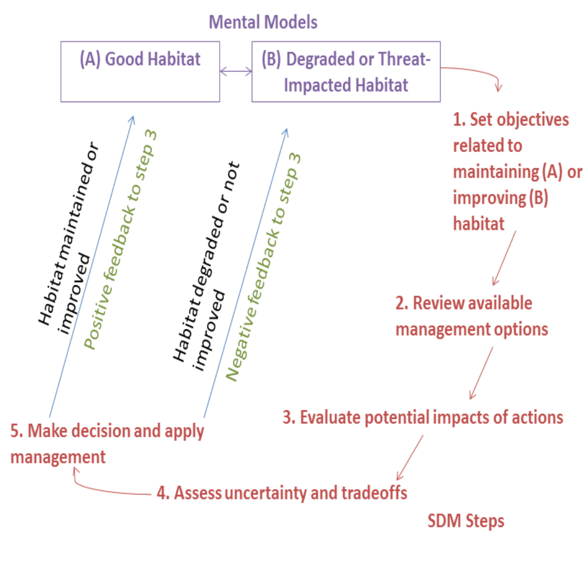

Figure 5. Diagram linking mental model structured decision making (SDM) processes that integrates adaptive management.

4) Workshop in Sage-Grouse Habitat: Planning to Practices

Objectives

Provide field-based training using habitat models and the associated manager guides developed as part of this project to gain greater consistency across agencies for assessing habitat and applying effective conservation practices at the landscape scale. A second objective was to refine the models and approach to assessing various management practices.

Methods

Workshop attendees participated in field-based activities designed to consider landscape scale management. Threat-based models were used as a framework for all learning presentations. The workshop was held at the 16,000 acre USDA Northern Great Basin Experimental Range, near Riley, OR. The program consisted of short presentations in a conference room followed by discussions on field concepts as they apply to landscape-scale management.

Instructors included: Tony Svejcar and Chad Boyd, USDA Agricultural Research Service; Jay Kerby, The Nature Conservancy; Dustin Johnson and David Bohnert, OSU Extension-Eastern Oregon Ag. Research Center; and Angela Sitz and Jackie Cupples, USFWS.

Outcomes

48 workshop participants registered representing the Bureau of Land Management (BLM), NRCS, USFWS, Idaho Dept. of Lands, Idaho and Oregon Department of Fish and Wildlife, Soil and Water Conservation Districts, conservation organizations and private companies over three states in the western sagebrush steppe. During the workshop, participants did the following:

- Discussed concepts regarding the application of assessing and applying conservation practices using habitat models.
- Worked to classify landscapes and address spatial issues in planning efforts with a separate session on assessing apparent trend using a point-in-time analysis in conjunction with trend indicators.
- Reported an increase in knowledge of plant community change resulting from disturbances.
- Gained experience in developing site specific plans or the planning process.

Selected comments from workshop participants:

"The practical experience of running through an assessment with a diverse group of people was very beneficial."

"Hearing about how other groups made decisions was equally valuable."

"Thank you, I feel like I learned a lot and can take this back and apply it to the information that is currently being used."

"Practical and easy method to use for assessment on a landscape scale."

"Getting out and being able to see the concept being applied and multiple agency input and perspectives."

Table 1 presents workshop evaluation questions and responses.

Conclusions

BLM staff requested additional training opportunities from the Sage-SHARE team in 2017 as BLM participants found the workshop and associated materials useful in their current work. Additionally, the planners involved in developing site specific plans for the Candidate Conservation Agreements with Assurances have also requested materials produced from this Sage-SHARE effort.

Breakout sessions collected participants' expert opinions for refinement of the decision support products; sessions resulted in a dialogue surrounding ideas such as mapping techniques and integrating the threat-based models with other habitat assessment tools (which were later incorporated into revised versions of the products).

Table 1. Evaluation Questions and Responses Presented at the Conclusion of the Workshop.

Workshop Evaluation Questions	Participants who "agree" or "strongly agree" (%)
The information I learned is very practical.	100
I will refer back to the material for career work.	97
The course materials were good quality.	100
Program provided a logical link between initial land assessment and subsequent management.	97
Instructors were prepared.	97
Instructors had good knowledge on the subject matter.	97
Mix of classroom and field settings were conducive to learning the subject matter.	97
I will be able to apply what I have learned to the lands I manage.	100
The workshop and guide will help improve my decision making for sage grouse habitat management.	97

5) Workshop on Utilizing the Sage-SHARE Database

Objectives

Introduce land managers to the database functions and resources. More specifically, to teach land management planners and implementers to utilize the database as a resource, and create input forms for building on current research and past land management projects.

Methods

Workshop attendees were trained in how the database was structured and how to use its designated functions in considering land management. The database user's guide was utilized as well as the database itself as teaching tools. The workshop was held at the Burns, OR BLM District office with remote communications for consumers outside of the BLM Burns District. Facilities enabled for short presentations in a conference room followed by discussion of concepts and data analysis as they apply to landscape scale management.

Instructors included Tony Svejcar, Dustin Johnson, and Sara Holman, OSU; and Chad Boyd, USDA Agricultural Research Service.

Outcomes

The following thoughts were captured from potential end-users:

- A simpler database and query system is required.
- Needs to be easily accessible for updating.
- Could be a good way to capture institutional knowledge.
- A separate employee or intern may be required to organize and enter past project information.
- Could try to link to existing databases (i.e., the USFWS/USGS Conservation Efforts Database and Land Treatment Digital Library).

Ideas to include for making the database more useful:

- There may need to be a way to categorize level of data robustness (i.e., trend indicators vs. large data sample).
- A time sensitive field should be included to flag when a project needs a monitoring action.
- Need to be able to identify successes vs. failures and indicate why.
- There should be a temporal component to indicate changes over time.

Conclusions

The combination of current monitoring protocol and information from the database can contribute to more informed decisions regarding sagebrush ecosystems. The ability to quickly filter through past projects will enable land managers to make more informed decisions while planning, implementing, and assessing future projects.

6) Comparison of Habitat Condition Mapping Methods and Products: Vegetation Mapping Accuracy Assessment

(Full report on pp. 65-89)

Objectives

Compare results of three remote sensing platforms to ground-based data to determine extent of agreement of remotely-sensed data with ground-based data.

Methods

Working in conjunction with federal and state collaborators, The Nature Conservancy (TNC) used remotely-sensed vegetation coverages to reclassify data into vegetation states at the 30 m scale for a 50,000 acre study area in southeast Oregon. The study area was comprised of sagebrush rangeland experiencing ecologically-based threats including exotic annual grasses and expanding conifer. Data layers of cover of major vegetation functional groups in the study area were generated by three different remote sensing/data manipulation techniques originating from Open Range Consulting (ORC), U.S. Geological Survey (USGS), and Institute for Natural Resources (INR). These values were then used to classify 30 meter pixels based on threats present (annual grass, annual grass + conifer, or conifer) and habitat condition using parameters taken from the draft Oregon GRSG Habitat Quantification Tool (HQT) Scientific Methods (see Guide 1 Appendix 3 on page 4.36 or accuracy assessment Appendix 1 on page 5.24 for specific parameters). Steps were taken to classify all datasets from all providers into identical bins with identical class values to make comparisons of similarity and dissimilarity.

MARXAN optimization software was used to find field plots via an algorithm searching for the lowest 'cost' set solution that will meet 'goals' for all 'targets'. In the end 149 plots were identified for accuracy assessment within the herbaceous and shrub dominated areas as mapped by the three methods. The datasets were then compared for accuracy in estimating tree cover, shrub cover, annual to perennial grass cover ratio, and perennial grass cover on the plots.

Outcomes

Tree cover plot estimates *(USGS data did not include tree cover)*:

- ORC correctly classified 89.6% of plots
- INR correctly classified 87.3% of plots

Shrub cover plot estimates:

- ORC—64.2%
- USGS—65.3%
- INR—62.4%

Perennial grass cover plot estimates:

- ORC—53%
- USGS—68%
- INR—57%

None of the methods effectively detected the annual grass to perennial grass ratio.

Conclusions

Our work suggests that there exist both opportunities as well as challenges associated with the use of remote sensing data to classify habitat conditions in large landscapes. For example, ORC has since used the findings of this report to improve their data collection methods and accuracy. Efforts invested in field verification for quality control could be prioritized towards areas of the landscape where remotely sensed data is less effective at detecting habitat condition.

7) Manager Guide 2: Rangeland Practices in the Western Sagebrush Steppe: Published Scientific Literature

(Full report on pp. 90-145)

Objectives

Assess the impacts of various conservation practices with a focus on prescribed and wildfire, prescribed grazing, rangeland seeding, mechanical treatments (e.g., mowing, chaining, or cutting) and herbicide application. Utilize the threat-based mental models and the structured decision making (SDM) process presented in Guide 1.

Methods

Information entered into the Sage-SHARE database was interpreted using analysis tools in Microsoft Excel and Access, and culminated in the scorecards (Table 3A-E) in addition to written summaries based on each conservation practice. Various tools included Sort and Filter, Pivot Tables, Query Design, and LOOKUP functions. Data were broken down based on elevation and precipitation zone for each practice. Data were also categorized into functional groups (e.g., annual grass, perennial grass, shrub, forb, and tree) for analysis. In-depth literature reviews were also conducted.

Outcomes

The practices guide synthesizes over 300 articles entered into the database while pointing out knowledge gaps and recommendations for future research. The Sage-SHARE database allows us to group articles into categories based on variables such as precipitation, elevation, dominant plant species, or vegetation threat, and determine if conservation practices have similar effects across categories (simple problems) or must be applied within a specific area to be effective (complex problems).

Priority knowledge gaps include:

- An under-representation of all practices at <4,000 ft., with numbers for mechanical and grazing studies being particularly low;
- Annual grasses tended to be under-represented compared to other functional groups in the grazing scorecard;
- No matter how the studies are parsed, there are many more studies on fire relative to other practices; and
- The general order of practices by total number of study sites is fire> grazing > seeding > mechanical> herbicide.

Scorecards (Table 3A-E) yielded a majority of mixed effects as a result of multiple treatments and/or rates of applied treatments. Since the scorecards cannot therefore be used in a prescriptive manner, it is necessary to dive into the details of certain studies to determine the clear effects of a specific treatment on a specific functional group.

Impacts

It is difficult to complete a meta-analysis of rangeland research, especially when data are often taken, reported, and interpreted differently from one study to another. The complex nature of rangeland management and data collection and analysis demonstrates support for applied adaptive management methods and use of SDM/ mental models to help simplify solutions.

8) Factsheet Scorecards: Conservation Practices

Objectives *(More about scorecards on pp. 100-105)*

Create scorecards for each conservation practice using information from the relational database. Show effects of various management practices on five functional groups, and describe the expected results of pairing certain practices with each group depending on a site's elevation and average precipitation.

Methods

The Access database was built with the following effects for response variables: negative, positive, increase, decrease, mixed, and none. Any functional group effects marked as negative or decrease went into the "-" column; any marked as positive or increase went into the "+" column; none were classified with "0"; and mixed stayed as "mixed".

Using Excel tools, data were organized into low (<4000 ft), medium (4000-5500 ft), and high (>5500 ft) elevations as well as four precipitation zones loosely correlated with the mental threat-based models described in Guide 1. Final scorecards were then created using Pivot Tables.

Outcomes

There were over 1800 response variable entries. Table 2 shows a summary of the response variable counts. There were 256 sites in the database on which annual grasses (a. grass) were measured, 384 on which perennial grasses (p. grass) were measured, and 241, 306, and 37 on which forbs, shrubs, and trees were measured, respectively. The highlighted variables were used in scorecard evaluations. Functional group response (a. grass, p. grass, forb, shrub, and tree) was analyzed by elevation band and precipitation zone, but here we present the elevation scorecards (Table 3A-E).

Table 2. Response variables entered into the database and associated number of entries made.

Response Variable	#
A. Grass	256
P. Grass	384
Forb	241
Shrub	306
Tree	37
All plant	115
Ammonium	3
Animal/Insect/ Bird	84
Bare Ground	30
Fire	5
Forage	7
Grass	38
Herbaceous	36
Invasive	65
Litter	12
Model Prediction	1
Nutrient/Energy Exchange	51
Runoff	4
Seed/Seedling/ Collection	35
Soil/Biological	89
Standing Crop	6
Weed	22
Total	**1827**

Impacts

Our perception of positive or negative, or what we believe to be "+" or "-" may not apply to the scoring system used to develop the scorecards. For example, we would take a decrease in cheatgrass cover to be positive, but for purposes of entering data and for the scorecards, that would be entered as decrease and go in the "-" column. Boiling down all the information in the database to plus, minus, or zero may lead to confusion.

Some response variables were not included in the scorecards such as "all plant" and "grass" (as opposed to specified annual or perennial grasses). It is also important to note that multiple results could be entered for each study, so it's possible that redundancy occurred. For example, it is possible that in one study both "p. grass" and "bluebunch" cover were marked as increasing whereas in another more general study, only "p. grass" was entered. This could skew the scorecard values.

Table 3. Scorecards by Elevation (ft.) Indicating the Number of Occurrences in which Functional Groups Responded in a Given Manner for (A) Fire, (B) Grazing, (C) Seeding, (D) Mechanical, and (E) Herbicide.

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass	1		1	14	16
Forb	1		2	2	5
P. Grass		2		3	5
Shrub		4	2	8	14
Tree		2		1	3
<4000					
A. Grass		5	1	9	15
Forb	2	2	3	4	11
P. Grass		7	9	12	28
Shrub		8	1	7	16
Tree				2	2
4000-5500					
A. Grass	1	10	7	32	50
Forb	7	5	10	13	35
P. Grass	4	9	18	49	80
Shrub		11	5	25	41
Tree			1	6	7
>5500					
A. Grass		3	2	43	48
Forb	10	3	24	18	55
P. Grass	2	7	8	38	55
Shrub		18	7	45	70
Tree		2	1	13	16

(B)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass		1	1	3	5
Forb		2	3	2	7
P. Grass	2	2	1	6	11
Shrub				3	3
<4000					
A. Grass		1		1	2
Forb		1			1
P. Grass	2	3	2	1	8
Shrub			2	1	3
4000-5500					
A. Grass	2	3	2	9	16
Forb	2	3	3	12	20
P. Grass	10	5	21	11	47
Shrub	2	3	23	7	35
Tree				1	1
>5500					
A. Grass		3	1		4
Forb	2	1	1	13	17
P. Grass	8	1	10	6	25
Shrub	3	3	15	7	28

(C)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass	1				1
Forb				1	1
P. Grass		1		1	2
Shrub			2	4	6
<4000					
A. Grass		3	1	7	11
Forb	2		2	2	6
P. Grass			7	14	21
Shrub		1	2	2	5
4000-5500					
A. Grass		1	6	14	21
Forb			1	7	8
P. Grass	2	5	8	30	45
Shrub	2	4		10	16
Tree				1	1
>5500					
A. Grass		1	3	6	10
Forb	1	5		9	15
P. Grass	2	2	4	20	28
Shrub	3	6		11	20
Tree			5		5

(D)

Elevation	0	-	+	Mixed	Total
Unknown					
Forb			2		2
P. Grass			2	1	3
Shrub		5		8	13
Tree		1			1
<4000					
A. Grass		2		4	6
Forb		2	1	2	5
P. Grass		2		3	5
Shrub		2	1	6	9
Tree				2	2
4000-5500					
A. Grass	1	3	7	18	29
Forb	5	2	12	19	38
P. Grass	3	7	7	28	45
Shrub		11	4	19	34
Tree				5	5
>5500					
A. Grass	3	3	2	22	30
Forb	3	9	12	14	38
P. Grass		5	4	27	36
Shrub		10	7	36	53
Tree		2	6	9	17

(E)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass				1	1
Forb			2	1	3
P. Grass			1	1	2
Shrub	1	3		5	9
Tree				1	1
<4000					
A. Grass		12		11	23
Forb		5		11	16
P. Grass		2		7	9
Shrub		2	3	1	6
4000-5500					
A. Grass		4	2	23	29
Forb	1	4	1	12	18
P. Grass	2	5	1	11	19
Shrub	1	5	1	5	12
>5500					
A. Grass		2		23	25
Forb		2			2
P. Grass		2	2	6	10
Shrub	1	9		2	12

Relationship between ESDs and Environmental Gradients in Harney County

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Objectives

Test the hypothesis that ecological site descriptions can be grouped using available environmental data that are predictive of key habitat condition threats (e.g., soil texture, elevation, precipitation zone). One way to answer the question is to sample environmental conditions within a population of ESDs and a) describe major environmental gradients, and b) determine how well the sampled environmental variables predict ESD membership.

Methods

To test these relationships, a dataset of environmental conditions covering Harney County was assembled. The georeferenced data were assembled from various sources (predominantly ILAP—Integrated Landscape Assessment Project, and LEMMA—Landscape Ecology, Modeling, Mapping, and Analysis) and represented environmental conditions with respect to soil type, topographic position, temperature, moisture, elevation, and variability in temperature/moisture conditions.

Data were sampled based on points within Map Unit Keys (MUKEYs); for our purposes, MUKEYs represent a geographic area within an ESD. There were 78 ESDs within the dataset and ESDs contained from 1 - 19 MUKEYs; of the 78 ESDs, 53 contained more than one MUKEY. We then used the MUKEYs as the experimental unit to run a canonical variate analysis (CVA). In this ordination-based analysis, we used environmental variables to predict membership of samples (MUKEYs) in a particular ESD. Put another way, the analysis indicates the percent of variation in ESD membership explained by sampled environmental variables.

Results

The CVA analysis is best discussed in pieces. The first piece of the analysis was to arrange environmental variables in ordination space (i.e., describe dominant environmental gradients). The result of this piece is seen in Figure 1, where arrows indicate the direction and magnitude of effect (synonymous with arrow length) of individual variables along dominant (x-axis) and sub-dominate (y-axis) gradients.

The second piece of the analysis was to determine the position of ESDs within ordination space; in this case, ordination space is defined by the gradients of environmental variables. Locations of ESDs in ordination space are displayed in Figure 2. In this figure, only the centroids of ESDs are displayed (i.e., as opposed to displaying the multiple MUKEYs that were used to define that point) in order to reduce the number of points and increase graphical clarity. The third piece of the analysis was to determine how much of the variation in ESD membership among samples is predictable based on scores for environmental variables. To that end, the CVA results suggest that all environmental variables when taken together explain about 14.5% of the variation in ESD membership among samples. This is probably an overestimate of the ability of environmental variables to explain ESD membership

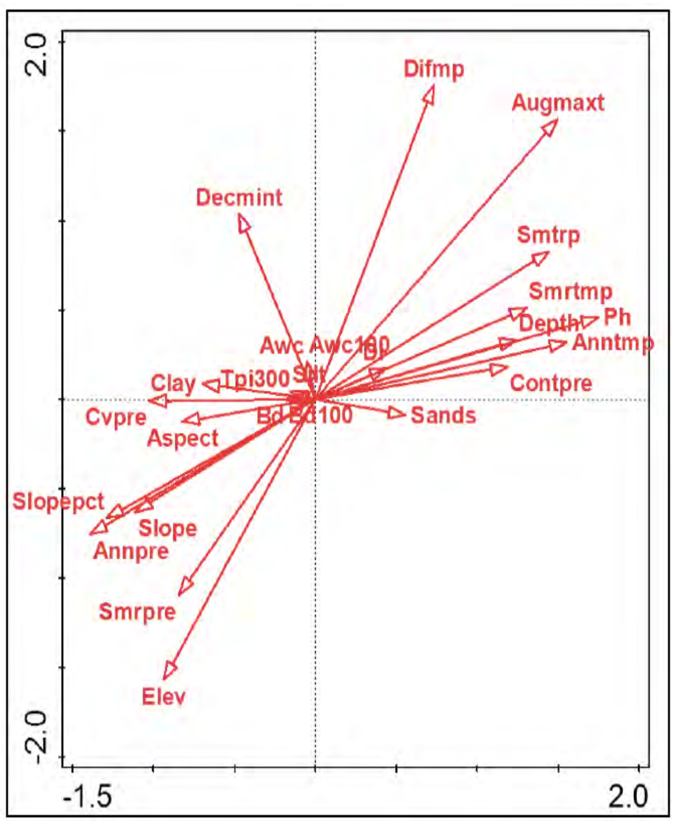

Figure 1. Ordination of sampled environmental variables used in ESD analysis. Longer arrows represent stronger gradients. The dominant gradient is defined by the "x"-axis and the "y"-axis denotes the sub-dominate gradient.

because 25 of the 78 ESDs had only one sample, which would have resulted in no variability in environmental variable scores for those ESDs. The five most impactful environmental variables explain about 34% of that variation; those variables include August maximum temperature (“Augmaxt”), the difference between August maximum and December minimum temperatures (“Difmp”), slope, mean annual temperature (“Anntmp”), and percent sand content in the soil (“Sands”).

Conclusions

Figure 1 suggests that the dominant environmental gradients in the study area are consistent with the use of our current three-model approach. These threat-based models (representing high, medium and low elevation bands) follow a continuum of variation in soil temperature and moisture conditions that is reflective of dominant environmental gradients within the study area. That said, these same variables did a very poor job in assigning samples to their correct ESD (i.e., environmental variables explained less than 15% of the variation in ESD membership). We believe this poor fit is reflective of strong within ESD variation in environmental properties. Put another way, multiple samples of the same ESD did not display similar values for environmental variables. The cause of this variability is either 1) mapping error, 2) an indication that ESDs are too loosely defined relative to the scale of actual environmental variability, or 3) both. This analysis suggests that assembling threat-based models based on ESD membership would be somewhat at odds with

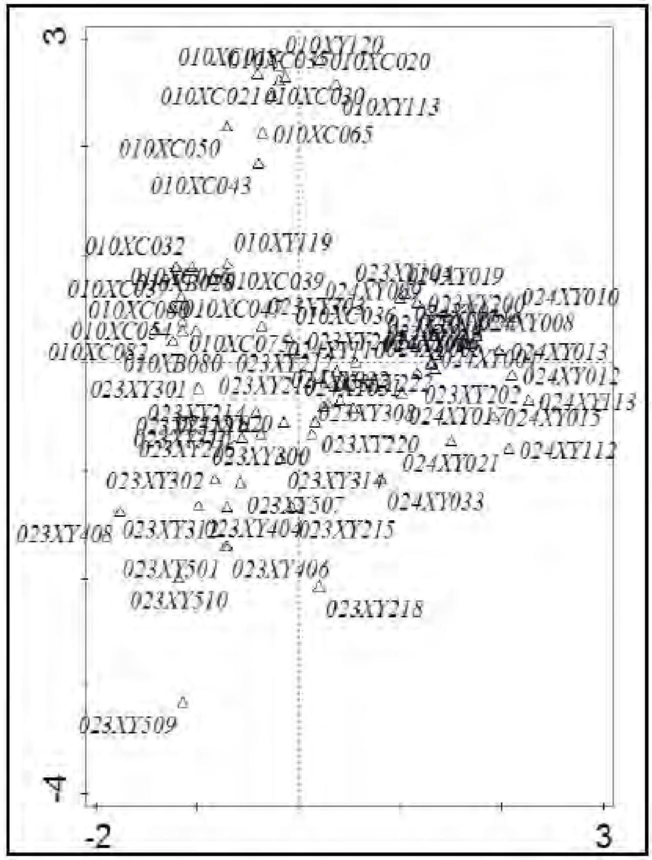

Figure 2. Ordination of site scores for ESDs in Harney County. The position of samples in ordination space correlates to dominant environmental gradients as depicted in Figure 1.

dominant environmental gradients, and further supports the idea of using dominant gradients of temperature and moisture, as encapsulated by elevation, to determine appropriate (broader) threat-based models.

SAGE-SHARE DATABASE USER'S GUIDE

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Introduction

The Greater Sage-Grouse (GRSG) is a sagebrush-obligate bird recently reviewed (2015) by the US Fish and Wildlife Service (USFWS) for protected status under the Endangered Species Act (ESA). It is an indicator species for the western sagebrush steppe, meaning its current state within the ecosystem can be used to effectively determine the state of the ecosystem on a broader scale. Although the decision was made to not list the GRSG, there are still implications for rangeland management and land use. Many of the affected stakeholders lack the best available science to implement broad scale management decisions in sagebrush ecosystems. The Natural Resources Conservation Service (NRCS) awarded a Conservation Innovation Grant (CIG) to The Nature Conservancy (TNC) and the Eastern Oregon Agricultural Research Center (EOARC) to develop new tools for land managers to more effectively and efficiently manage and restore sagebrush steppe habitat with a focus on the Great Basin.

Although a depth of research has been conducted within these systems, plant community response to treatments depends on a complex combination of variables including soils, microclimates, invasive species, fire and other disturbance regimes, current habitat state, historical impacts, and more. We were interested in summarizing existing literature for effects of key management practices on sagebrush steppe ecosystems and GRSG habitat. This effort—**Sage-Steppe Habitat Response (Sage-SHARE)—**resulted in the development of a relational database in Microsoft Access® of rangeland management research conducted mainly in western sagebrush steppe ecosystems. We are not aware of a similar effort and the value of this database lies in its ability to produce a compiled literature review based on multiple search criteria including initial ecological site condition, plant species, applied treatment and study results. The expected outcome of this searchable database will be to inform land manager decision-making for site-specific best management practices. The database was targeted, focusing specifically on rangeland conservation practices. One value to the database approach is that it can help guide future research efforts to areas where information is lacking.

This user's guide was created to assist in the utilization of the Sage-SHARE database including data entry and query. The guide will serve to inform the users of database structure, field definitions, data entry practices, shortcuts and resulting query records. Special attention should be given to the querying section of the guide, as it highlights how the database structure influences search results through a series of example queries and their resulting records.

Sage-SHARE Database: An Extensive Literature Review

Sage-SHARE is a relational database, meaning it is structured to recognize relationships between data. Sources chosen for entry into the database were found via exhaustive keyword searches pertaining to sagebrush, sage-grouse, invasive weed management, conifer encroachment, and the five conservation practices of focus: prescribed (and wild) fire, grazing, seeding/revegatation, mechanized treatments, and herbicide application. The searches yielded peer reviewed articles, reviews, and theses/dissertations. Unpublished projects were not included is this iteration of the database, but may be useful to include in future versions.

Literature compiled was cataloged on EndNote Web, online software providing flexible tools for searching, organizing and sharing research (www.myendnoteweb.com). The benefits of EndNote Web included no service or subscription fee, the ability to import citations, upload resource PDFs, and share research with other EndNote Web users. This allowed for a versatile and license-free mechanism for managing the literature library.

Database Structure

For each publication or journal article, a single s*ource* was created. Associated with each source were the s*ite description*, e*xperimental design* and *results*. The records created within each of the sections were assigned unique identifiers enabling the connection of information between tables (i.e., relationships). The platform used for the database was Microsoft Access®.

The Main Form is the portal to data entry, querying and record review (Figure 1). The buttons on the left side of the Main Form are for entering data into the database while those on the right are for querying data.

Multiple site description records can be created within each source (see sections on site description and experimental design for greater detail). t*reatments* are associated with each site description, and a single site can have more than one type of treatment (i.e., herbicide and prescribed grazing). However, a single site cannot have multiple records of the same type of treatment (i.e., two herbicide treatments within a single site). Figure 2 is a conceptual map showing data entry fields and potential relationships between them. The green bubbles represent forms to fill out for each source, the blue bubbles are fields within each form, and the purple bubbles show some of the options for filling out further details. Because the database is relational in structure, much of the information captured can be sorted and analyzed. For example, the data could be filtered and sorted to yield herbicide practices at low elevations for a particular soil composition. This type of process can lead to products meant to take a broadened view of the data.

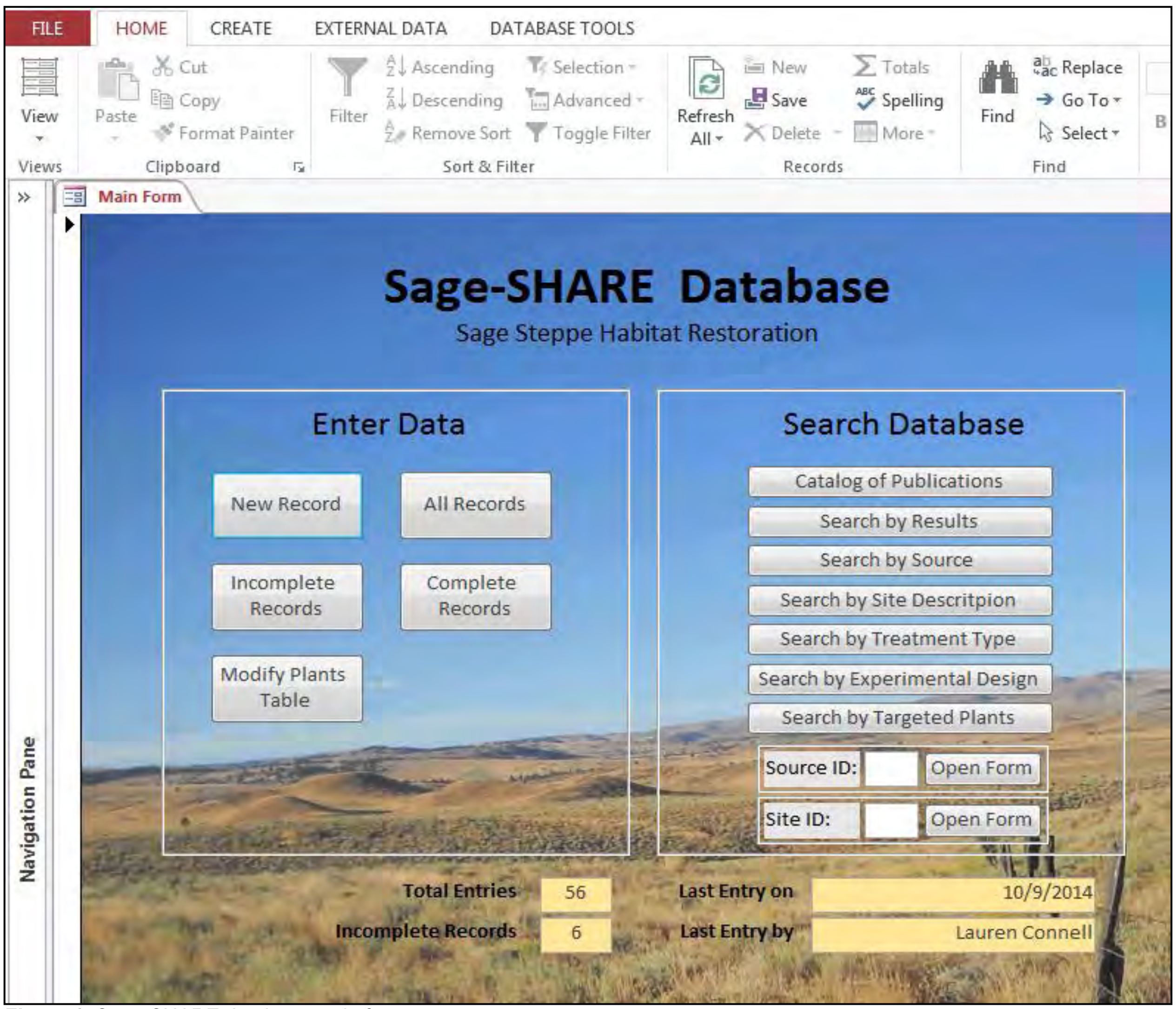

Figure 1. Sage-SHARE database main form.

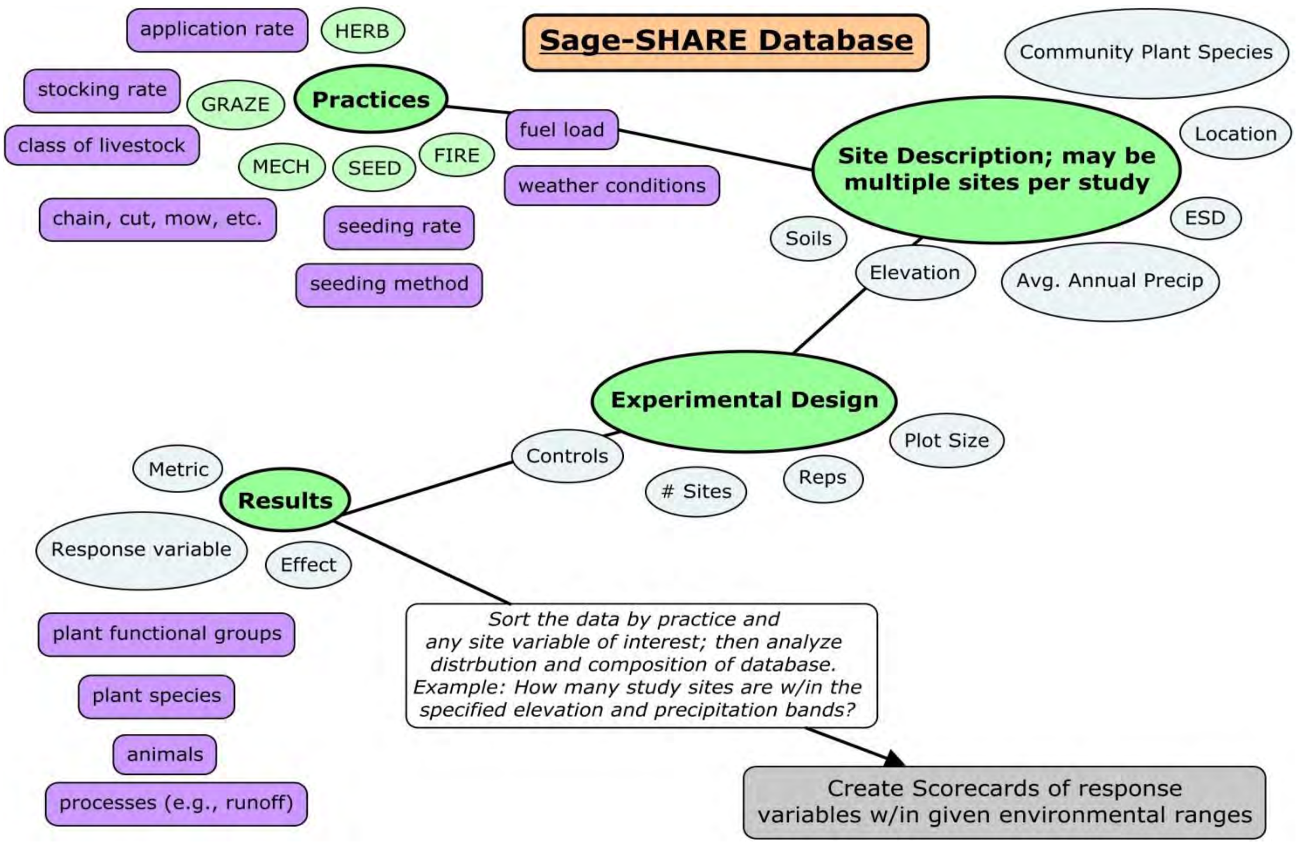

Figure 2. Conceptual map of the relational database. Green denotes separate entry forms, blue are entry fields, and purple are potential details that can be captured. Relationships created by querying data can be grouped and analyzed for products such as the scorecards detailed in Guide 2.

Data Entry

Data was entered as presented in each publication with no intended interpretation of results, ecological site description (ESD), soils, or other attributes. Counties, site coordinates, and other variables were estimated when sufficient information was available. Whenever possible, fields were not left blank to ensure information was not overlooked during data entry. Instead, 'ND' was entered to indicate a field was not described. Numerical fields were left blank when data was not given.

Entry Forms

Each study was entered as an individual *source* with subsequent forms for *site description*, *treatment(s), experimental design*, and *results*. The site description form was meant to represent a single study site with the possibility of entering multiple sites. The other forms did not have the capacity for multiple entries (e.g., the possibility did not exist to fill out two results forms for one study). Treatment methods, target species, application rates, weather conditions, and other information were entered on the respective treatment forms. If multiple site records were created, treatment forms were filled out for each site. Detailed information not captured by existing fields was entered in general comment fields available on each form.

Searching the Database

The search tools on the right side of the Main Form of the database were designed to be simple and straight forward. The queries they perform were based on few, limited criteria. By limiting the complexity of the search criteria, potential users are ensured broader results. The search results were designed to include a wide range of attributes from the database to present a range of similar records. Results can be narrowed by exporting the data into Microsoft Excel® and using analysis tools such as

"Sort" and "Filter" (exported results will not be updated automatically with the database, so queries must be re-executed when needed). All search results contain a unique Source ID and/or Site ID to enable retrieval of full records.

Functionality is limited mainly to each search tool's respective form. In other words, using the Search by Experimental Design tool will only search for records within Experimental Design forms. The *Search by Site Description tool* and *Search by Targeted Plant tool* do have options allowing the user to identify records from multiple forms (e.g., cheatgrass targeted by an herbicide treatment).

Plants were entered into the Sage-SHARE database as they appeared in each publication, even if the scientific name has changed. Since the name provided in the paper was entered into the database with the common name, it is recommended that queries for plant species are performed once with the scientific name and once with the common name to capture more desired records.

Each search tool includes the ability to perform a keyword search. Keyword searches are not case sensitive and can look for a text string or phrase if the string or phrase is an exact match. See Appendix 1 for a list of commonly used keywords.

Additionally, users may develop their own queries when looking for sources with specific keywords or attributes using the query design wizard found on the tool bar of Microsoft Access ®.

Conclusions and Products

The creation of the Sage-SHARE database has culminated in an extensive western sagebrush steppe literature review not been previously documented. Although the database contains data from over 300 publications, it is still just a sample of the relevant literature. It is a tool that can continue to gather information on peer reviewed studies as well as land management projects.

Two products that resulted from analyzing information in the database were scorecards and knowledge gaps. The scorecards organize the effects of various treatments on different plant functional groups (annual grass, perennial grass, forb, shrub, and tree) into different elevation and precipitation bands, making it possible to look broadly at which treatments may work the most effectively under a range of environmental conditions.

The other product to come out of the database was the recognition of knowledge gaps. Gaps arise from the variation in how research is conducted and reported. For example, not all sources entered reported a latitude/longitude or average annual precipitation, making it harder to compare the data. Furthermore, there are countless variables that can affect research (i.e., within a study site, from equipment, human error), and there are differing definitions for whether or not a result was successful.

From the data itself, knowledge gaps in treatments, elevations, etc. were also found. Finding the gaps can lead to further research where it is needed and gathering more sources. It is recommended that input forms be created and distributed for gathering data in the future. See Guide to Rangeland Practices in the Western Sagebrush steppe: Published Scientific Literature for detailed literature review and recommendations for further research.

Appendix 1. Table of Commonly Used Keywords

Keywords	Fields where Keyword is Usually Found
Cheatgrass (*B. tectorum*)	Abstract, Hypothesis, Site Description, Results
Bureau of Land Management	Site Description
Eastern Oregon Agriculture Research Center	Site Description
Exotic	Abstract, Hypothesis, Site Description, Results
Forage (or foraging)	Abstract, Hypothesis, Site Description, Results
Freezing	Abstract, Hypothesis, Site Description, Results
Grazing	Experimental Design, Results
Greenhouse	Site Description, Experimental Design, Results
Humboldt–Toiyabe National Forest	Site Description
Juniper	Abstract, Hypothesis, Site Description, Results
Kaibab National Forest	Site Description
Laboratory (or Lab)	Site Description, Results
Medusahead (*T. caput-medusae*)	Abstract, Hypothesis, Site Description, Results
Microsites	Abstract, Hypothesis, Results
Northern Great Basin Experimental Range	Site Description
Recruitment	Results
Runoff	Results
Sagebrush	Abstract, Hypothesis, Results
Sage-grouse	Abstract, Hypothesis, Site Description, Results
Study sites	Site Description
Tiller	Experimental Design, Results
Wildfire	Site Description, Historical Information, Results

APPLYING THREAT-BASED MENTAL MODELS TO GREATER SAGE-GROUSE CONSERVATION

MANAGER'S GUIDE 1

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Introduction

Greater sage-grouse (*Centrocercus urophasianus*) are a widely distributed ground-nesting bird. Prior to European settlement they inhabited portions of what are now 13 states and three Canadian provinces (Schroeder et al. 2004). They currently occupy slightly more than half of their historical range with populations in 11 states and two Canadian provinces (Knick and Connelly 2011). During September of 2015 a decision was made by the U.S. Fish and Wildlife Service to not list the Greater Sage-grouse (GRSG) a threatened or endangered species under the Endangered Species Act of 1973. However, the decision will be reevaluated in 2020. The scrutiny on GRSG creates an impetus for land managers to remain focused on improving or maintaining the habitat for this species.

Given the broad distribution of GRSG, many stakeholders are needed for successful conservation of the species and its habitat. Threats to GRSG, at least across much of the range, dictate that conservation must be ongoing; a long-term emphasis will be necessary to influence the losses of habitat that result from factors such as invasive species. This report is focused on the western portion of GRSG range which lies mostly within the Great Basin.

Threats to Greater Sage-Grouse

The primary threats to GRSG were clearly articulated in the Greater Sage-Grouse Conservation Objectives: Final Report (U.S. Fish and Wildlife Service 2013). Given the broad range of GRSG, there are a wide variety of threats, from mining and human development, to conifer and invasive annual grass expansion. In the western portion of the GRSG range there is limited energy development, some mining activity, and some agricultural conversion and ex-urban development. The major threats in the region west of the Rocky Mountains are pinyon-juniper expansion, invasion of native sagebrush communities by annual grasses, and wildfires (U.S. Fish and Wildlife Service 2013). These are all ecosystem threats— rather than threats to GRSG directly—which will require a long-term and persistent effort to ameliorate (see Knick and Connelly 2011; Boyd et al. 2014).

The Western Portion of the Greater Sage-Grouse Range

There are major differences in both climate and vegetation in the eastern vs. western portions of the sagebrush steppe and therefore it is important to not treat the entire sagebrush steppe as a uniform biome.

The climatic differences probably explain at least a portion of vegetative differences. One of the major differences is the temporal distribution of annual precipitation. The maps in Figure 1 illustrate the dramatic differences in precipitation between sagebrush steppe west of the Rocky Mountains (eastern Oregon, Nevada, western Idaho, and northern Utah) compared to east of the Rockies (eastern Montana, eastern Wyoming, and western North and South Dakota).

An example of similar seasonal distribution of annual precipitation for two sites is presented in Table 1. In this example, spring and fall are not dramatically different, but summer and winter are. More summer precipitation is thought to favor the shallow-rooted grass species, and more winter precipitation is thought to favor shrubs (e.g., Comstock and Ehleringer 1992; Cook and Irwin 1992). The western portion of the sagebrush steppe can be viewed as shrub/grass communities with shrub cover values often exceeding grass cover values. The opposite is generally true in the eastern portion of the sagebrush steppe where the communities are grass/shrub, with grass cover

Table 1. Comparison of Seasonal Distribution of Precipitation in the Eastern (Miles City, MT) and Western (Riley, OR) Portion of the Sagebrush Steppe (data provided by www.weatherdb.com).

Location	Total Precipitation	Winter	Spring	Summer	Fall
Miles City, MT	11.67[1]	0.63	3.88	4.93	2.23
Riley, OR	11.43	3.53	3.56	1.78	2.56

[1] Values are inches per year averaged over the last 30 years.

exceeding that of shrubs. For example, in eastern Montana canopy cover of shrubs for both Ecological Sites Shallow Clay (SWC) RRU 58A-C 11-14 P.Z. and Clayey (CY) LRU 53A-Y is listed as 1-5% (see section "Relationship between USDA-NRCS Ecological Sites and Threat Model" for more details on ESDs occurring in the western sagebrush steppe). Within the Shallow Clay ESD Wyoming big sagebrush is a significant community component; in the Clay ESD there is a mix of shrubs.

In eastern Oregon, Wyoming big sagebrush cover values are often above 10% for Wyoming big sagebrush communities which have not burned recently, and at times the values can exceed 20% (Davies et al. 2006). There are clearly differences in climate and thus vegetation patterns from east to west. Some of the major threats to GRSG, such as annual grass invasion, conifer expansion, and wildfire also vary along this gradient. Most of the material contained in Guide 1 and Guide 2 will focus specifically on the western portion of the sagebrush steppe.

Mental Models

Getting Stakeholders on the Same Page

One of the challenges to GRSG conservation is the scale of habitat necessary for this species to be successful (e.g., Knick and Connelly 2011), and thus the number of stakeholders necessary for success. In the western portion of the sagebrush steppe there are also complex land ownership patterns with mixes of federal, private, and to a lesser extent state-owned lands. As Boyd et al. (2014) point out, GRSG conservation is maybe the largest and most complex effort ever attempted under the auspices of the Endangered Species Act. While there are clearly threats to GRSG other

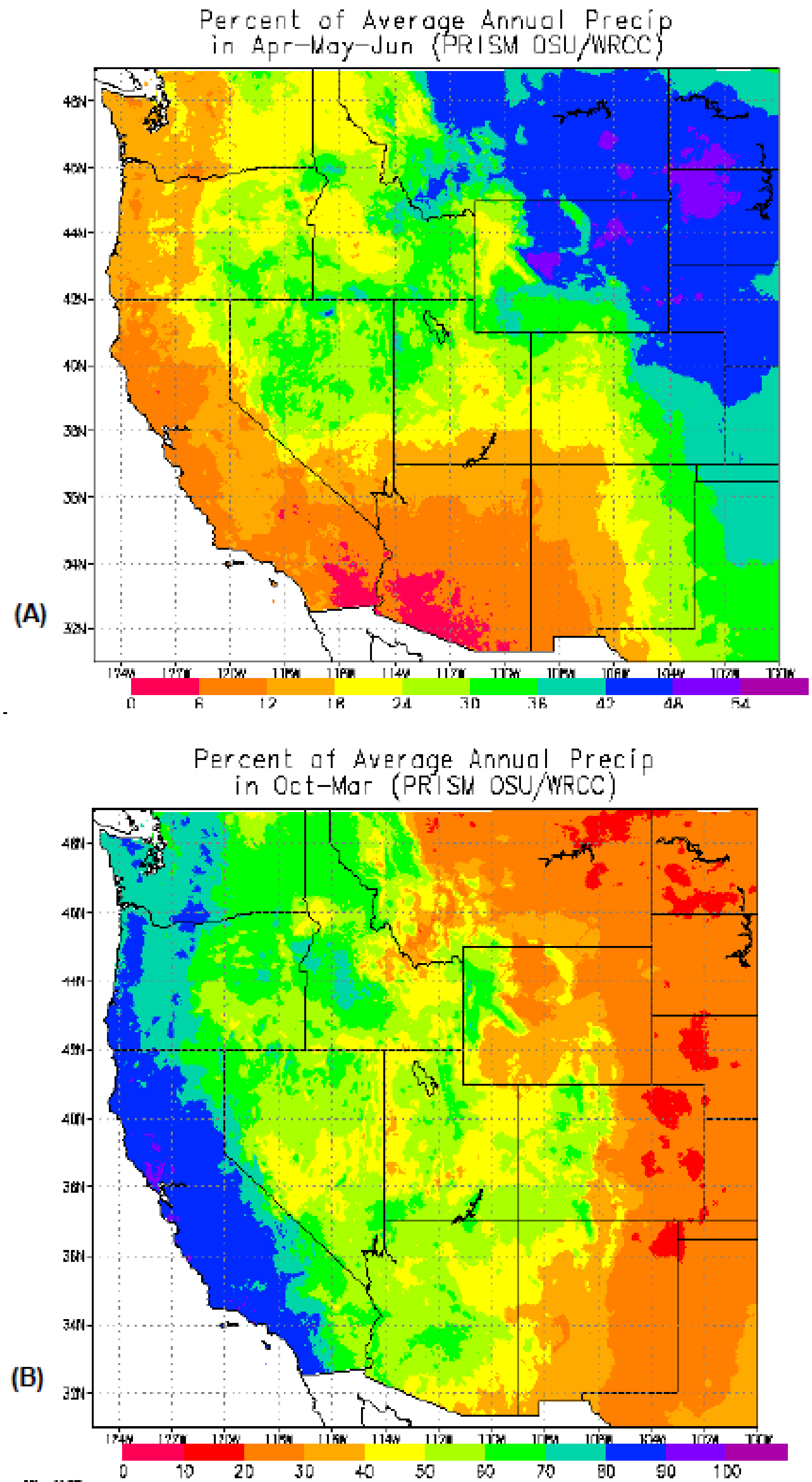

Figure 1. Proportion of annual precipitation during April-June (A) and October-March (B). Scale coloring between A and B is different.

than habitat loss, we will focus entirely on issues related directly to vegetation change, and acknowledge that good habitat for GRSG can be compromised by other factors. Simple mental models are a tool that can accommodate many stakeholder values and serve as a bridge between ecological complexity of vegetation change and conservation decision-making.

The importance of simple mental models increases for issues such as plant community dynamics because stakeholders are likely to have widely varying opinions about both the nature of plant communities and the factors which influence change. Individuals may disagree about impacts of a power line or energy development on GRSG, but these sorts of human impacts are easy to visualize and map.

The concept of mental models evolved from psychology literature of the 1950s (as briefly described by Abel et al. 1998). Mental models allow us to understand the world, predict outcomes of events, and react to new information (Abel et al. 1998). These authors define the primary advantage of mental models as the "structuring and simplification of highly complex reality". Too much information can reduce the effectiveness of decision-making. In their example with grazers in Australia, Abel et al. (1998) point out that complex models would not result in more effective land management by the grazers. They believe that "overelaboration of a theory weakens its power", and that if the grazers expanded on their model (i.e., made it more complex) it would not result in more successful management.

Jones et al. (2011) note that mental models are constructed in working memory and can function as "computer simulations" to test different possibilities before acting. Mental models have been explored as an important component in conservation planning (e.g., Abel et al. 1998; Biggs et al. 2011; Jones et al. 2011) and in risk management planning associated with natural disasters (Wood et al. 2012). In the second case, risk managers recognized the value of aligning policies with stakeholder beliefs, and that mental models were an important part of this process (Wood et al. 2012).

In the case of conservation planning, a lack of alignment between policies or plans and actions leads to a planning-implementation gap (Biggs et al. 2011). There is increasing recognition that conservation planning is often not translated into effective conservation actions (e.g., Knight et al. 2008; Biggs et al. 2011). This lack of follow-through on plans may be one of conservation's greatest challenges (Biggs et al. 2011). In situations where maintenance of habitat may require continual inputs and management attention (as is the case for GRSG), this problem may be particularly acute. Biggs et al. (2011) argue that one reason for the planning-implementation gap is the lack of a shared vision among diverse stakeholder groups. They suggest that mental models can help provide that framework, and they outline examples from a variety of fields including business, organizational science, risk analysis, education, natural resource management, and climate change adaptation. Abel et al. (1998) provide an example where grazers, researchers, and extension agents used a mental modeling process to improve grazing management.

Sage-grouse conservation associated with challenges including complex ecological threats, numerous stakeholder values, and complex ownership and management patterns may benefit from the use of simple mental models. But given the broad distribution of GRSG and the variety of land types, plant communities, and stakeholders, where does one start in the development of mental models for GRSG conservation? One reasonable place to start might be with the major threats. In the western portion of the sagebrush steppe, two of the primary threats are conifer expansion and exotic annual grass invasion. Since these threats (either singularly or jointly) influence 33 of the 39 major GRSG populations (USFWS 2013), they clearly must be a major part of any conservation plan.

Is it Enough to Map Threats?

Many discussions of biodiversity tend to focus on "threats" to either individual species or entire ecosystems. The threats may revolve around invasive species, land development, climate change, or specific human activities not tied directly to land development. Habitat loss and degradation is by far the most common threat to global biodiversity and thus site-based conservation is often necessary (Rodrigues et al. 2006). Threat mapping is a common exercise in conservation planning. Tulloch et al. (2015) point out that threat mapping may not be sufficient for biodiversity conservation if the maps are not tied to clear management objectives. These authors suggest that threat mapping in the absence of objectives linked to social, political, economic, and biodiversity outcomes can result in unintended consequences or misallocation of resources. In other words, plans for addressing threats that include the socioeconomic component must be closely linked to threat maps. Tulloch et al. (2015) suggest using structured decision making (SDM) as means of ensuring that management actions are developed along with threat maps. They characterize SDM as a rigorous, transparent, and interactive approach that involves stakeholders. The basic elements of SDM are:

1) Define clear, quantifiable objectives and constraints relative to the problem;

2) Identify management actions;

3) Evaluate the potential effects of management actions as they relate to initial objectives;

4) Address uncertainty (which may result from either temporal and spatial variability or lack of knowledge); and

5) Assess trade-offs and select a decision.

We agree with Tulloch et al. (2015) that threat maps are only a starting point, and true conservation success will require clearly thought-out management actions. Involving a good cross-section of stakeholders will increase the chances that socioeconomic concerns are considered when developing conservation efforts.

Mental Models in the Real World

In the last two sections we have discussed the value of simple mental models and threat mapping plus SDM in conservation efforts. Two common elements of mental model and SDM discussions are that multiple stakeholders are involved in the process. The real value in these exercises is that a broad array of people from different backgrounds can come together during the developmental process, ensuring that a wide variety of opinions and inputs are considered. While this process involves a good deal of work on the front end, it also increases the chances of success in the long term. We provide some recent examples of this approach in Appendix 1.

As mentioned earlier, the habitat issues for GRSG in the western sagebrush steppe will require participation by many stakeholders over long periods of time. Planning for GRSG conservation is a daunting task, but there are examples where some of the principles outlined in the previous two sections have already been applied. The rest of this guide will focus on the process of developing simple mental models for GRSG conservation, past and ongoing efforts to develop management alternatives, and how this information and the resulting tools can be used in conjunction with ESDs.

The previous sections of this document cover a variety of topics that we believe can be woven into a plan for GRSG conservation. This approach will involve several sequential steps: identify the primary threats, develop a simple mental model that incorporates the threats into habitat (or vegetation) dynamics, and apply SDM for evaluating management alternatives and best use of individual practices.

We see real value in linking a mental model to SDM because the model can be developed to allow for quantifiable objectives. For example, if an area dominated by conifers is a part of the model, then reducing that area (by a percentage or some number of acres) could be a quantifiable objective for Step 1 of SDM. The next steps would involve identifying the management practices needed to meet the objectives, evaluating the effects, uncertainty, and tradeoffs of those practices, and selecting a decision. The same process would apply to invasive annual grasses, or any other threat.

One example of an effort of this sort for GRSG conservation is the “Greater Sage-Grouse Programmatic Candidate Conservation Agreement with Assurances for Private Rangelands in Harney County, OR" (CCAA). The process of developing mental models and management practices to address GRSG habitat threats took place during 2011 to 2013 and involved stakeholders from a wide variety of perspectives. Included were private landowners, local government, non-governmental organizations, state and federal agencies staff involved in land management and GRSG conservation, and state and federal research and extension organizations staff. During the development period, meetings were held monthly with smaller working groups meeting more frequently.

Although the initial focus was not to develop mental models for sagebrush steppe conservation, it was essentially the outcome. For this broad group to come to agreement it was necessary to simplify the approach. The following paragraph was provided by Dr. Chad Boyd, who was involved in the CCAA process from start to finish:

> *In our experience, disagreements over often contentious practices such as live-stock grazing can be the result of differing mental models of habitat ecology. Working through the process of defining strategy, goals, and objectives for dealing with complex natural resources problems will benefit greatly from development of a mental model that depicts both the ecology of plant communities, and the relationship of that ecology to wildlife species of interest. This is particularly true when multiple uses/interest groups have a stake in management outcomes. Without a common mental model, the human tendency is to focus—either positively or negatively—on actions and management tools. How that focus plays out will differ from individual to individual, based on past experiences and values-based biases. In such cases, the differing mental models of stakeholders can make progress in building consensus on management strategies, goals, and objectives all but impossible. Thus, before discussing objectives or specific management practices it is necessary to ensure sufficient commonality in perceived ecology and the relationship of that ecology to wildlife species of interest in order to move in a productive direction.*

As stated previously, a useful way to begin this process is to ask participants to verbally "paint a picture" of their idealized version of habitat condition. In our experience, this idealized condition can be surprisingly similar between disparate participants, and can be used as a starting condition for building a group mental model. From there, common plant community deviations from the idealized condition are used to populate the model with additional conditions (see Figure 2). These deviations can be defined by either ecological significance, significance to the wildlife species of interest, or both. The aim here is not to create a comprehensive and detailed catalog of all possible plant communities, but rather, to define a small number of conditions that represent common plant community forms and management challenges. Our experience suggests that more than five conditions can result in a model that is overly complex and may not be intuitive to all participants. Once all conditions have been identified they are assigned qualitative values defining ecological properties (e.g., resistance and resilience, Chamber et al. 2014) and habitat values for the wildlife species of interest (Boyd et al. 2014).

The CCAA discussions settled on the concept of separating the landscape into three general elevation bands based on the primary threats to GRSG habitat in the western sagebrush steppe. However, exact threat elevations are dependent on site potential (as influenced by precipitation, aspect, soil depth, etc.). The identified threats were a shift in plant community dominance to: 1) invasive annual grasses only, 2) invasive annual grasses and/or conifer, or 3) conifer only. Generally, the invasive annual grasses are more of a threat at low to mid elevation sites, and conifers are a threat at mid to high elevation sites.

The three CCAA threat models are combined and slightly modified into a single diagram (Figure 2). Figure 2 was developed to show the relationship of each threat-based model to each other. The colored boxes represent habitat conditions in terms of potential utilization by GRSG: green represents potential year-round GRSG habitat, yellow is potential seasonal habitat, and red is non-habitat. For example, areas with adequate sagebrush cover and an understory dominated by annuals in the lower elevation in non-conifer sites can provide seasonal habitat. This is also true of very early phases of conifer encroached sagebrush steppe. However, as conifer cover exceeds about 4%, potential for GRSG habitat is greatly reduced (Baruch-Mordo et al. 2013). Conditions such as these are represented in Figure 2 as red or non-habitat.

This relatively simple model represents a broad array of possible plant communities which occur in the western sagebrush steppe. The model depicts increasing productivity and site resilience from left to right. The concepts of resistance and resilience are approaches for capturing the effects of abiotic site characteristics on vegetation response to disturbance or susceptibility to a threat (e.g., Chambers et al. 2014).

The CCAA models are structured according to threats rather than elevation for several key reasons. While elevation is certainly a major factor in productivity with higher sites being generally more productive, other factors such as aspect, slope, soil type, and topographic position can all play a role. Precipitation generally increases with elevation, but these other factors create interactive effects that influence a site's resistance and resilience.

To illustrate this point, consider that the Great Basin is characterized by pronounced local gradients in precipitation that can be explained by highly variable topography of the region (Hidy and Klieforth 1990). When topography causes air to rise, the water vapor cools and condenses, resulting in precipitation (generally on the windward or western side of mountain ranges). But as elevation declines on the leeward side, water vapor has been depleted and lowlands can be very dry (Hidy and Klieforth 1990). Thus a given elevation in windward position may receive more precipitation than the same elevation in a leeward position. Again, productivity generally correlates with elevation, but there are many exceptions.

The productivity gradient represented in Figure 2 means that Wyoming big sagebrush and annual grasses will be more common on the left side of the figure, and Mountain big sagebrush and conifers will be more common to the right. The mixed threat areas are probably more difficult to conceptualize than either single threat part of the landscape. Keep in mind that the intention is only to provide a generalized diagram. There can be areas with only Wyoming big sagebrush where both conifers and annual grasses are a threat, and the same can be true for Mountain big sagebrush. The purpose of the model is to provide a means of quickly assessing threats to GRSG habitat, not to describe all existing plant communities in detail.

The color-coded boxes in Figure 2 follow the protocol used in the Harney County and other Programmatic CCAAs, but the conceptual figure does not include the letters representing potential habitat conditions corresponding to each color (see Tables 2, 3, and 4 for conservation measures linked to habitat conditions by letter). Included in this diagram are phases of conifer encroachment, which did not appear in the original CCAA models.

The top two wide boxes in Figure 2 represent a large productivity gradient in addition to a resistance and resilience (R&R) gradient. For example, the green box will always be dominated by big sagebrush and large perennial bunchgrasses, but on the left (lower productivity and R&R) sagebrush cover might be 8-12% and bunchgrass density 5-10 plants/m^2 in eastern Oregon (values may be lower in northern Nevada). To the right these values could easily double as site potential increases.

The spectrum of habitat restoration potential is indicated along the y-axis of Figure 2. Restoration is most difficult in highly degraded sites that offer little to no habitat utility for sage-grouse (red boxes). Restoration potential increases with increased site productivity and resilience, but also with a decrease in the presence of habitat threats. See Appendix 3 for accepted metrics of habitat condition by threat model.

Tables 2, 3, and 4 provide further detail pertaining to the annual invasive grass, conifer, and dual threat models, respectively, and retain the color-coding representing potential habitat use with the alphabetical notation used for the CCAAs. For example, in Table 2 (which represents sites with an invasive annual grass threat) both B and C provide seasonal habitat (yellow), but C is much less ecologically stable than B because deep-rooted perennial grasses are depleted. Examples of each habitat condition by threat model are also shown in Appendix 2. It is important to note that the habitat condition is not always clear-cut (e.g., a definite B or C), and that there is a range of productivity within each habitat condition (as mentioned above).

For each habitat condition, Tables 2, 3, and 4 provide a general conservation objective and examples of conservation measures to achieve the objective. Relevant trend indicators for measuring progress towards achieving the conservation objective are also included. As with the previously mentioned example in Table 2, conservation measures and objectives are different for the two habitat conditions that occur within a seasonal habitat (yellow) B and C condition. Similarly, in Tables 3 and 4, red still represents non-habitat, but differences in the nature of the habitat threats between models dictate that C, D, and E will require slightly different conservation objectives and measures. Table 2 focuses on habitat conditions that occur in sites predominately threatened by annual invasive grasses. Note that the habitat conditions A through D correspond to the four habitat condition boxes on the left side of Figure 2. Table 3 corresponds with the conditions described in the middle portion of Figure 2, while Table 4 corresponds with those to the right.

Site-Specific Plans (SSPs) are required by the terms of the Programmatic CCAA, and Tables 2, 3, and 4 can be useful tools in the development of SSPs. Once the threat(s) on a site have been identified, these tables can help determine the appropriate conservation objective(s), associated conservation measure(s) to limit or remove the threat(s), and short- and long-term monitoring protocol(s) that indicate trend within a site. SSPs incorporate adaptive management styles while providing a strong communication link among stakeholders. Refer to Appendix 1 for more details on how the SSP format is a good example of applying mental models in land management.

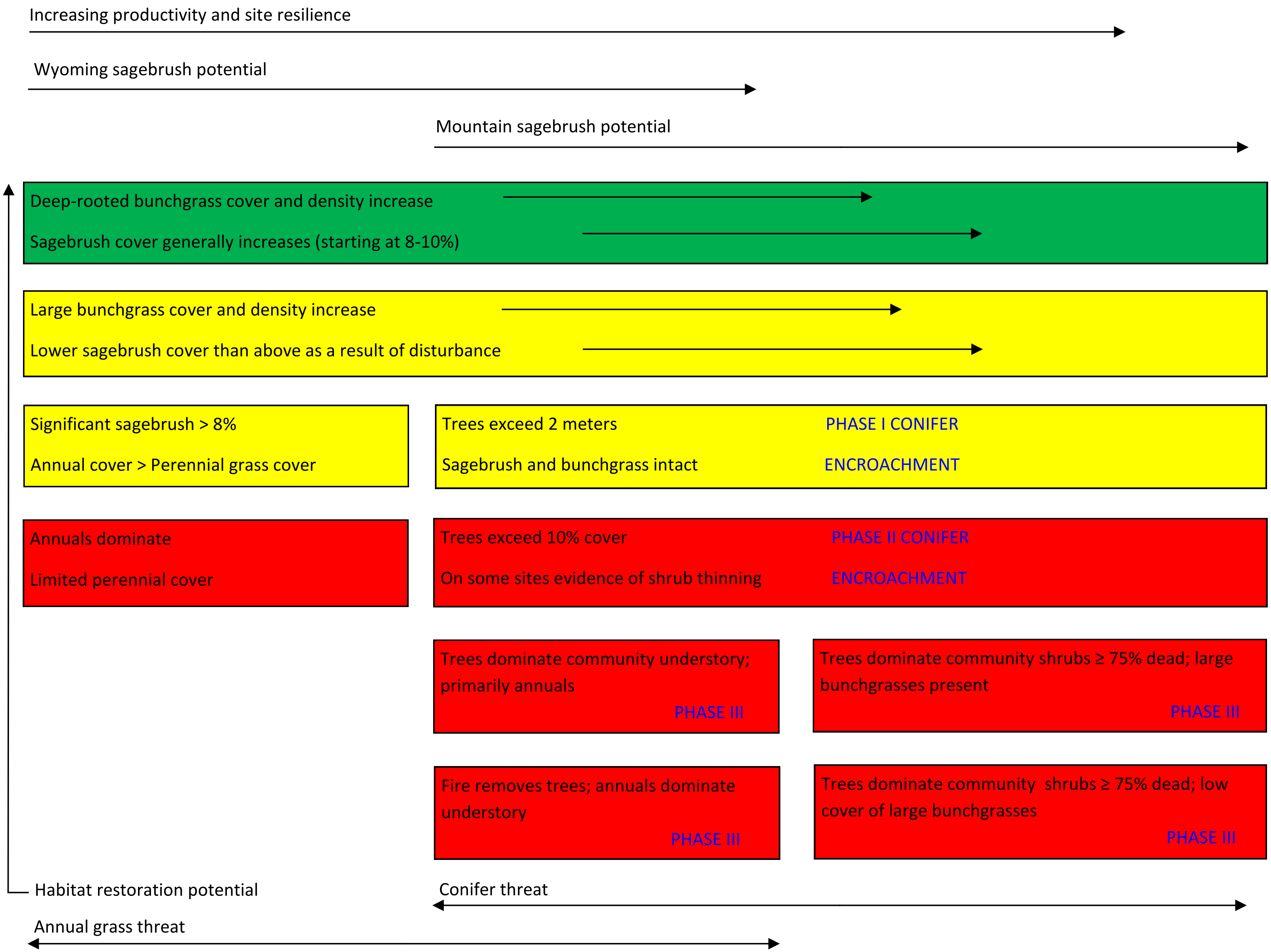

Figure 2. Combined threat-based models for the western sagebrush steppe where green represents year-round habitat, yellow represents seasonal habitat, and red represents non-habitat. Site productivity and resilience increase from left to right. Annual grass is a larger threat for Wyoming versus mountain big sagebrush-dominated communities, while conifers are a larger threat in mountain sagebrush-dominated communities (see Appendix 2 for sample sites). Restoration is more difficult in non-habitat when compared with seasonal habitat.

Table 2. Conservation Objectives and Measures, and Relevant Trend Indications for Annual Grass Threat-Based Model.

Habitat Condition	General Conservation Objective	General Conservation Measure(s)	Relevant Trend Indicators
A	Maintain cover of sage-brush and the density of desired, deep-rooted perennial grasses.	Maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed if estimate of apparent trend is downward.	Sagebrush cover. Density and juxtaposition of deep-rooted perennial grasses.
B	Promote recruitment of sagebrush cover and maintain density of de-sired, deep-rooted peren-nial grasses.	Prevent fire and maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed in addition to preventing fire if apparent trend is downward.	Sagebrush cover. Density of deep-rooted perennial grasses.
C	Maintain sagebrush cov-er and promote recruit-ment of deep-rooted perennial grasses.	Prevent fire. Experimentation with methods that maintain sagebrush cover and promote perennial grass establishment.	Sagebrush cover. Density and juxtaposition of deep-rooted perennial grasses.
D	Promote recruitment of deep-rooted perennial grasses.	Reduce fire risk, control weeds, and revegetate with desired deep-rooted perennial grasses.	Density of deep-rooted perennial grasses.

Table 3. Conservation Objectives and Measures, and Relevant Trend Indications for Annual Grass/ Conifer Threat-Based Model.

Habitat Condition	*General Conservation Objective*	*General Conservation Measure(s)*	*Relevant Trend Indicators*
A	Maintain cover of sage-brush and the density of desired, deep-rooted perennial grasses.	Maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed if estimate of apparent trend is downward.	Sagebrush cover. Density and juxtaposition of deep-rooted perennial grasses.
B	Promote recruitment of sagebrush cover and maintain density of desired, deep-rooted perennial grasses.	Prevent fire and maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed in addition to preventing fire if apparent trend is downward.	Sagebrush cover. Density of deep-rooted perennial grasses.
C	Maintain the cover of sagebrush and the density of desired, deep-rooted perennial grasses.	Remove conifers.	Sagebrush cover. Conifer cover. Density and juxtaposition of deep-rooted perennial grasses.
D	Promote recruitment of sagebrush and desired, deep-rooted perennial grasses.	Remove conifer overstory and revegetate with desired deep-rooted perennial grasses.	Sagebrush cover. Conifer cover. Density of deep-rooted perennial grasses.
E	Promote recruitment of deep-rooted perennial grasses.	Reduce fire risk, control weeds, and revegetate with desired deep-rooted perennial grasses.	Density of deep-rooted perennial grasses.

Table 4. Conservation Objectives and Measures, and Relevant Trend Indications for Conifer Threat-Based Model.

Habitat Condition	*General Conservation Objective*	*General Conservation Measure(s)*	*Relevant Trend Indicators*
A	Maintain cover of sagebrush and the density of desired, deep-rooted perennial grasses.	Maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed if estimate of apparent trend is downward.	Sagebrush cover. Density and juxtaposition of deep-rooted perennial grasses.
B	Promote recruitment of sagebrush cover and maintain density of desired, deep-rooted perennial grasses.	Prevent fire and maintain current management if apparent trend is stable or upward. Management changes or conservation measures may be needed in addition to preventing fire if apparent trend is downward.	Sagebrush cover. Density of deep-rooted perennial grasses.
C	Maintain the cover of sagebrush and the density of desired, deep-rooted perennial grasses.	Remove conifer.	Sagebrush cover. Conifer cover. Density and juxtaposition of deep-rooted perennial grasses.
D	Promote recruitment of sagebrush and maintain density of desired, deep-rooted perennial grasses.	Remove conifer overstory.	Sagebrush cover. Conifer cover. Density of deep-rooted perennial grasses.
E	Promote recruitment of sagebrush and desired, deep-rooted perennial grasses.	Remove conifer overstory and revegetate with desired deep-rooted vegetation.	Sagebrush cover. Conifer cover. Density of deep-rooted perennial grasses.

Relationship between USDA-NRCS Ecological Sites and Threat-based Model

We previously described some of the major characteristics of the western sagebrush steppe. The focus of this section of the guide is to provide background on the USDA-NRCS Major Land Resource Areas (MLRAs) and Ecological Site Descriptions (ESDs) common to our area of interest in the western sagebrush steppe. Our priority area includes significant portions of Oregon, Idaho, and Nevada, with smaller areas in California and Utah (Figure 3).

Land classification systems generally have had a goal of grouping similar areas based on land capability and characteristics. Ecological land classifications use factors such as soils, landform, hydrology, vegetation, climate, or habitats to group land units. Agricultural land classification can include many of the same factors, and also relevant crop species, productivity, and erosion potential.

In the U.S., a land hierarchical classification system has been developed by the USDA-NRCS. Within the hierarchy, MLRAs describe large areas. In the western sagebrush steppe, the MLRAs we will describe are eight to eighteen million acres in size, whereas the ESDs within each MLRA may represent areas in excess of two million acres or less than 1000 acres. Within each ESD is a description of physiographic, climatic, water, soil, and plant community features. Each ESD also contains a state-and-transition model (STM) intended to represent the potential plant communities and causes of transition from one community to another. The ESDs were developed as a standardized method to be used by the USDI-BLM, USDA-Forest Service, and USDA-NRCS to provide a conceptual division of the landscape (Caudle et al. 2013). The ESD concept includes the abiotic and biotic factors listed above, as well as notes about response to management and natural disturbance regime. Given our area of interest we will include MLRA 23, 24, and 25 in this

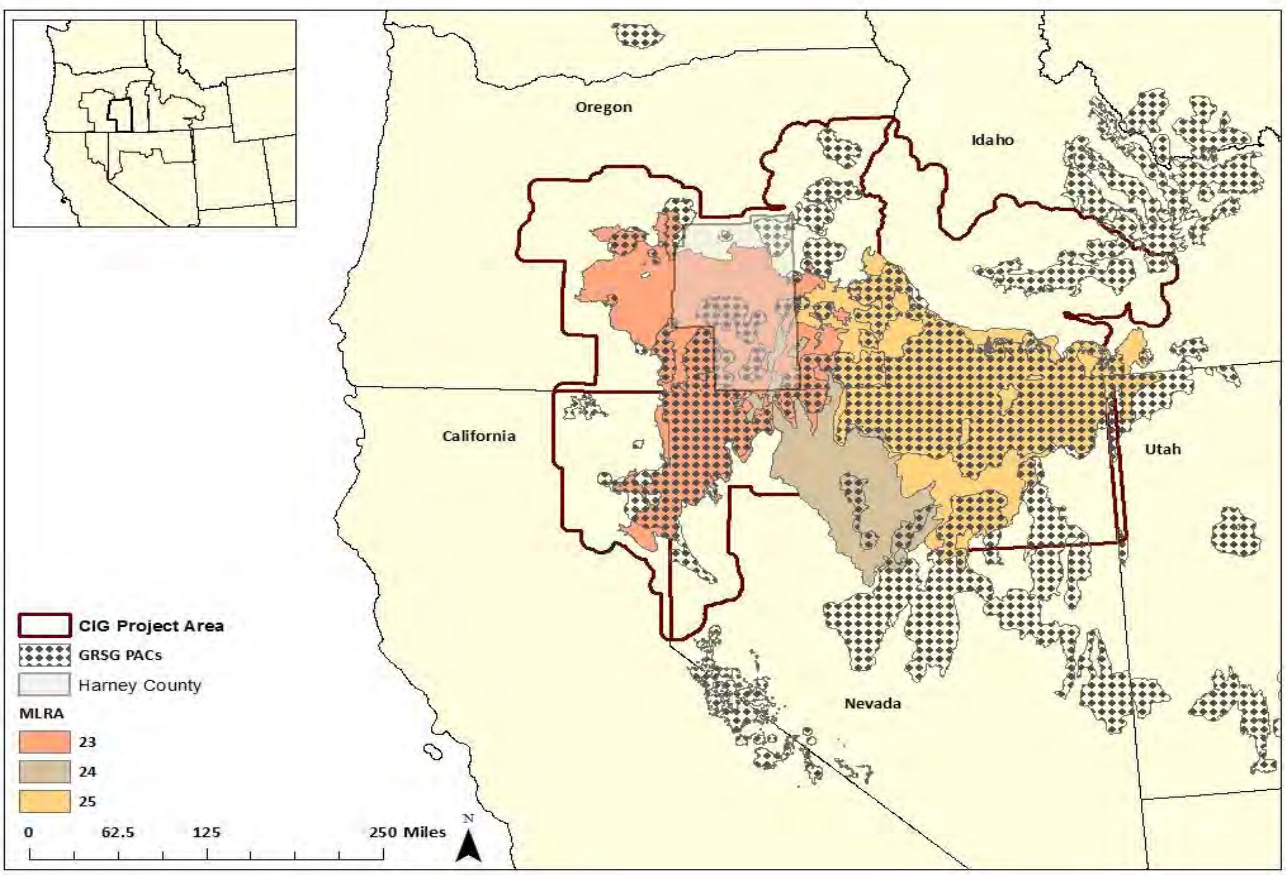

Figure 3. Project area showing GRSG Priority Areas of Conservation (PACs), and MLRAs 23, 24, and 25 with Harney County (OR) outlined in red.

discussion.

The general characteristics of these MLRAs can be found in Land Resource Regions and Major Land Resource Areas of the United States, the Caribbean, and the Pacific Basin (USDA-NRCS 2006). The information provided below on the MLRAs comes directly from this publication.

The Malheur High Plateau (MLRA 23) crosses three states with 67% in Oregon, 25% in Nevada and 8% in California. The total area of MLRA 23 is 14,652,800 acres. The primary land use is grassland at 84%. Grassland is the combination of pasture, range, brush and tundra categories from the National Resources Inventory conducted by USDA-NRCS. The Humboldt Area (MLRA 24) is 94% in Nevada and 6% in Oregon. The total area of MLRA 24 is 8,115,200 acres, and in this case 95% of land use is classified as grassland. The Owyhee High Plateau (MLRA 25) is 52% in Nevada, 29% in Idaho, 16% in Oregon and 3% in Utah. The total area of MLRA 25 is 18,515,200 acres, and again 95% of the area has grassland as the primary land use.

All three of the MRLAs described above are subject to potential invasive annual grass and conifer encroachment threats (in addition to wildfire). Therefore, it is appropriate to consider them—and their associated ESDs—in the context of the combined threat-based model presented in Figure 2. Based on STMs and professional experience, the major ESDs can be grouped into three threat categories: annual grass only, annual grass and conifer, or conifer only (Figure 5).

This enables the conceptualization of information using a nested approach in which the detailed ESD information is viewed through the lens of a simplified mental model consisting of generalized ecological threats.

Detailed ESDs can be found at: http://esis.sc.egov.usda.gov/Welcome/pgReporLocation.aspx?type=ESD

This site allows sorting of ESDs by state or MLRA, and the supporting information can be accessed by clicking on an individual ESD Site ID from the table of all ESDs within a particular MLRA. All ESDs from MLRA 23 in Oregon are available, but ESDs from MLRA 23, 24, and 25 in Nevada are currently under review.

Relavent Ecological Threats			
	Annual Grass Threat ◆———◆		
		Juniper Encroachment Threat ◆———◆	
Major Land Resource Areas (MLRAs) and Example Ecological Site Descriptions (ESDs)			
MLRA 23	**Loamy 10-12** (023xy212 OR) 1,235,938 acres	**Loamy 12-16** (023xy318 OR) 140,739 acres	**Shallow Loam 16-2** (023xy501 OR) 43,636 acres
MLRA 24	**Loamy 8-10** (024xy005 NV) 1,013,062 acres	**Loamy Slope 12-14** (024xy021 NV) 219,193 acres	**No conifer-only ESDs** *Dr. Tamzen Stringham, pers. comm.*
MLRA 25	**Claypan 10-12** (025xy018 NV) 404,724 acres	**Loamy 8-10** (025xy019 NV) 2,554,757 acres	**Loamy Slope 16+** (025xy004 NV) 133,319 acres
	Increasing productivity and site potential ——→		

Figure 5. A comparison between the ecological threat models and examples of Ecological Site Descriptions (ESD) from selected Major Land Resource Areas within the project area (see Figure 1 above). ESD descriptions include name, unique identifier, and acreage. Example ESDs are arranged in order of increasing productivity and site potential.

A Strategy for GRSG Habitat Conservation

Previously we described the steps in SDM as presented by Tulloch et al. (2015). These authors suggest that threat maps alone "may be insufficient for making cost-effective conservation decisions", and that developing objectives and associated management practices will be necessary for better conservation decisions and follow-through. In his book *Seven Habits of Highly Effective People* (1989), author Stephen Covey suggests that time and effort spent on problems within our circle of knowledge will expand the circle over time, effectively increasing our capacity to deal with more complex problems. Thus, it is imperative that managers be honest about their scale of knowledge and be strategic in allocation of resources between the problems we can address and the problems where we lack capacity.

Knight et al. (2008) show that published conservation assessments do not include implementation plans and therefore do not result in action, creating a research-implementation gap. The combination of a threat-based mental model (Figure 2) and SDM should provide the detail necessary to develop an overall strategy for GRSG habitat conservation in the western sagebrush steppe. A broad diagram of the integration of a simple mental model and SDM is presented in Figure 6 with two habitat categories ("good habitat" and "degraded or threat-impacted habitat"). In this section we will focus on Step 1 of the SDM process—the development of clear, quantifiable objectives and constraints related to the problem, and measurable attributes of each. For the threat-based models presented in this report, the clear objectives should be to reduce threats (annual grasses, conifers, or both).

Setting Objectives

The process of setting objectives is critical for any planning effort. Our knowledge of vegetation dynamics and management actions are integrated via the objectives (see Figure 6). In other words, the

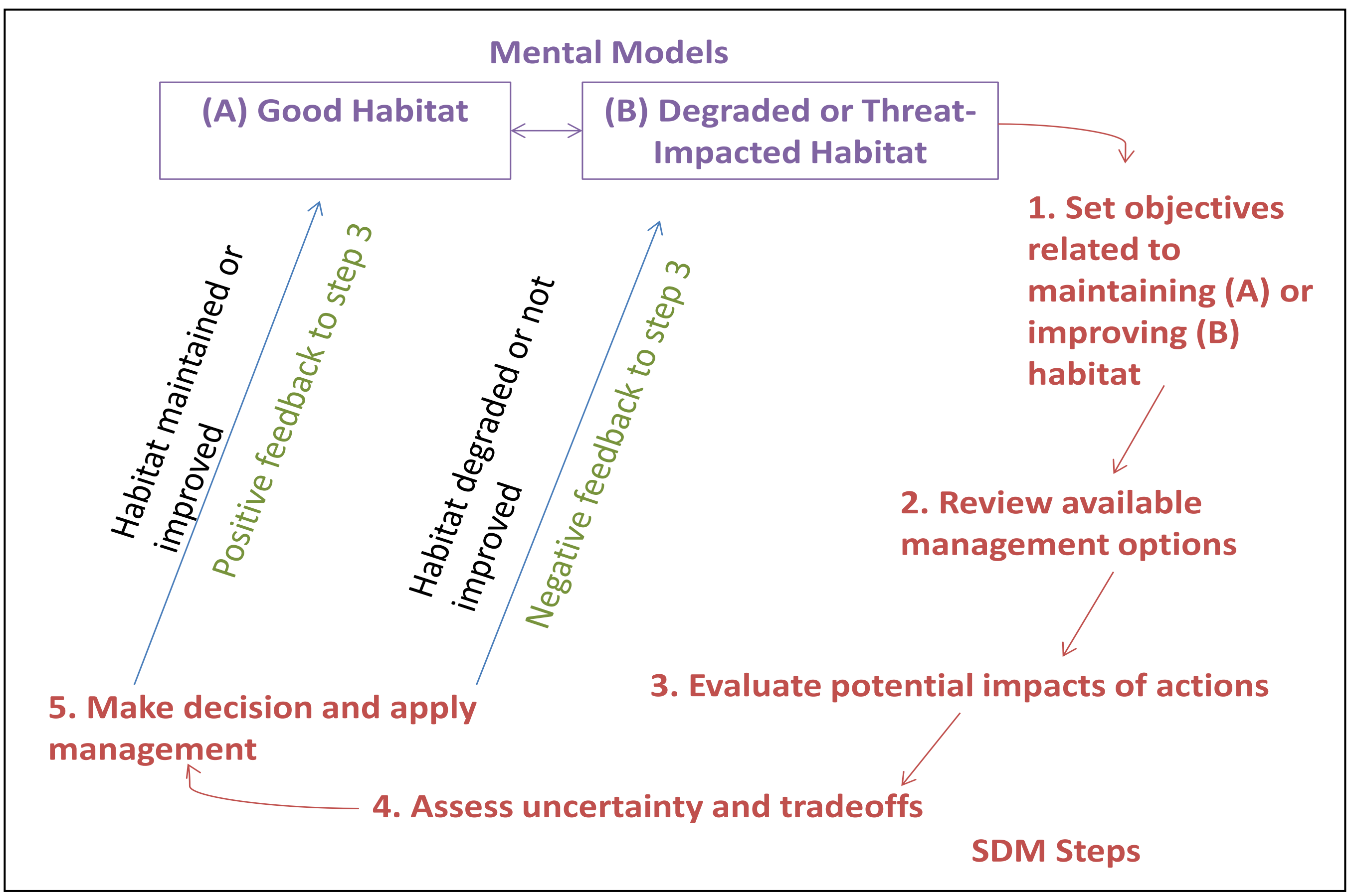

Figure 6. Diagram linking mental model structured decision making (SDM) processes that integrates adaptive management.

objectives link our mental model of vegetation change to the actions we select for conservation.

Objectives must meet several criteria in order to be useful in managing GRSG habitat. First, they must be measurable so that we can track progress or evaluate competing management alternatives. Second, objectives are directly linked to an assessment process. Third, objectives should include ecological processes that can be influenced either through manipulation or maintenance. For purposes of this discussion, an ecological process is anything that can cause change in the composition, structure, or function of a plant community.

Quantitative objectives are often useful in a production context where specific levels of production equate to specific levels of change in ecological properties. However, quantitative objectives may not always work well when dealing with complex problems because potential of the system is often not known or varies to the extent that quantitative objectives are difficult to determine.

The type of objective chosen will impact the intensity of monitoring needed to determine achievement of the objective. Qualitative objectives define a desired direction of change in the current plant community that is consistent with a desired future condition. For example, let's say the objective is to achieve a density of five bunchgrass plants per square meter within a management area that contains multiple ecological sites and extreme topographic variation. Sampling effort needed to accurately determine bunchgrass density across the management area may be logistically intensive. In contrast, if the objective was to increase the density of perennial bunchgrasses, monitoring trend (as opposed to a specific density value) could be appropriate. In this case, an upward trend in bunchgrass density in a few key areas could suggest performance within the management area that is consistent with the objective. Use of trend objectives is appropriate in many instances where we have an incomplete knowledge of site potential, and how that potential varies across and within management units and through time.

The conservation measures presented in Tables 2, 3, and 4 demonstrate clear objectives based on either maintenance of desirable habitat condition, or improvement of habitat condition. For example, potential year-round sage-grouse habitat (as designated by the green boxes in Tables 2, 3 and 4) is characterized by adequate sagebrush cover and deep-rooted perennial grasses, so maintenance is the objective. In the annual grass threat model (Table 2), habitat condition C denotes seasonal habitat because sagebrush cover is adequate, but is at risk of conversion to habitat condition D (annual invasive grass dominance) with a wildfire. Lack of deep-rooted perennial grasses creates this risk, so increasing the density of these species should be a conservation objective.

In these cases the general conservation objectives are presented to favor maintenance or recovery of the species that have shown the ability to compete with annual grasses, and maintain the annuals as a minor component of the community (e.g., Davies et al. 2011). The reality is that invasive annuals are nearly ubiquitous in the western sagebrush steppe, so eliminating them entirely from a community is probably an unrealistic objective. Maintenance of perennial grass cover (or density) is a fairly straightforward objective that can be easily communicated to a broad base of stakeholders. At this point it is useful to outline the potential components of a GRSG conservation strategy (see Table 5). This particular strategy focuses entirely on habitat, and is presented only as an example. Most successful strategies (including those required in setting up SSPs for CCAA-enrolled lands) will include local input and involvement of the stakeholders who will implement the strategy. . The first step in SDM is to set clear, quantifiable objectives with measurable attributes. In our generalized strategy (Table 5) we list objectives that can be quantified in a number of ways. For example, if we were on a site in habitat condition C in which sagebrush and large bunchgrasses were adequate, and where only conifers were a threat, the clear tactic would be to remove the conifers. By mapping the initial pre-treatment area in this condition and remapping after management had been applied it is possible to quantify the objective.

Table 5. Generalized Strategy for GRSG Habitat Conservation.

Goal: Reduce the two primary vegetation threats to GRSG habitat (annual grasses and conifers).
Objectives: 1) Where sagebrush habitat condition A[1] exists, maintain the cover of sagebrush and large bunchgrasses. 2) Where sagebrush habitat condition B[1] exists, promote recruitment of sagebrush and maintain large bunchgrasses. 3) Where sagebrush habitat condition C[1] exists, maintain sagebrush cover, maintain or promote large bunchgrasses, and remove conifers if present. 4) Where sagebrush habitat condition D[1] exists, promote recruitment of sagebrush and if necessary large bunchgrasses; remove conifers if present. 5) Where sagebrush habitat condition E[1] exists, remove trees if present and restore sagebrush and large bunchgrass cover.
Tactics (Conservation Measures): The first step in developing tactics appears as general conservation measures in Tables 2, 3 and 4. Clearly, the tactics will revolve around reducing annual grasses and conifers, or conversely increasing or maintaining cover of sagebrush or large bunchgrasses. Tactics can be rather complex in this case, depending on site potential, initial habitat phase and available resources. The second guide in this series will focus on current knowledge of a variety of rangeland management practices, and how these practices vary based on landscape position. [1]Letter designations (A-E) shown in Tables 2-4.

In this example acres of habitat improvement would be the metric. This is a simple example because the sagebrush and large bunchgrasses were intact and removing conifers was the only conservation measure needed (Figure 2 and Table 4).

Conclusions

There is a great deal of complexity and a multitude of factors involved in managing the western sagebrush steppe. This ecosystem is characterized by high spatial and temporal variation, complex ownership patterns, and lands that are often managed for multiple uses. Greater sage-grouse have become a focal species in this region, and conservation of this species will require a wide variety of stakeholders working together on common goals.

During the past decade there have been a growing number of conservation scientists highlighting the value of simple mental models in conservation. The potential for mental models to help bridge the planning/implementation gap has been a theme from this area of the conservation literature (see citations from earlier in this document). Some conservationists list the planning/implementation gap as one of the biggest challenges for present day conservation. Coordinating stakeholders for conservation efforts is greatly enhanced when there are mental models and simple methods of linking management efforts to positive conservation outcomes. Outlining a course of action in a two-page document is more likely to be successful than using a 100-page document because initially maintaining a simple approach increases communication among stakeholders and creates a common focus on desirable outcomes. More complex tools can be integrated into conservation efforts once the basic framework has been established. For example, geospatial analysis may be important for assessing GRSG habitat because this species requires large tracts of intact sagebrush steppe habitat. But, to maintain large tracts of sagebrush steppe, multiple stakeholders will need to work in a coordinated fashion to reduce habitat threats.

In this guide we provide:

1) a generalized mental model of two major threats to GRSG habitat—annual grasses and conifers (Figure 2), 2) some of the primary conservation objectives and measures associated with these threats (Tables 2, 3 and 4), and 3) a simple strategy for GRSG habitat conservation (Table 5).

The tactics for how to apply the strategy will revolve around management practices. The second guide in this series will focus on rangeland management practices, specifically fire, grazing, seeding, mechanical treatments, and herbicides.

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Appendix 1: Example Conservation Programs Combining Mental Models & SDM

A systematic approach to conservation is critical for planning, implementing, and monitoring conservation efforts. This is the reason for the Open Standards for the Practice of Conservation (www.cmpopenstandards.org) – to learn what does and does not work and why. We proposed the use of mental models in conjunction with structured decision-making (SDM) as a means to overcome the conservation planning and implementation gap. Presented below are examples of conservation programs which have adopted a similar approach. One example is broad while one is specific, but both share two characteristics: 1) an emphasis on the link between human needs and conservation, and 2) communication among a broad base of stakeholders.

The first example is The Nature Conservancy's recent Conservation by Design (CbD) 2.0 approach. In the summary document, four major advances are highlighted: 1) explicitly consider linkages between people and nature, 2) design interventions focused on creating systemic change, 3) integrate spatial planning with the development of new conservation strategies and 4) robustly draw upon and build the evidence base for conservation. The first two points are critical and consistent with what was proposed in the mental model/SDM approach. The inclusion of stakeholders early in the process and recognition of the link between humans and nature is critical if conservation programs are to be successful long term. Creating this linkage early in the process can lead to systemic change. If socioeconomic concerns are addressed early in the process the opportunity for support is greatly broadened.

The basic steps in CbD 2.0 are:

- Identify Challenges and Goals,
- Map Strategies and Places,
- Finalize Outcomes and Develop Measures,
- Take Action, and
- Evaluate and Adapt

In the case of the GRSG, the first step has largely been determined – the challenge is to maintain GRSG populations, and the primary threats have already been identified. CbD 2.0 does not specifically enlist mental models, but it does indicate close collaboration with key stakeholders to analyze evidence to describe current and predicted future situations. That description could easily take on a mental model format.

The second example is a document developed cooperatively by several eastern Oregon Soil and Water Conservation Districts (SWCD) and the U.S. Fish and Wildlife Service (USFWS). The title is *Guide for Completion of a Site-Specific Plan (SSP) Under the Following: Greater Sage-Grouse Programmatic Candidate Conservation Agreement with Assurances for Private Rangelands*. The title may be long, but the simple intent is to provide a playbook for implementing the Candidate Conservation Agreement with Assurances (CCAA) program we referenced in the body of this guide.

The steps outlined in this document are also consistent with what we have proposed in the Guide:

- Accurately identify the problem (or threat) through information gathering;
- Determine conservation objectives that will reduce/resolve the threat;
- Identify, evaluate and select options. Incorporate SMART principles (specific, measurable, achievable, realistic, time-specific) in conservation measure development;
- Implement conservation measures;
- Monitor progress and evaluate results; and
- Adapt and adjust the plan.

Mental models are inherent to SSP development because the CCAA effort was built around the threat-based models we described in the main body of this document. And one stated purpose of developing a SSP is to provide a communication tool among landowners, USFWS, and SWCD staff.

Both the examples described above place an emphasis on involving stakeholders, facilitating communication, and using a relatively simple, stepwise process for conservation planning.

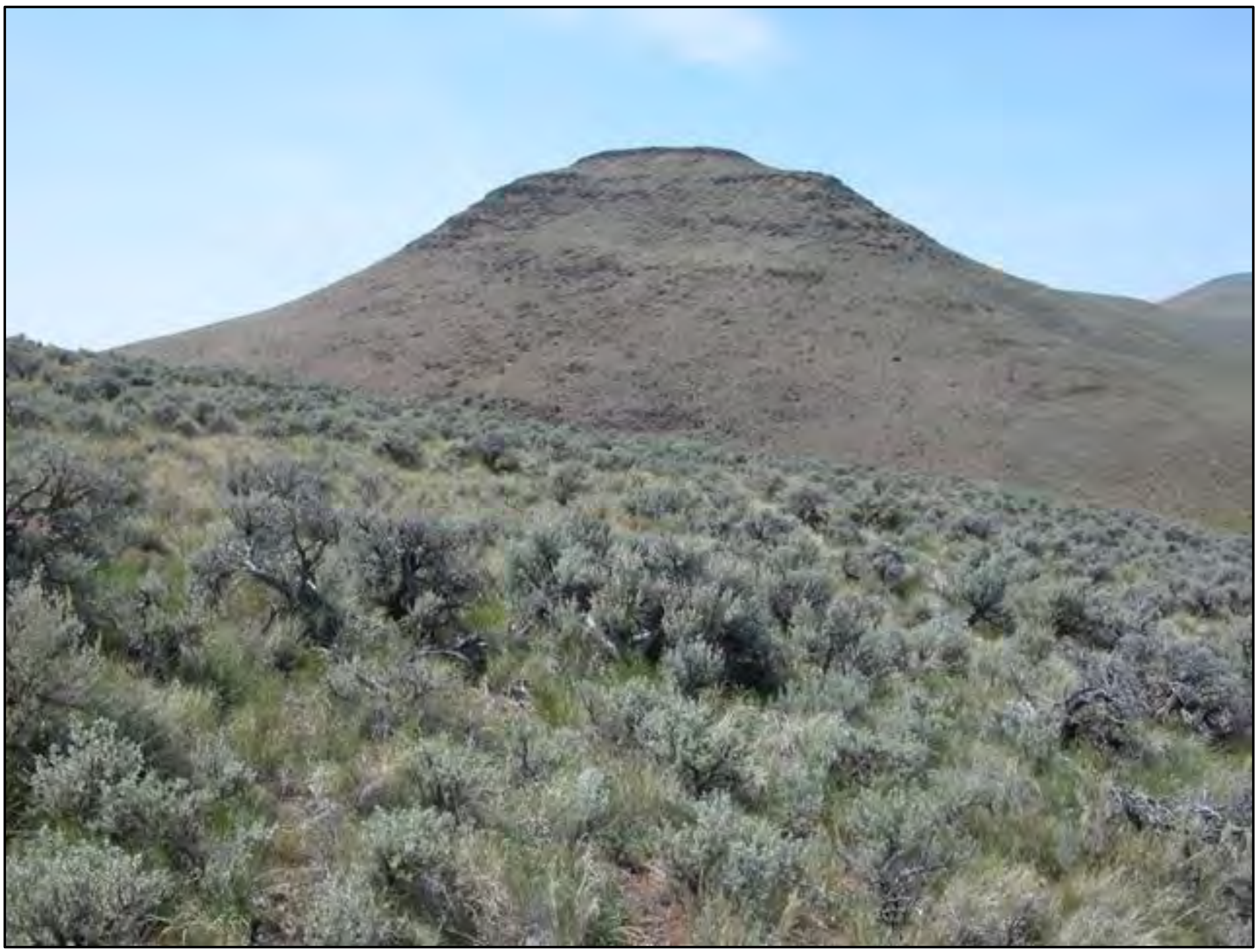
Appendix 2: Example Habitat Conditions

The following figures show examples of each habitat condition within each threat-based model. To demonstrate the potential range of productivity within a habitat condition, several show more than one example. It is important to note that the letter designation pertaining to a site may not always be clear-cut. There are also transitional habitat conditions that are clearly in between two states (either A-C or B-D), and may easily shift towards a more degraded state with disturbance or neglect. It may also work the other way into an improved condition with minimal effort.

Threat: Annual Grass

Example habitat condition: A

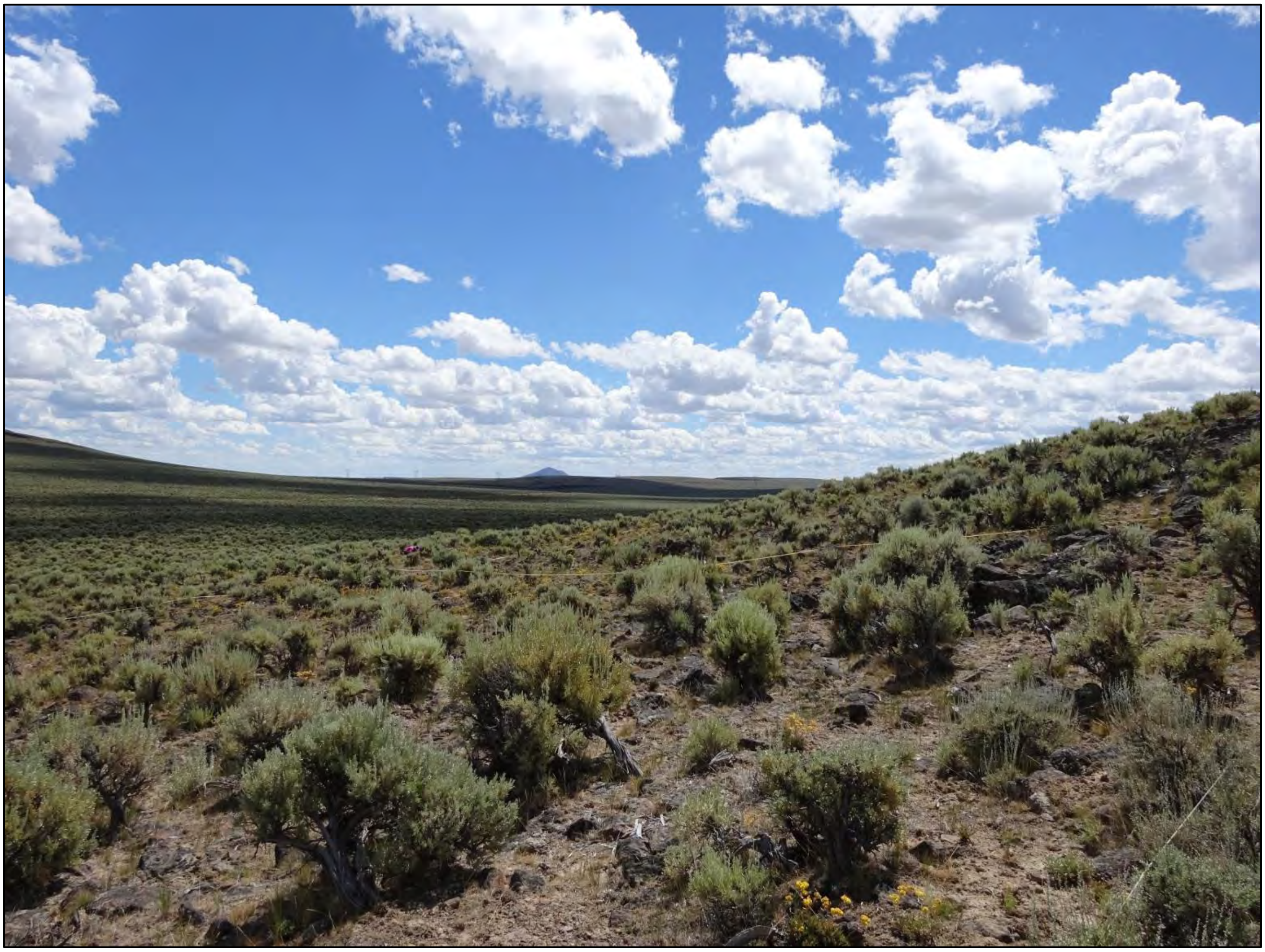
Description: Relatively high cover of sagebrush and perennial grasses; no conifer or annual grass cover. Sagebrush cover is 13%, large perennial grass density is 20 plants/m2, and Sandberg's bluegrass cover is 10%



Threat: Annual Grass

Example habitat condition: A

Description: Annual grass not present, perennial grass cover 3%, sagebrush cover 10% with forbs present



Threat: Annual grass

Example habitat condition: A-C transition

Description: Just under 3:1 annual to perennial grass ratio with 10.8% annual grass cover and 4.2% perennial grass cover, but shrub cover >16% and herbaceous cover is 17.5%

Threat: Annual grass

Example habitat condition: B

Description: No annual grass present, no shrub cover, 48% perennial grass cover with similar herb percent cover

Threat: Annual grass

Example habitat condition: B

Description: 2.5% annual grass cover, 22.5% perennial grass cover, 4% shrub cover with forbs present

Threat: Annual grass

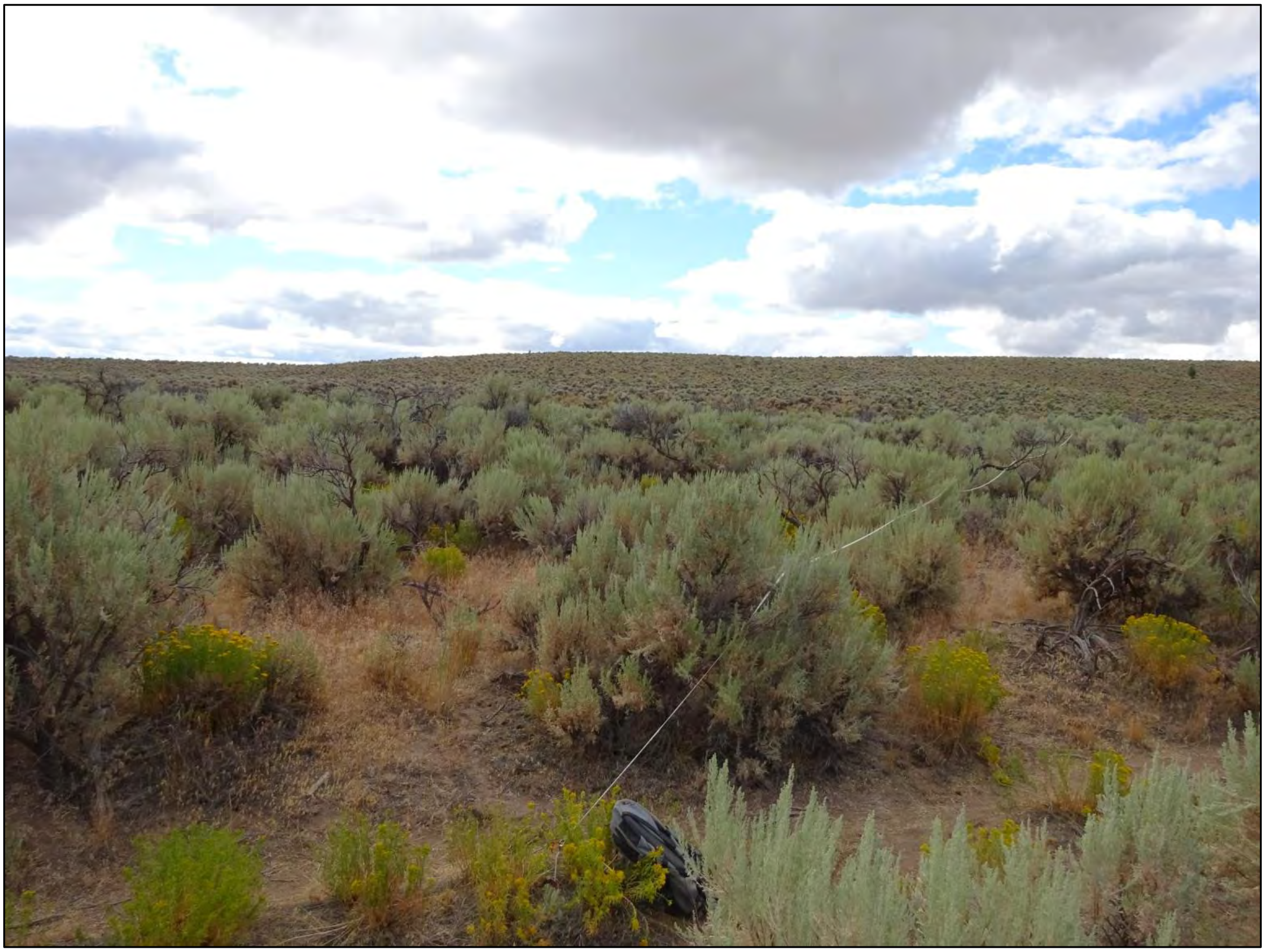
Example habitat condition: B-D transition

Description: >8% annual grass, >3% perennial grass, little to no shrub cover with >11% herb cover

Threat: Annual grass

Example habitat condition: C

Description: 5:1 annual to perennial grass ratio with >8% annual grass cover and <2% perennial grass cover, and 60% shrub cover



Threat: Annual grass

Example habitat condition: D

Description: Nearly 6% annual grass cover and about 1.7% perennial grass cover with no shrub cover

Threat: Annual grass / conifer expansion

Example habitat condition: A

Description: 10% annual grass, nearly 12% perennial grass, 14% shrub cover, short juniper (<6ft) nearby

Threat: Annual grass / conifer expansion

Example habitat condition: B

Description: This is an example where the habitat condition is not so clear-cut. While there appears to be juniper cover, it is slight and the trees appear dead. There is <1% annual grass cover, <1% perennial grass cover, 7.5% shrub cover, and 6% juniper cover

Threat: Annual grass / conifer expansion

Example habitat condition: C

Description: About 11% a grass cover, 2.5% juniper cover, 15% p grass cover, and <1% shrub cover

Threat: Annual grass / conifer expansion

Example habitat condition: D

Description: >13% annual grass, >13% juniper cover, 10% perennial grass, shrubs absent

Threat: Annual grass / conifer expansion

Example habitat condition: E

Description: Junipers nearby and high annual grass cover with no perennial grasses visible and few shrubs

Threat: Conifer expansion

Example habitat condition: A

Description: No conifers present, high abundance of sagebrush and low annual to perennial grass ratio

Threat: Conifer expansion

Example habitat condition: A

Description: No conifers present, low sagebrush density and high perennial grass density with few annual grasses.

Example habitat condition: Top—C; Bottom—D

Threat: Conifer expansion

Example habitat description: C

Description: Top—Conifers nearby, bunchgrasses hard to see, which can mean they are restricted to locations under the sagebrush. Annual grasses are sparse;

Threat: Conifer expansion

Example habitat description: D

Description: Conifers dominate this site with minimal bunchgrass cover and shrubs absent.

Threat: Conifer expansion

Example habitat condition: E

Description: No shrubs or perennial grasses present, high density of conifers and high canopy cover.

Appendix 3: Accepted Metrics for Ecological Habitat Conditions

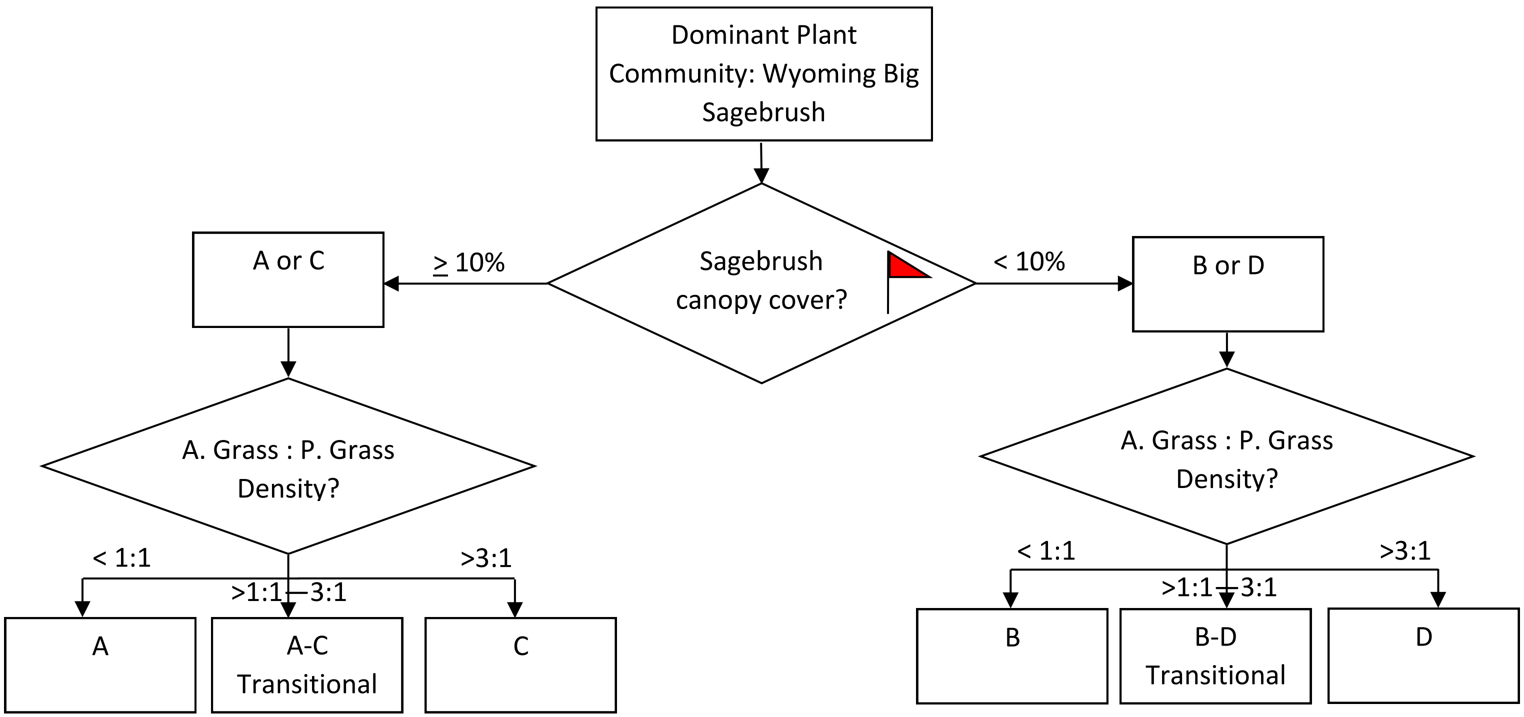


Figure 6. Invasive annual grass threat: Wyoming Big Sagebrush and Associated Low Sagebrush plant communities decision tree to assess ecological state. Users should flag numbers close to the cut-off values for remotely sensed data especially for sagebrush canopy cover (SageCon 2015).

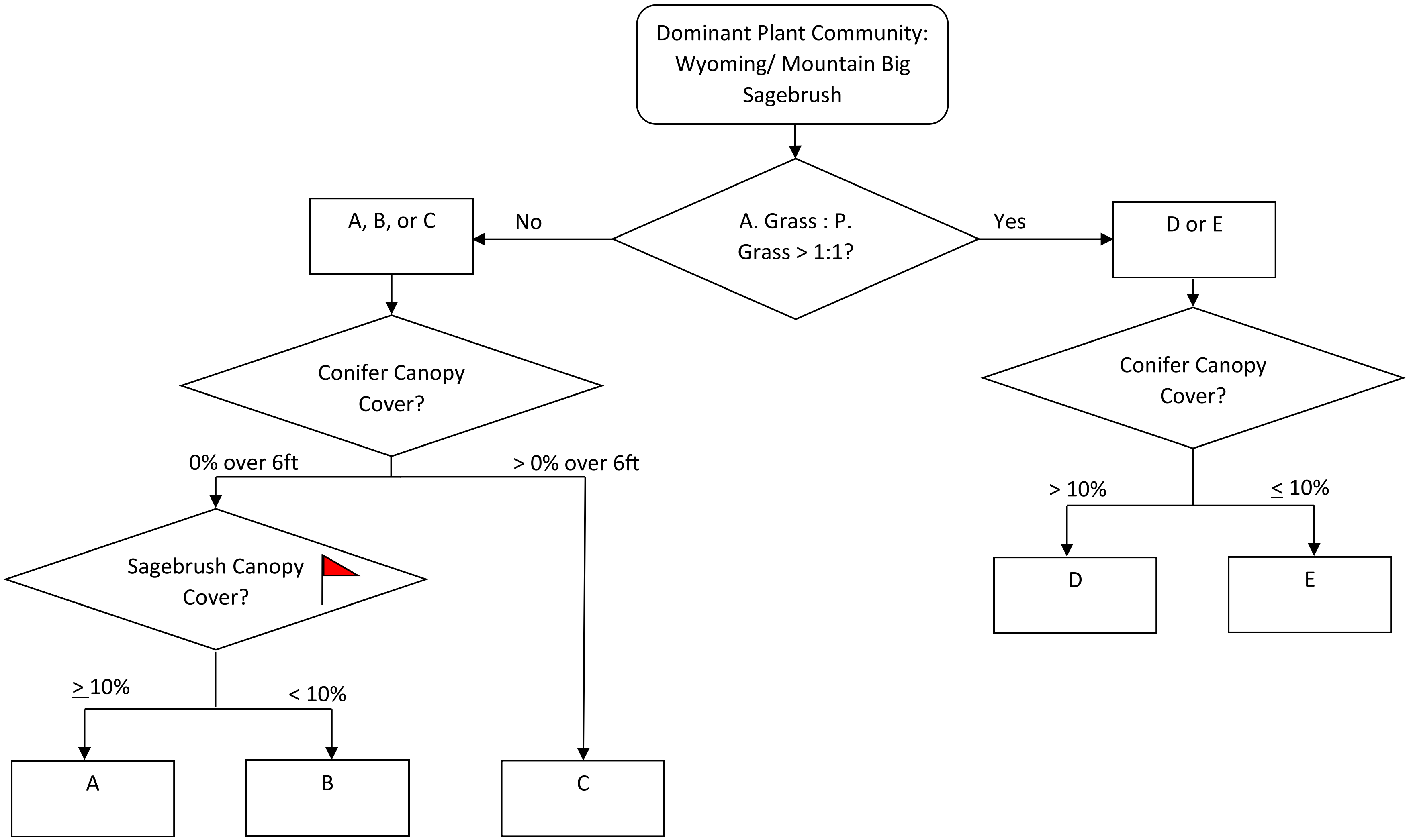


Figure 7. Invasive annual grass / conifer expansion threat: Wyoming or Mountain Big Sagebrush and Associated Low Sagebrush plant communities decision tree to assess ecological state. Users should flag numbers close to the cut-off values for remotely sensed data, especially for sagebrush cover (SageCon 2015).

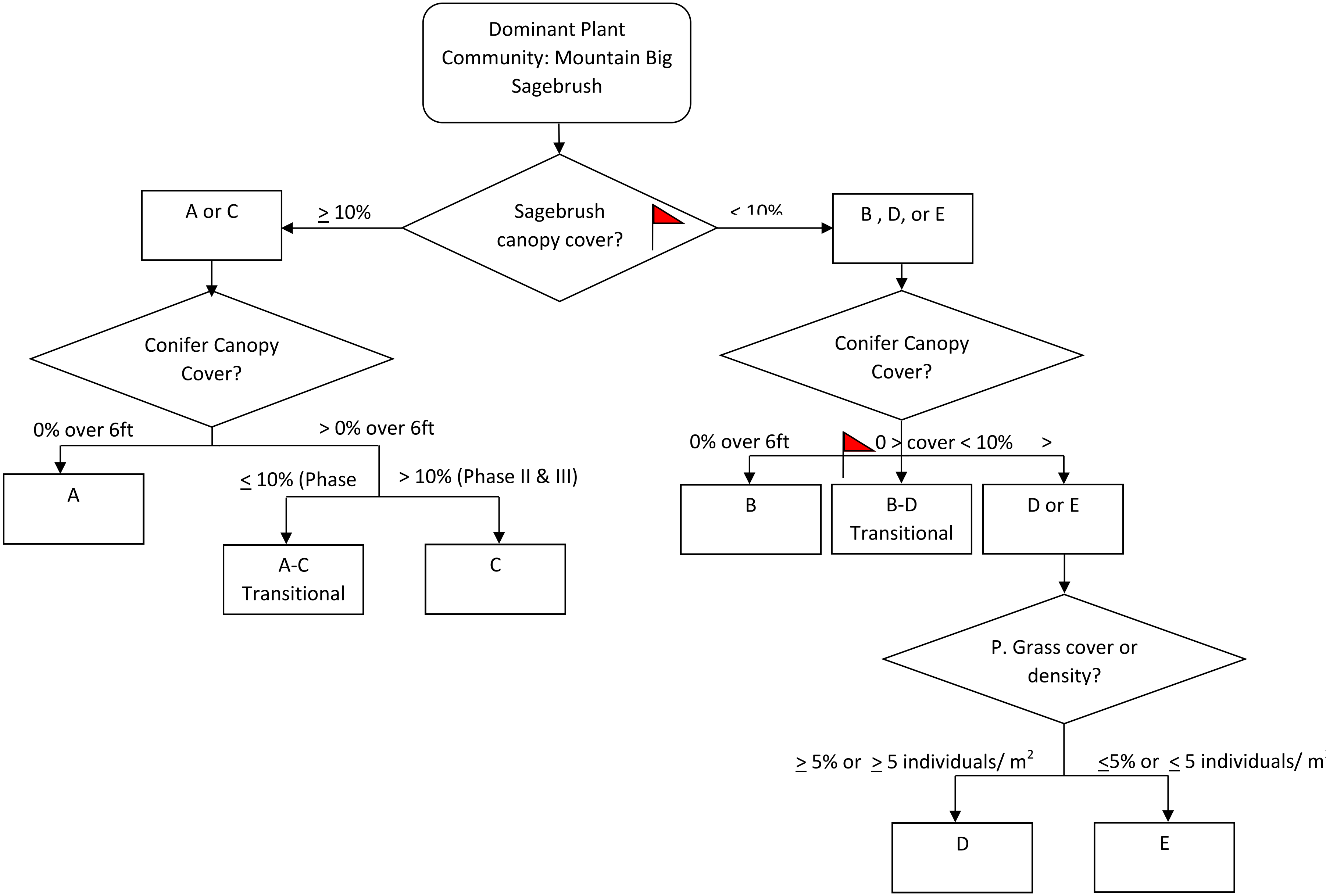


Figure 8. Conifer expansion threat: Mountain Big Sagebrush and Associated Low Sagebrush plant communities decision tree to assess ecological state. Users should flag numbers close to the cut-off values for remotely sensed data, especially for sagebrush cover and if B-D Transitional State is determined. Phase I and II refer to the following: Phase I, trees are present but shrubs and herbs are the dominant vegetation that influence ecological processes on the site; Phase II, trees are co-dominant with shrubs and herbs and all three vegetation layers influence ecological processes on the site; and Phase III, trees are the dominant vegetation and the primary plant layer influencing ecological processes on the site (SageCon 2015).

VEGETATION MAPPING ACCURACY ASSESSMENT

COMPARISON OF HABITAT CONDITION MAPPING METHODS AND PRODUCTS

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Introduction

Managing ecologically-based threats to GRSG habitat (e.g., exotic annual grasses, wildfire, conifer expansion) involves:1) determination of the presence of specific threats, 2) understanding how management can mitigate those threats, and 3) relating dynamic plant community conditions to the habitat needs of GRSG.

Threat-based habitat models of plant community dynamics have been used as a tool to help inform these tasks in the development of Candidate Conservation Agreements with Assurances (CCAAs) for greater sage-grouse in Oregon. Identification of habitat conditions is largely based on recognition of easily identifiable qualitative indicators of plant communities and associated threats (e.g., presence of conifers, abundance of annual grasses relative to large perennial grasses). While useful at small to mid-scales, visual and qualitative mapping becomes logistically limiting in larger landscapes and could be enhanced by the use of various remote sensing platforms to determine threats and membership in specific habitat conditions. The accuracy and precision of available remote sensing platforms for those tasks is currently not known.

Objectives

Working in conjunction with federal and state collaborators, The Nature Conservancy (TNC) used remotely-sensed vegetation coverages to reclassify data into habitat conditions at the 30 m scale for a 50,000 acre study area in southeast Oregon. The study area was comprised of sagebrush rangeland experiencing ecologically-based threats including exotic annual grasses and expanding conifer. Data layers of cover of major vegetation functional groups in the study area were generated by three different remote sensing/data manipulation techniques. These values were then used to classify 30 m pixels based on threats present (annual grass, annual grass + conifer, or conifer) and vegetation habitat condition presented in Guide 1. The objective was to compare results of these three platforms to ground-based data to determine extent of agreement of remotely sensed data with ground-based data.

Background

Habitat condition models have been created by the ARS to inform the management of rangelands. Recently, these models have also been adapted for use in both the Multiple County CCAAs and the Habitat Quantification Tool (HQT) that will be used in the Greater Sage-Grouse Habitat Mitigation Protocol. These threat-based models define different ecological habitat conditions for each habitat classified. Figure 1 shows an example of attributes within the conceptual models separated into quantifiable breakpoints used by in HQT; these metrics were used to interpret the remote sensing data (Appendix 1 is a table containing the accepted habitat condition metrics).

Within each habitat there are five or six habitat conditions that have been defined as ranging from “good” ecological condition (dominated by native species of bunch grass and sagebrush) to “poor” condition (degraded by encroaching/invasive species such as juniper and annual grasses). These models are useful for rapid assessment of habitat conditions and have the potential to integrate with more complex habitat assessment methods such as the Habitat Assessment Framework (Stiver et al. 2010).

Mitigation banking and implementation of conservation agreements will require mapping of these habitat conditions across large geographies. Data collected through remote sensing methods will be paramount in these broad-scale mapping efforts. This project evaluated three remote mapping methods that have been implemented across 100,000 acres in western Harney County, Oregon. Data produced by all three methods were evaluated for accuracy relative to the habitat conditions within each threat-based model, and for their cost. This information will help land managers determine the trade-offs amongst these products when determining which may be best for a particular project. The three methods include:

- **Open Range Consulting (ORC):** Continuous maps of functional land cover attributes including perennials, annuals, sagebrush, trees and bare ground at 1% cover intervals and 1-m resolution (Sant et al. 2014).

- **Collin Homer and Cameron Aldridge (USGS):** Plots of continuous vegetation cover data are used to create nine products including: vegetation estimates at a 30 m pixel/resolution for all shrubs (can separate sagebrush from big sagebrush), all herbaceous, annual herbaceous, bare ground, litter, and estimates for shrub and sagebrush height (Xian et al. 2015).
- **Institute for Natural Resources (INR):** Species lists along with cover values are imputed to 30 m pixels using Random Forest Nearest Neighbor (RFNN) methodology (Nielson and Noone 2014; Nielson et al. 2014). Maps are produced from plot data of existing vegetation and environmental variables, combined with remotely sensed imagery. Any vegetation attribute measured in the supporting plot data can be mapped.

Not all data providers mapped each vegetation variable called out in the threat-based models, and some have mapped additional variables that aren't used directly. For example, ORC provided total shrub cover but did not call out sagebrush specifically as described in the models. Therefore, we tested the ORC total shrub cover dataset. Similarly, USGS did not provide tree cover data, so that test was confined to the ORC and INR datasets. Annual and perennial grasses are not called out as standalone variables in the threat-based models, but because they are used in the annual/perennial calculation, we tested both. This provided specific feedback on which variable may be causing issues in cases where that metric is performing poorly.

Data Preparation Methods

Open Range Consulting

ORC delivered five floating point raster datasets to TNC in January of 2016. All five were aggregated from their 1-meter source data to the 30 m snap raster provided by TNC. The datasets included:

- Bare ground: Values represent percent cover of bare ground within each cell. Values across the study area range from 3 – 100% with a mean of 46.89%.
- Cheatgrass: Values represent percent cover of annual invasive grasses within each cell. Values across the study area range from 0 – 16.08% with a mean of 2.29%.
- Herb: Values represent percent cover of annual and perennial herbs within each cell. Values across the study area range from 0 – 58.71% with

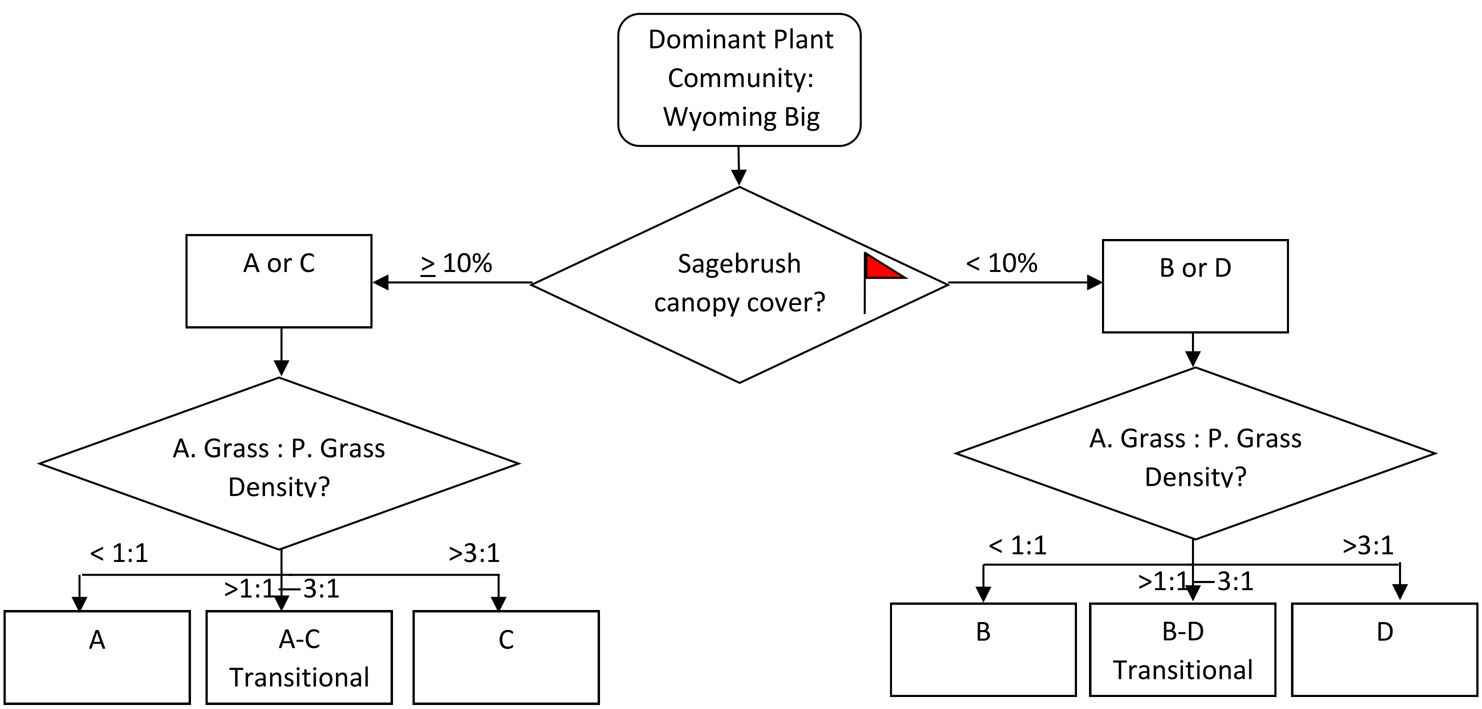

Figure 1. Wyoming Big Sagebrush and Associated Low Sagebrush plant communities decision tree to assess habitat condition. Users should flag numbers close to the cut-off values for remotely sensed data especially for sagebrush canopy cover (SageCon 2015). See Appendix 3 in Guide 1 for all habitat conditions under each threat-based model.

a mean of 8.83%.

- Juniper: Values represent percent cover of trees within each cell. Values across the study area range from 0 – 85% with a mean of 1.01%.
- Shrub: Values represent percent cover of shrubs within each cell. Values across the study area range from 0 – 37.55% with a mean of 11.65%.

Perennial herb cover was calculated by subtracting the cheatgrass raster from the herb raster. Values of this calculation range from -5 – 54.62% with a mean of 6.54%. Negative values occurred in 137 pixels (or 0.06% of the project area), and were reclassified to '0' so subsequent calculations would not be affected. This output was saved as perennial_herb_GTE0, with values ranging from 0 – 54.62% with a mean of 6.53%.

The ratio of annual to perennial herbs was calculated by dividing the cheatgrass raster by the perennial_herb_GTE0 raster. This output was saved as ann_per_ratio, with values ranging from 0 – 69.8 with a mean of 0.35.

Where 'No Data' values occurred in portions of the classed annual/perennial herb, causing division by '0' errors, we took two actions to appropriately modify those values. First, all 'No Data' cells were added into the 3:1 bin. Second, the subset of those cells without annuals were then placed in the 'No grasses' category.

Four of these datasets were then reclassified to match the breakpoints as specified in the threat-based models. Perennial herb cover was reclassified into two bins: 0 – 5% and > 5% (perennial_herb_2class). Total shrub cover was classified per the sagebrush cover breaks called for in the models: 0 – 10% and > 10% (shrub_2class). The annual/perennial herb layer was binned into 4 classes: No grasses; <= 1:1; 1:1 – 3:1; and > 3:1 (ann_per_ratio_4class). Tree cover was broken into three classes: No trees; 0 – 10% canopy cover; and > 10% canopy cover (tree_3class). See Figure 2 for ORC dataset examples.

U.S. Geological Survey

The USGS has produced four sage-steppe related vegetation datasets across 85,323,155 acres of the western US. The versions we extracted to the footprint of our study area were released on 10/30/2015 and include:

- usgs_nw_all_sage_103015_est: Values represent percent cover of all sage species within each cell. Values across the study area range from 0 – 38% with a mean of 18.58%.
- usgs_nw_big_ sage_103015_est: Values represent the percent cover of big sage within each cell. Values across the study area range from 0 – 33% with a mean of 15.63%.
- usgs_nw_ann_herb_103015_est: Values represent percent cover of annual species within each cell. Values across the study area range from 0 – 51% with a mean of 26.64%.
- usgs_nw_herb_103015_est: Values represent percent cover of all herbaceous species within each cell. Values across the study area range from 1 – 61% with a mean of 30.51%.

It should be noted that the USGS data were published before we received information from INR or ORC, so the USGS data were used to create the snap raster provided to the other researchers. Therefore, all other data used in these tests were aggregated and snapped to the USGS grid in the USGS projection (USA Contiguous Albers Equal Area Conic USGS version). All intermediate and final datasets were also snapped to the USGS grid and utilized the USGS projection.

Sage cover, annual herb cover and herb cover were selected for our testing. Annual herb cover was subtracted from herb cover to produce perennial herb cover. Annual herb cover was then divided by perennial herb cover to create the annual/perennial ratio with values ranging from 0 – 44, with a mean of 1.45. The annual/perennial herb layer was binned into 3 classes: <= 1:1, 1:1 – 3:1; and > 3:1 (ann_per_ratio_3class).

'No Data' values occurred in the portions of the classed annual/perennial herb layer without perennial forbs (division by '0'). These cells were added into the 3:1 bin as the lowest cover value of annuals across the study area was 1%. This did result in the road along the northern edge of the DSL parcel being added to the > 3:1 class, but as plots will be excluded from a buffer around the road, this will not affect the accuracy values of the dataset.

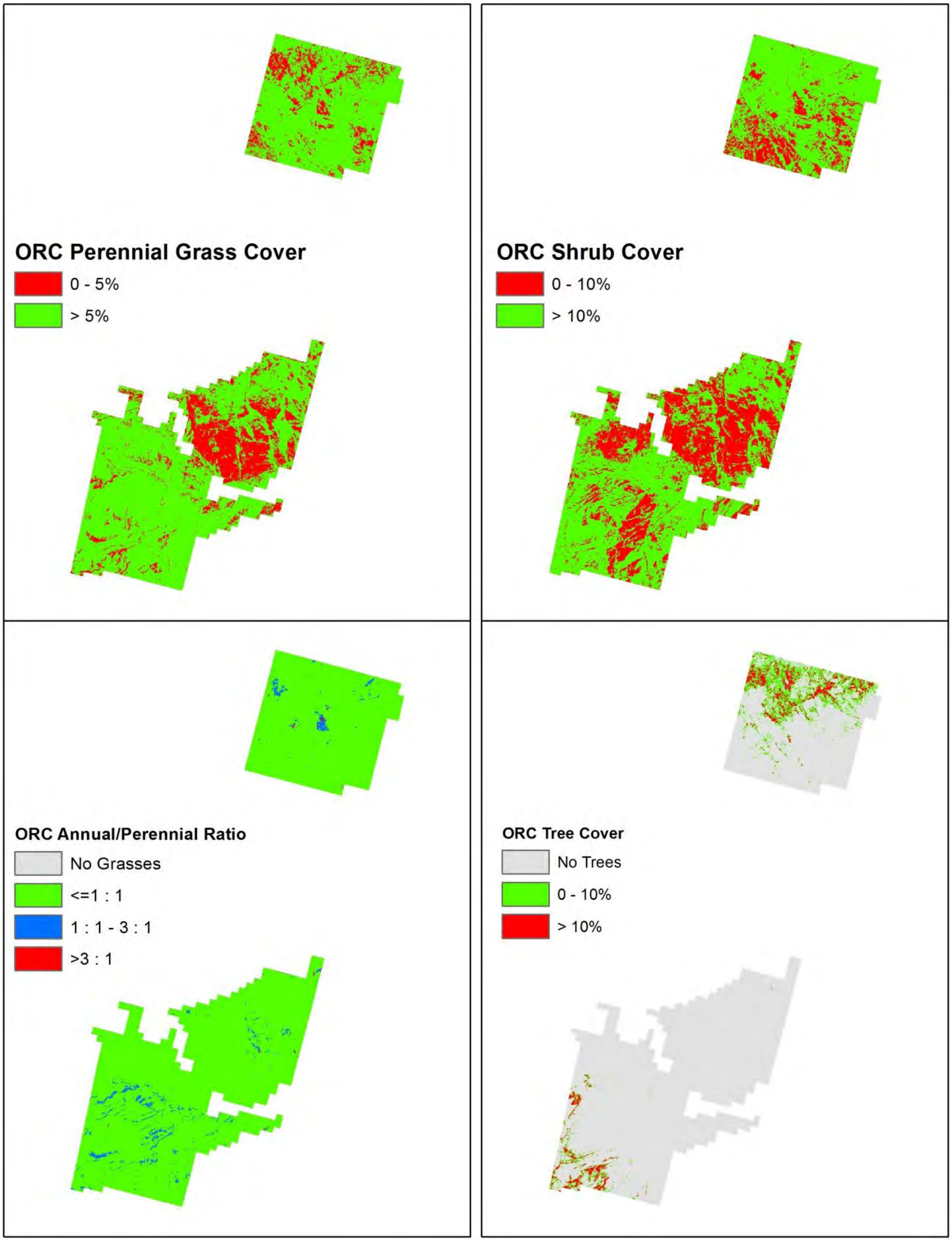

Figure 2. Example datasets from *Open Range Consulting*.

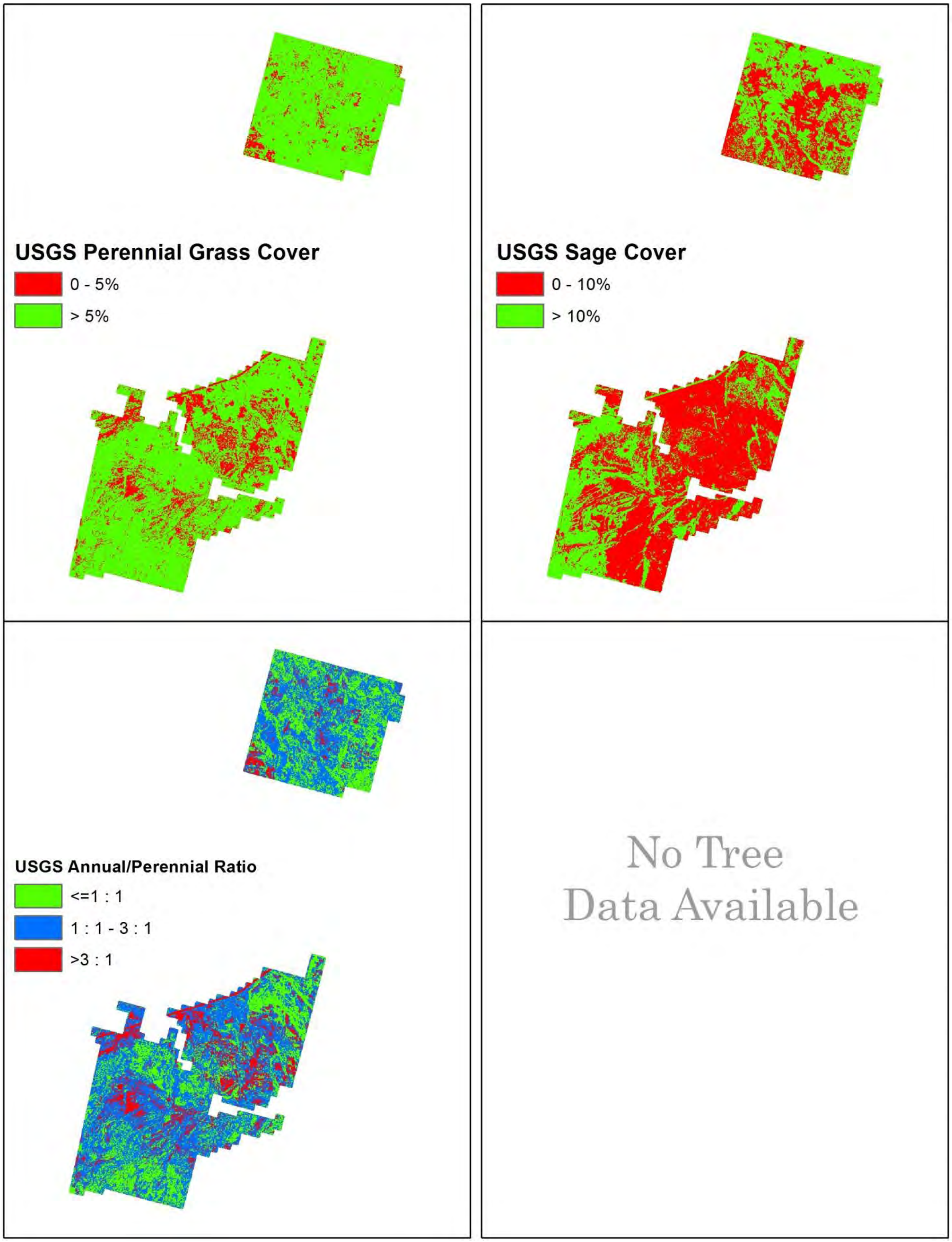

Figure 3. Example Datasets from The U.S. Geological Survey.

The two remaining datasets were then reclassified to match the breakpoints as specified in the threat-based models. Perennial herb cover was reclassified into two bins, 0 – 5% and > 5% (perennial_herb_2class). Sage cover was also classified per the sagebrush cover breaks called for in the models: 0 – 10% and > 10% (sage_2class). See Figure 3 for USGS dataset examples.

Institute for Natural Resources

INR prepared their data by aggregating their source data to our 30 m grid and binned them per the habitat condition class breaks (see Figure 4 for INR dataset examples). The datasets include:

- exotic_annual_grass_perennial_grass_ratio: Values represent the class break bins specified in the models for the annual/perennial ratio: No grasses (0); <= 1:1 (1); 1:1 – 3:1 (2); and > 3:1 (3).

- exotic_annual_grasses_cover: Values represent the percent cover of annual grasses within each cell. Values across the study area range from 0 – 82.14% with a mean of 5.88%.

- Juniper_cover: Values represent the class break bins specified in the models for juniper cover within each cell: No Trees (0); 0 – 10% cover (1); and > 10% cover (2).

- perennial_grasses_gr_5 : Values represent the class break bins specified in the models for perennial grass species within each cell: 0 -5% cover (0) and > 5% cover (1).

- Shrub_2_class : Values represent the class break bins specified in the models for the cover of all sage species within each cell: 0 -10% cover (1) and > 10% cover (2).

The previous steps classified all datasets from all providers into identical bins with identical class values to make comparison possible.

Comparison of Datasets

With all of the data prepped, the next step was to identify zones of similarity and dissimilarity across all three methods for each variable. A mask was constructed to limit the comparison analyses to areas 1) removed a distance from parcel boundaries, and 2) away from roads and other non-vegetated linear features. Both of these zones are more likely to have impacts from maintenance activities that could create localized variations in vegetation condition that might skew values for the associated 30 m pixels.

The test area polygons were buffered (-30) meters to remove the portions near the parcel boundaries. The area surrounding the buildings on the research station was also removed.

Roads data from the BLM Ground Transportation Dataset were superimposed over NAIP imagery zoomed to a 1:10,000 scale. Arcs were added for roads and other linear features visible at this scale that did not appear in the BLM data. Some arcs from the BLM dataset were also moved to better reflect ground condition.

The edited arcs were then buffered by 15 m, for a total width of 30 m to match the scale of our raster cells. The road buffers were then used to erase roaded portions of the modified test area polygons.

Finally, the polygons were converted to a raster, with erased portions classified as ‘No Data’ and all other areas set to a value of ‘1’. All subsequent data comparisons were constrained to the areas classified as ‘1’ in the Analysis Mask raster (Figure 5). The ArcGIS ‘Cell Statistics’ tool was then used to compare each of our classed variables across all three methods. Two statistics were calculated for each variable: Variety and Majority. The Variety metrics measured the degree of dissimilarity between the datasets while Majority identified the class value—if any—that was present in more than one dataset.

For the three binned variables tested (perennial grass cover, annual/perennial ratio, and shrub cover), the Variety metrics were summed to see which portions of the test area showed the greatest dissimilarity across all three variables. 9.5% of the unmasked test area showed complete agreement among all methods and variables. Another 28% only showed small dissimilarities. Less than 1% of the area fell into the maximally dissimilar class. See Figure 6 for a Comparison of Mapped Variable Classes across Methods. Annual grass cover was produced by all three methods.

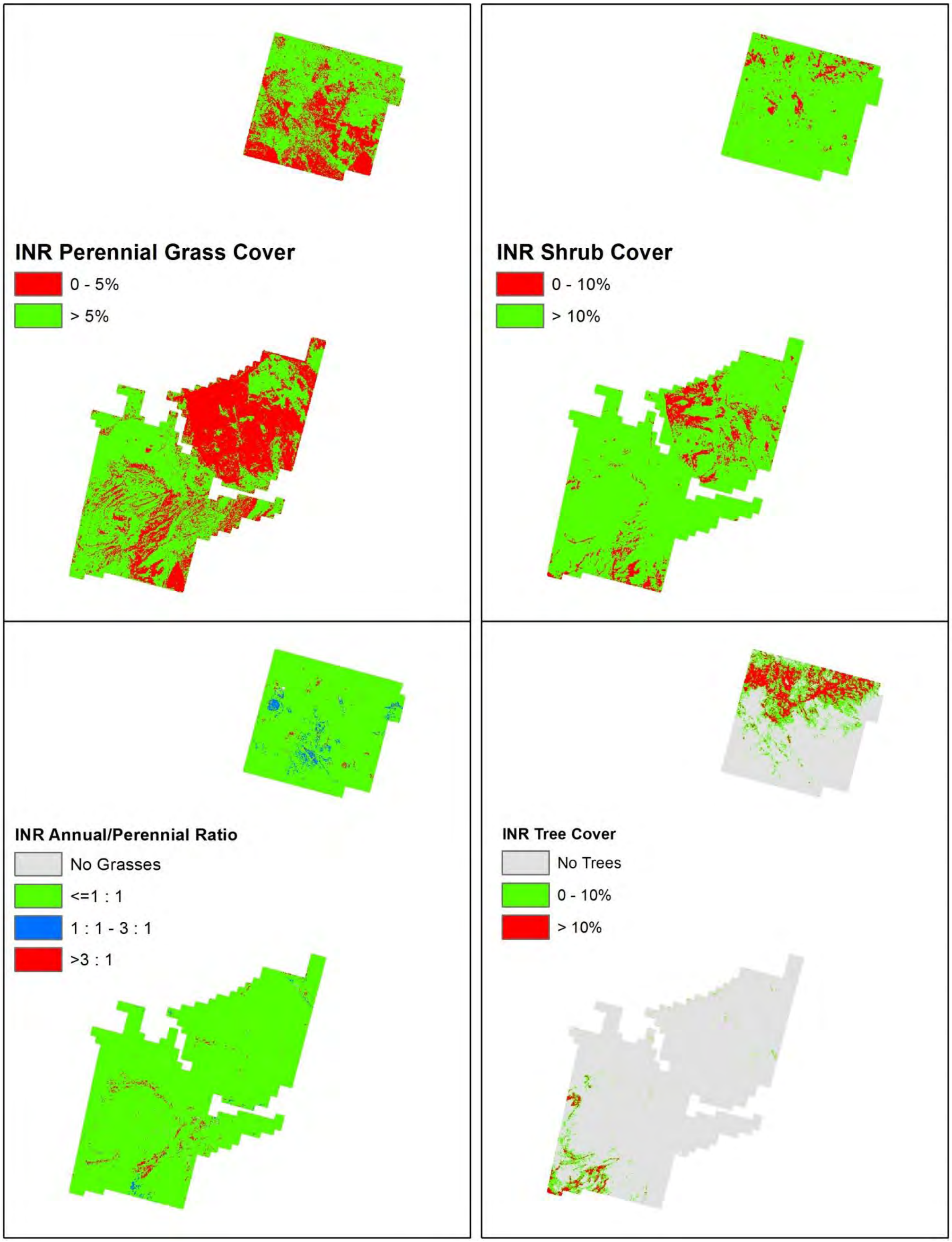

Figure 4. Example Datasets from *The Institute For Natural Resources*

Though not called out separately in the habitat condition metrics, given the importance of annual grass as an indicator of sage-steppe health, plots should also be placed along the dissimilarity gradient among the methods to see which method performed best. To calculate this dissimilarity, the Standard Deviation statistic was generated from the raw annual grass cover values using the Cell Statistics tool (see Figure 7 for annual grass dissimilarity). Only two methods produced tree cover, so those were tested separately from the other variables using the Variety metric with the Cell Statistics tool (see Figure 8 for tree canopy dissimilarity).

Selection of Field Sampling Locations

The goal of field plot data collection is to assess the accuracies of each researcher's products while also sampling across the gradient of agreement between researchers. To accomplish both goals entails placing sufficient numbers of plots within each class, strata, and method while simultaneously looking for zones of agreement and disagreement between methods.

As the data were all aggregated to 30 m raster pixels, the plots should also correspond to the size and shape of those cells.

After removing cells within exclosures along roads and property boundaries, approximately 219,000 potential plot locations remained. Those cells were converted to points (using the cell centroid) and the points were attributed with all the class assignments from the 11 individual strata datasets. Each cell was also assigned a code indicating the level of agreement between researchers across all strata and classes. A random number generator was then used to assign each point a random integer ranging from 0 – 100,000.

As each point represents 11 individual strata and all classes within each, every plot collected could potentially be used to assess the accuracy of all datasets. However, for this to be true, the plots would have to capture a proportional representation of each class within each of the 11 strata datasets. Increasing this complexity, we also wished to sample across the gradient of agreement between the datasets. The complexity of identifying a least set that satisfied both sets of criteria precluded a manual process. We therefore used MARXAN optimization software to find potential solutions.

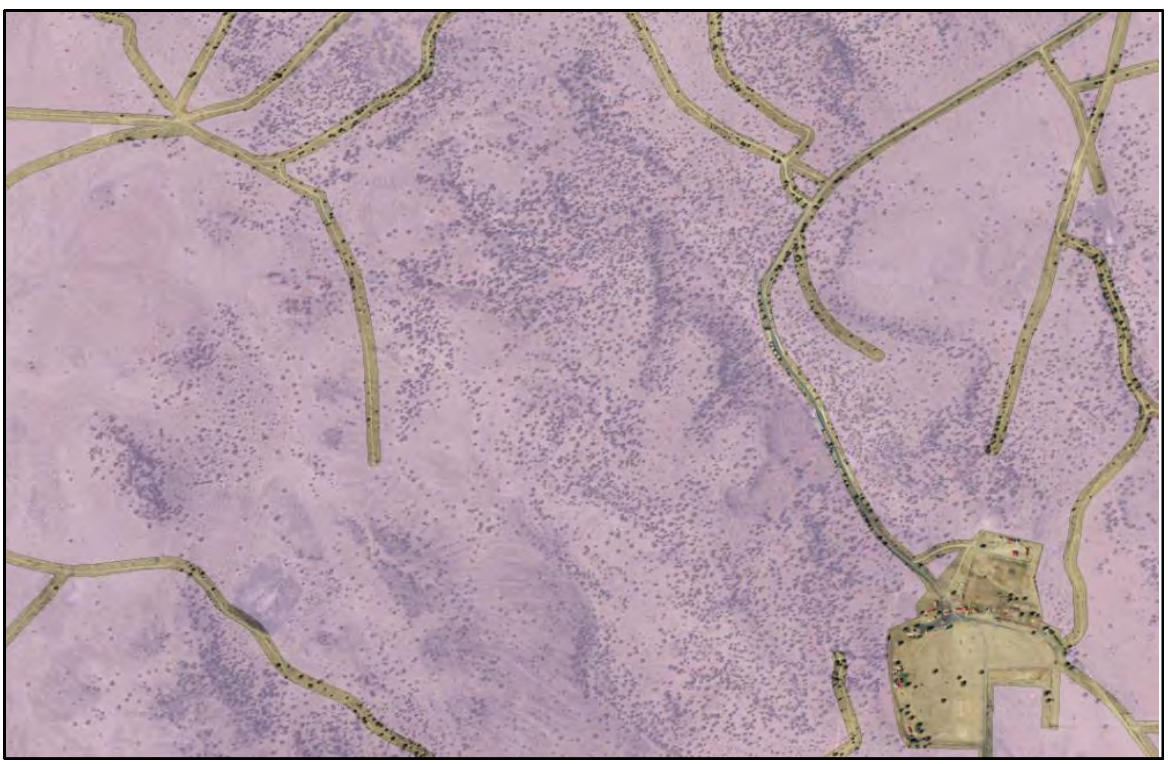

Figure 5. Analysis Mask polygon from the vicinity of the ARS research station.

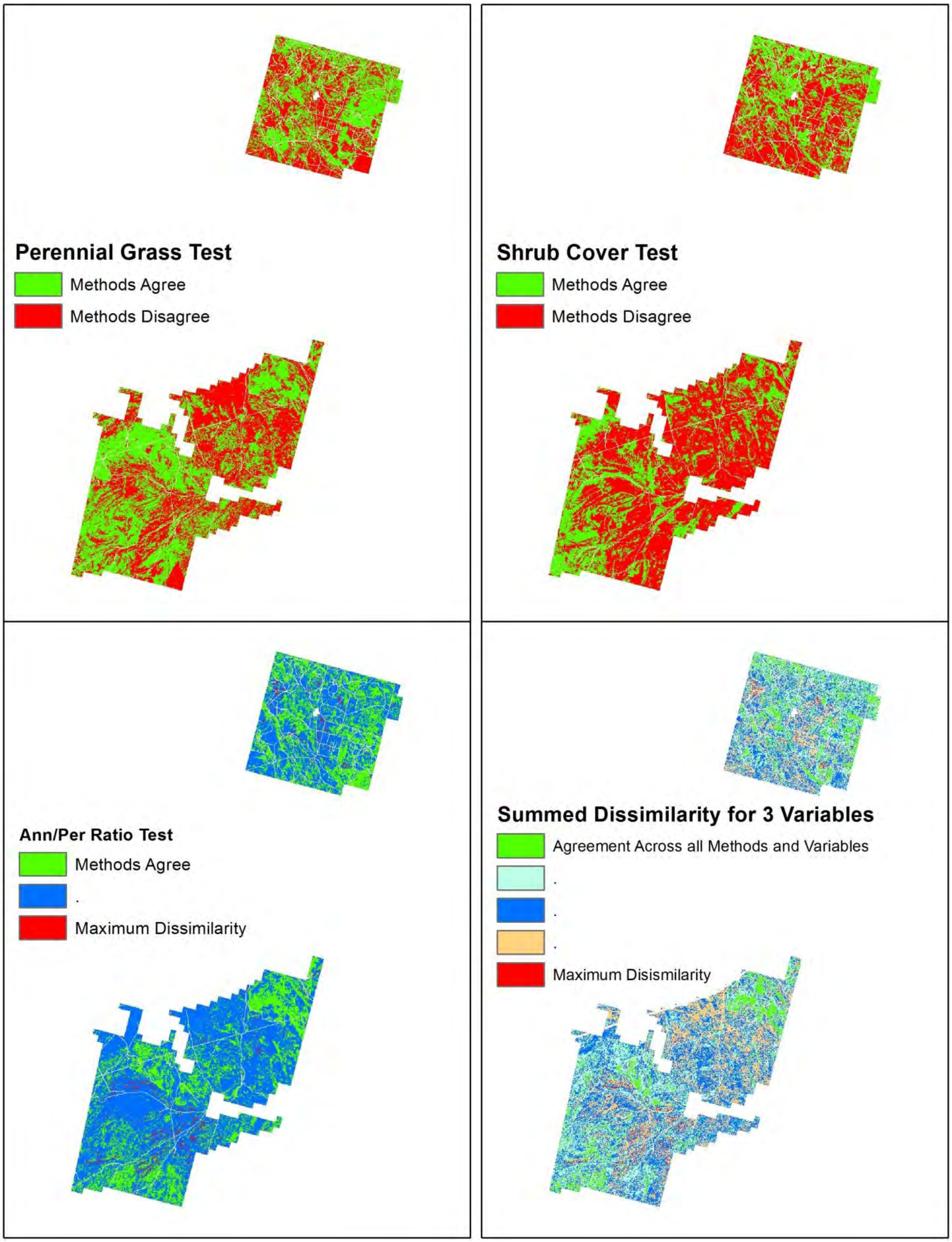

Figure 6. Comparison of Mapped Variable Classes across Methods.

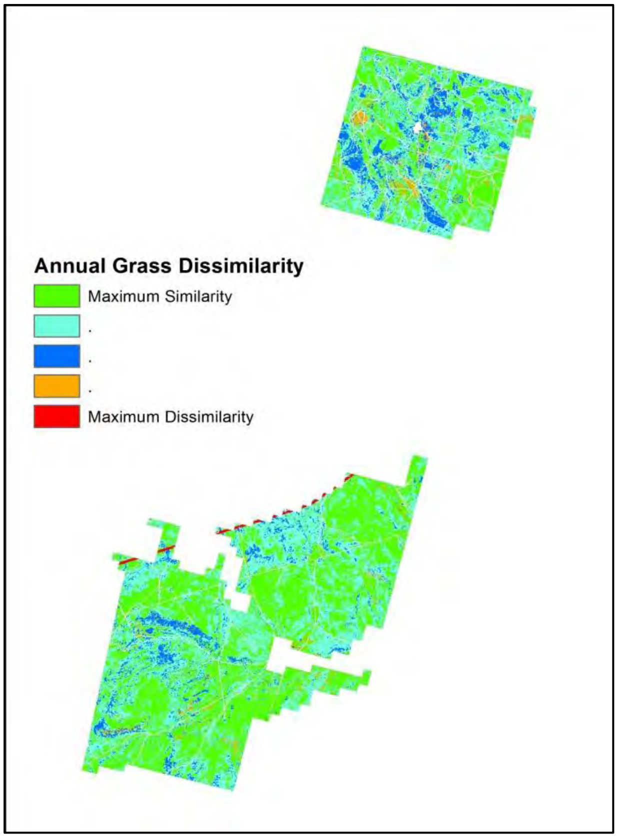

Figure 7. Annual grass dissimilarity.

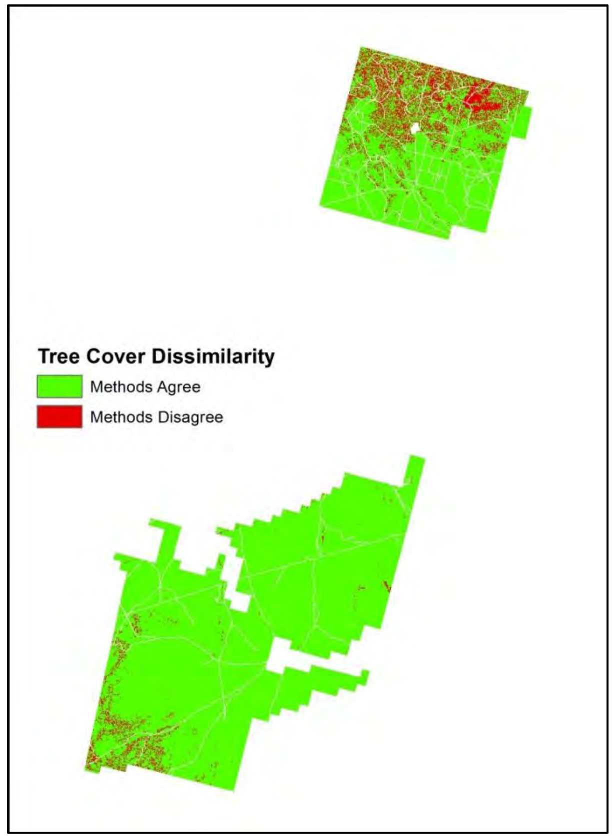

Figure 8. Tree canopy dissimilarity.

The MARXAN optimization algorithm searches for the lowest 'cost' set solution that will meet 'goals' for all 'targets'. 'Target' refers to the entities that MARXAN is attempting to capture in the solutions. 'Goal' refers to the desired proportion, or total abundance, of each target represented in the solutions. 'Cost' refers to any set of economic, social and environmental uses and protections that are present in a particular geography that might be considered during planning.

MARXAN requires that all data be attributed to polygonal assessment units (AUs) as they are the basic units of analysis for the optimization algorithm. In this case, our 30 m cells were used as AUs. Our 'Targets' were defined as all combinations of classes and strata, as well as the level of agreement between researchers across all classes and strata. 'Goals' were set for each target as the number of plots required for proportional representation, and iteratively adjusted through the analytical process so as not to exceed 200. 'Cost' was calculated in two ways: each cell was assigned the same cost (100) so that no cell would be favored over another and the solution would trend towards the smallest possible footprint; and a second set of analyses were performed using the random integer assigned to the cell as the 'cost'. This could potentially increase (slightly) the number of plots required to meet our goals, but it would also more thoroughly randomize the selection of AUs.

MARXAN finds reasonably efficient solutions to optimization problems (Possingham et al. 2000; McDonnell et al. 2002) by minimizing the 'Objective Function', or the sum of:

- The total cost for all selected AUs,
- Penalty factor, which is the penalty for not meeting stated goal levels,
- Length of the outer perimeter of the solution set, and
- "Boundary Length", which correlates with fragmentation (this parameter is optional, and was not used in these analyses).

See Table 1 for sample MARXAN goals, targets, and results.

In general terms, MARXAN works to minimize the overall footprint of selected assessment units while meeting user-defined goals. The MARXAN algorithm begins by selecting a random set of AUs (i.e., a random solution). The algorithm then iteratively explores improvements to this initial solution, as measured by the value of the 'Objective Function', by randomly adding or removing AUs.

The solution for each iteration is compared with the prior solution, and the iteration with the lower 'Objective Function' value is accepted. This process repeats for the user-specified number of iterations. The algorithm uses a method called simulated annealing (Kirkpatrick et al. 1983) to search for optimal solutions, thus greatly increasing the chances of converging on a highly efficient solution.

Tree canopy cover was only mapped by two of the three researchers. As these vegetation products are produced from aerial/satellite imagery, and tree canopy cover shields the herbaceous and shrub strata from view, we decided to exclude areas mapped as 'Greater than 10% tree canopy cover' in either tree dataset from selection by MARXAN. This ensured that plots selected for their herbaceous and shrub characteristics were fairly assessed.

In most cases, MARXAN targets are not evenly distributed across the landscape. This can often cause MARXAN to concentrate on a few areas that are particularly target-rich while avoiding areas that are less so. To ensure that the selected sites were evenly distributed across the study area, we broke the geography into three separate MARXAN regions, with goals allocated proportionally within them. Region 1 was the ARS Research station. Region 2 was the northern half of the DSL parcel and Region 3 was the southern half. Each had roughly the same number of cells available to MARXAN for selection and each was run as an independent MARXAN analysis.

Three MARXAN runs were performed for each region. 20,000,000 iterations and 10 restarts were used for each run within each region. The first run used the random number as the cost and had relatively low goal levels. The second used identical costs for each AU, set at '100', and maintained the goal levels from the first run. These results indicated that substantially more AUs were required to meet the same goal levels using random costs versus equal costs. As MARXAN utilizes Monte Carlo randomization in its selection of AUs, it was deemed counterproductive to try and impose additional randomization on top of that. The third run, therefore, used equal costs with goals adjusted to capture approximately 150 plots, well below the 200 plots that could feasibly be sampled within our budget. The remaining 50 plots were held back for allocation within the heavily treed portions of the study area.

In all, MARXAN selected a total of 150 plots across the study area. These were mapped in GIS and inspected against imagery. Sixteen locations were judged unsuitable for our purposes, either due to inaccessible/rugged terrain or anthropogenic modification, and alternate locations were identified. Alternates were found by selecting all locations with identical attributes to an unsuitable point, sorting those by the random number, and selecting the one

Table 1. MARXAN Targets, Goals and Results.

Target ID	Goal	Target Name	Proportion of Goal Met
1	30	ALL METHODS AGREE	1.00
2	30	MAXIMAL DISAGREEMENT	1.10
10	20	ORC AP RATIO NOGRASS	1.00
11	20	ORC AP RATIO LTE1	4.70
12	20	ORC AP RATIO GT1 LTE3	1.10
13	20	ORC AP RATIO GT3	1.05
20	40	ORC PCLASS 0 5PCT	1.58
21	40	ORC PCLASS GT 5PCT	2.35
31	40	ORC SCLASS 0 10PCT	1.88
32	40	ORC SCLASS GT 10PCT	2.05
40	20	INR AP RATIO NOGRASS	1.05
41	20	INR AP RATIO LTE1	4.70
42	20	INR AP RATIO GT1 LTE3	1.05
43	20	INR AP RATIO GT3	1.05
50	40	INR PCLASS 0 5PCT	2.25
51	40	INR PCLASS GT 5PCT	1.68
61	40	INR SCLASS 0 10PCT	1.23
62	40	INR SCLASS GT 10PCT	2.70
71	27	USG AP RATIO LTE1	2.22
72	27	USG AP RATIO GT1 LTE3	2.52
73	27	USG AP RATIO GT3	1.07
80	40	USG PCLASS 0 5PCT	1.18
81	40	USG PCLASS GT 5PCT	2.75
91	40	USG SCLASS 0 10PCT	2.43
92	40	USG SCLASS GT 10PCT	1.50
		Average Goal Attainment	**1.89**

at the top of the sorted list. In one case, no alternate could be found and that plot was dropped. In two other cases, only one alternate existed. Thus 149 plots were identified for accuracy assessment within the herbaceous and shrub dominated areas as mapped by the three methods.

Overall, utilizing the MARXAN optimization algorithm yielded a highly efficient sample design. At least 781 samples were captured across the 149 plot locations, well distributed across the three methods, their mapped classes, degrees of agreement, and the study geography (see Figure 9 for map of plot locations).

As can be seen in Table 1, some targets were captured well over their goal levels, while others were very close to their goals. This is quite common, and is caused by the uneven abundance of the various targets across the landscape. Targets that are quite abundant, and are represented in a large proportion of AUs are often swept in with cells that are selected for rarer targets.

Lastly, we allocated the final 50 plots among the cells mapped with tree cover. The cells we had held back

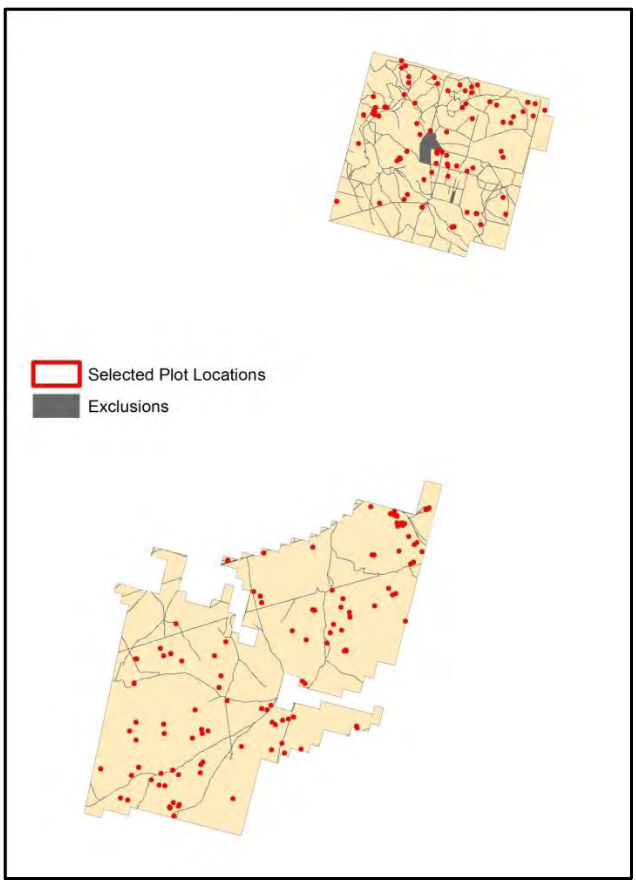

Figure 9. Plot Locations (including tree plots).

from MARXAN analyses were those mapped with 10% or more tree cover in either of our two tree datasets. Five situations were expressed in this subset: Method 1 predicted no tree cover while Method 2 predicted high tree cover; Method 1 predicted low tree cover while method 2 predicted high tree cover; Method 1 and 2 both predicted high tree cover; Method 1 predicted high tree cover while Method 2 predicted low tree cover, and; Method 1 predicted high tree cover while Method 2 predicted no tree cover. Each of these were allocated 10 samples, split (where possible) between the ARS and DSL parcels. However, as the ARS parcel has more treed area, more of these plots fell in that area. Examples of each of the five combinations were identified, sorted by random number with the smallest random numbers selected from within the two parcels.

Field Sampling Methods

Field study sites corresponded with the 200 locations selected via MARXAN's optimization process previously described. Corners of each study site were located using a handheld Trimble GPS (GeoExplorer 6000 Series GeoXT) and temporarily marked. Each study site contained three parallel 20 m long sampling transects spaced 7.5 m apart and 7.5 m from the parallel plot edge. Line point intercept was used to estimate cover of functional groups including live sagebrush, dead sagebrush, live 'other' shrubs, dead 'other' shrubs, large perennial bunchgrasses, Sandberg bluegrass, annual grasses, perennial forbs, annual forbs, bare ground, litter, rock, and microphytic crust. Each transect contained 40 equally spaced sample points (120 sample points per study site). At a point, all intercepts were recorded in the order they occur along a downward projection; the last intercept being ground cover. If present, juniper cover was assessed using line intercept technique along the 20 m transects. Estimated height of all juniper intersecting transects was recorded by height class (0-0.5m, 0.5-2m, 2-4m, >4m). Using ground-based vegetation data, all study sites were classified based on threats present and habitat condition (see Guide

1 and Appendix 1 of this document) following the same class assignments previously described for remotely sensed data.

In cases where scattered small juniper were present and may not intercept the 20 transects, we used an alternative protocol in which major and minor diameters of all juniper within a 2 m wide belt transect centered along each 20 m transect were recorded. Only the portion of these trees within the 2 m wide belt transect were measured. Cover of juniper was estimated by determining the area represented by each tree, summing these areas, and dividing the total by the area represented by belt transects within a study site.

Data were analyzed for agreement with habitat condition assignments and vegetation class assignments which constitute each threat-based model. Functional group scores generated by each remote sensing technique were compared to ground-based estimates using simple linear or non-linear regression across study sites. Habitat condition classification and vegetation classification accuracy based on remotely-sensed data were characterized with respect to percent agreement with habitat condition classification based on ground-based data. Data were then summarized into frequency tables. Statistical tests of predictive power, where valid, used the Chi Square statistic in cases where frequency table cell counts were greater than or equal to five and Fisher's Exact Test in cases where frequency table cell counts were less than five. The total number of field sites sampled was 198 (excluded sites were inaccessible due to topography and ground cover—i.e., cliffs and boulders). Additionally, preliminary analyses suggested that sites with crested wheatgrass (n=25) were strongly skewing results for all remote sensing methods evaluated. Thus, results are shown only for sites without crested wheatgrass (n=173).

Results: Vegetation Class Assignment

Tree Class Assignment

ORC and INR methods both effectively detected tree cover classes (Table 2). ORC correctly classified tree cover on 89.6% of the plots (Fisher's Exact Test $P < 0.0001$) while INR correctly classified tree cover on 87.3% of the plots (Fisher's Exact Test $P < 0.0001$). USGS data did not include tree cover.

Shrub Class Assignment

All methods effectively detected shrub cover classes (Table 3). ORC correctly classified shrub cover on 64.2% of the plots (Chi-square <0.0001). USGS correctly classified shrub cover on 65.3% of the plots (Chi-square <0.0001). INR correctly classified shrub cover on 62.4% of the plots (Chi-square <0.0001).

Annual to Perennial Grass Ratio Class Assignment

None of the methods effectively detected the annual grass to perennial grass ratio ('A/P Ratio' – Table 4). ORC correctly classified the annual grass to perennial grass ratio on 54.9% of the plots (Fisher's Exact Test $P = 0.0708$). USGS correctly classified the annual grass to perennial grass ratio on 49.7% of the plots (Fisher's Exact Test $P = 0.0546$). INR correctly classified the annual grass to perennial grass ratio on 52.0% of the plots (Fisher's Exact Test $P = 0.2687$).

Perennial Bunchgrass Class Assignment

Though perennial bunchgrass cover class is not used alone to assign ecological habitat condition in this exercise, it is a constituent of the annual grass to perennial grass ratio and is helpful in explaining some of the results. USGS effectively detected perennial bunchgrass cover class (Table 5), correctly classifying perennial bunchgrass cover on 68% of the plots (Chi-square <0.0001). INR correctly classified perennial bunchgrass cover on 57% of the plots (Chi-square = 0.0690). ORC correctly classified perennial bunchgrass cover on 53% of the plots (Chi-square = 0.9540).

Table 2. Tree Cover Classes and Count of Agreement/ Disagreement by Two Remote Sensing Methods (No values for USGS).

	Count of *Tree Cover Class* Agreement by Method					
	ORC			INR		
Tree Cover Class	**0**	**1**	**2**	**0**	**1**	**2**
0	143	7	1	144	4	3
1	4	4	4	6	4	2
2	0	2	8	0	7	3

Table 3. Shrub Cover Classes and Count of Agreement/ Disagreement by Three Remote Sensing Methods.

	Count of *Shrub Cover Class* Agreement by Method					
	ORC		INR		USGS	
Shrub Cover Class	**1**	**2**	**1**	**2**	**1**	**2**
1	57	45	45	57	67	35
2	17	54	8	63	25	46

Table 4. Annual Grass to Perennial Grass Ratio Classes and Count of Agreement/ Disagreement by Three Remote Sensing Methods.

	Count of *A/P Ratio Class* Agreement by Method											
	ORC				INR				USGS			
A/P Ratio Class	**0**	**1**	**2**	**3**	**0**	**1**	**2**	**3**	**0**	**1**	**2**	**3**
0	3	2	0	1	0	6	0	0	0	1	4	1
1	15	88	15	13	12	85	20	14	0	72	39	20
2	0	12	1	4	3	13	0	1	0	4	9	4
3	1	11	4	3	2	11	1	5	0	2	12	5

Table 5. Perennial Bunchgrass Cover Classes and Count of Agreement/ Disagreement by Three Remote Sensing Methods.

	Count of *P. Grass Cover Class* Agreement by Method					
	ORC		INR		USGS	
P. Grass Cover Class	**0**	**1**	**0**	**1**	**0**	**1**
0	25	41	39	27	29	37
1	41	66	48	59	18	89

Ecological Habitat Condition Assignment

Ecological habitat condition assignment is dependent on the identified threat model. In practice thus far (i.e., Sage-grouse CCAA site planning, BLM pilot permit renewal project), threat model has been determined by on-the-ground staff using a number of sources, including but not limited to NRCS ecological site descriptions, aerial imagery, visual observation, plot or transect data, and land management history. Because we did not a-priori identify the threat model, to equitably compare methods we assigned threat model based on presence or absence of juniper observed in the field data plots, and excluded from analysis any plot where remotely-sensed data did not have "agreement" of presence or absence of juniper. We concluded based on past experience that the annual grass threat model and dual threat (conifers + annual grasses) model dominate our study area and that the conifers-only threat model represented a relatively small footprint (Table 6). We again excluded plots with crested wheatgrass from the analysis.

Annual Grass Threat Model

For all three methods, the most effectively detected habitat condition was 'A', which was the second most commonly occurring habitat condition (Table 7). ORC, USGS, and INR correctly assigned habitat condition A on 67%, 63%, and 70% of its occurrences, respectively. Detection of other habitat conditions was inconsistent. ORC had annual grass model agreement on 143 plots and habitat condition agreement on 42% of those plots. The most commonly mistaken ORC habitat conditions were A:B (n=9) and B:A (n=21) errors, both of which are defined by shrub cover differences. USGS had annual grass model agreement on 143 plots and habitat condition agreement on 38% of those plots. The most commonly mistaken habitat conditions by USGS were B:D (n=10), which is defined by shrub cover and annual grass to perennial grass ratio, and B:A (n=9), which is defined by shrub cover. INR had annual grass model agreement on 144 plots and habitat condition agreement on 42%. The most commonly mistaken habitat conditions by INR was B:A (n=17) and B:B-D (n=10), which is defined by the annual grass to perennial grass ratio.

Dual Threat Model

Our approach for defining ecological model (presence or absence of juniper) constrained the possible habitat conditions to C, D, and E. Our sample size was further constrained by only including plots where there was agreement between each remote sensing method and observed ecological model. We continued to exclude plots with crested wheatgrass. USGS data did not include tree and were thus excluded from this comparison. ORC had dual threat model agreement on 18 plots and habitat condition agreement on 72% of them (Table 8). INR had dual threat model agreement on 16 plots and habitat condition agreement on 69%.

Table 6. Observed Habitat Condition on 173 Field Plots.

Model	Ecological State						Total
	A	*A-C Transitional*	*B*	*B-D Transitional*	*C*	*D*	
Annual Grass Threat	50	10	63	5	8	15	151
	A	*B*	*C*	*D*	*E*		
Dual Threat	0	0	17	2	3		22

Table 7. Annual Threat Model Frequency Tables Comparing Efficacy of Three Remote Sensing Datasets at Assigning Habitat Conditions.

Habitat Condition	A	A-C	B	B-D	C	D	Total	# Disagree	% Wrong	% Right
	ORC Habitat Condition Estimate						Assignment			
A	31	0	9	5	0	1	46	15	33	67
A-C Transitional	5	1	1	0	0	1	8	7	88	13
B	21	1	24	7	0	9	62	38	61	39
B-D Transitional	1	0	1	1	0	2	5	4	80	20
C	5	0	1	0	2	0	8	6	75	25
D	4	0	5	4	0	1	14	13	93	7
Total							143	83	58	42
	USGS Habitat Condition Estimate						Assignment			
A	29	4	0	5	0	8	46	17	37	63
A-C Transitional	3	1	0	1	2	1	8	7	88	13
B	9	4	19	16	4	10	62	43	69	31
B-D Transitional	0	2	0	0	0	3	5	5	100	0
C	2	2	1	3	0	0	8	8	100	0
D	1	0	1	5	1	6	14	8	57	43
Total							143	88	62	38
	INR Habitat Condition Estimate						Assignment			
A	32	9	3	2	0	0	46	14	30	70
A-C Transitional	5	1	2	1	0	0	9	8	89	11
B	17	9	27	10	0	0	63	36	57	43
B-D Transitional	5	0	0	0	0	0	5	5	100	0
C	3	3	1	1	0	0	8	8	100	0
D	5	3	4	1	0	0	13	13	100	0
Total							144	84	58	42

Table 8. Dual Threat Model Frequency Tables Comparing Efficacy of Three Remote Sensing Datasets at Assigning Habitat Conditions.

Habitat Condition	A	B	C	D	E	Total	# Disagree	% Wrong	% Right
	ORC Habitat Condition Estimate					Assignment			
A	0	0	0	0	0	0	0	0	0
B	0	0	0	0	0	0	0	0	0
C	0	0	13	0	1	14	1	7	93
D	0	0	2	0	0	2	2	100	0
E	0	0	2	0	0	2	2	100	0
Total						18	5	28	72
	INR Habitat Condition Estimate					Assignment			
A	0	0	0	0	0	0	0	0	0
B	0	0	0	0	0	0	0	0	0
C	0	0	11	0	0	11	0	0	100
D	0	0	2	0	0	2	2	100	0
E	0	0	3	0	0	3	3	100	0
Total						16	5	31	69

Implications

Our work suggests that both opportunities as well as challenges exist associated with the use of remote sensing data to classify habitat conditions in large landscapes. Where challenges exist, it is our hope that this report helps to focus future remote sensing research by pointing out deficiencies of the various methods in predicting specific vegetation cover class attributes.

In general, there was relatively high agreement across the entire study area among methods and very little area with maximum dissimilarity (Figures 6-8). Both ORC and INR reliably predicted tree cover class, all techniques predicted shrub class > 62% of the time, and USGS exceled at predicting perennial grass class (68%). All techniques also reliably predicted habitat condition A in both models and ORC and INR predicted habitat condition C in the dual threat model with near perfect or perfect performance, respectively. Furthermore, closer examination of habitat condition prediction errors suggests that the most egregious errors, such as mistaking a State A for a State D, are rare; errors of this type were 2%, 17%, and 0% for ORC, USGS, and INR respectively (Table 7). From a management standpoint, the ability to accurately map occurrence of habitat condition A is important given that these habitat conditions are relatively “intact” and conservation of such communities typically experiences a higher rate of success than restoration of non-intact plant communities (e.g., annual grass threat habitat conditions C or D). Along the same lines, recognition of habitat condition C in the dual threat model could help managers to recognize plant communities in danger of transitioning to conifer dominance prior to crossing conifer thresholds that limit management options.

Our data also indicate challenges in the use of remotely sensed data for determining some vegetation attributes and assignment of habitat condition associated with those attributes. Determination of annual to perennial grass ratio class was uniformly poor and only USGS predicted perennial grass class values at a level appreciably different than by chance alone. In some cases, the data provided by a remote sensing platform was categorically course relative to our needs in defining habitat conditions. For example, all platforms grouped shrubs by species whereas our classification system relies on cover values for sagebrush specifically. This creates conflict when shrub species other than sagebrush (e.g., rabbit-brush) are common at a site (e.g., see the “Disagreement: Shrub Type” image in Appendix 2). However, in discussions with remote sensing experts, we anticipate that this challenge is relatively minor. Additionally, data for annual grass abundance were collected, in all cases, in different years than our field data collection which could have led to inaccuracies in class assignment for annual to perennial grass ratio. This discrepancy may explain why habitat condition A—characterized by >10% sagebrush cover and dominant large perennial bunchgrasses (which fluctuate less in response to interannual variation)—was reliably detected whereas some other habitat conditions, such as habitat condition C in the annual grass threat model, were not. The importance of annual grasses to habitat condition assignment within the annual grass threat and dual threat model suggests that the level of effort needed to procure and process remotely sensed data in a timely fashion will greatly affect the utility of these techniques in future habitat condition classification efforts.

Another ongoing challenge will be the imperfect nature of any habitat condition classification system. For example, in the “Disagreement: Shrub Type” image in Appendix 2, field data classified this site as habitat condition B because sagebrush cover was < 10% (it was actually 9%) and because the annual to perennial grass ratio was <1. However, most field observers would have classified this site as a habitat condition C due to the abundance of shrub cover and lack of understory perennial grasses. This discrepancy has some very real management implications. For example, habitat condition B is considered to be fairly resistant to annual grasses due to the abundance of perennial grasses. In this case, perennial grass cover was very low (< 1%) but annual grasses were absent, which results in a very low annual to perennial grass ratio and subsequent classification as habitat condition B. Because of the near absence of perennial grasses, this plant community probably has very low resistance to future annual grass invasion (a ruderal annual forb species, *Alyssum desertorum*, was abundant). One solution would be to put in place a minimum value of, say, 5% cover of perennial bunchgrasses for inclusion in habitat condition B. This is problematic, though, due to variation in both a) site potential for perennial grass production as well as b) the amount of perennial grasses needed to impart resistance to annual grasses under variable environmental conditions. To us, this conundrum suggests that there is no ecologically perfect classification system. The upshot of this reality is that when remotely sensed data is used for purposes of habitat condition classification, it would be wise to put in place quality control measures to ensure that classified values mesh with on-the-ground reality. Our results suggest that efforts invested in field verification for quality control could be prioritized towards fractions of the landscape where remotely sensed data is less effective at detecting habitat condition.

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Appendix 1: Table of Accepted Habitat Condition Metrics

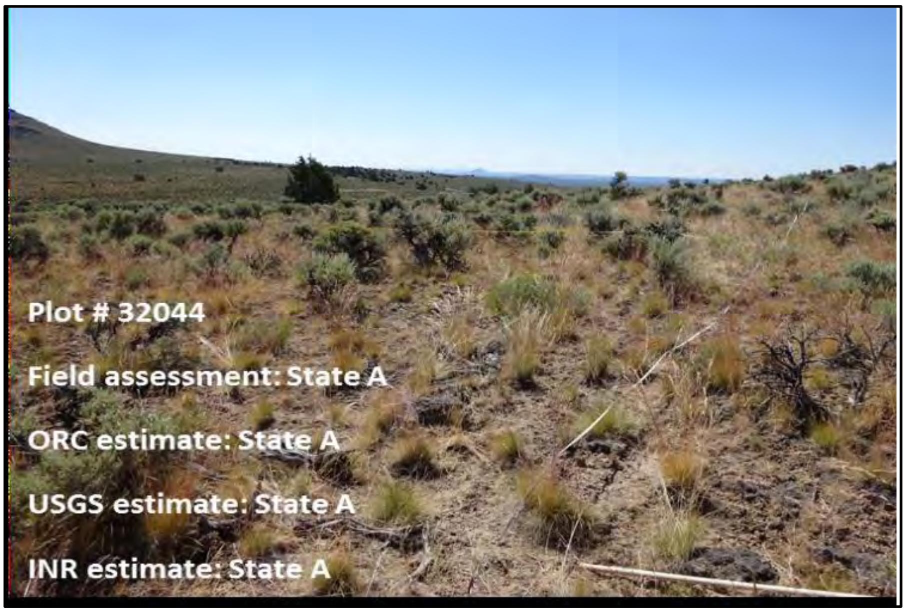
Threat Model	Dominant Sagebrush Spp.	Habitat Condition	Sagebrush Canopy Cover (%)	A. Grass : P. Grass cover	P. Grass cover or density (% or individuals/ m^2)	Juniper Canopy Cover (% - over 6ft)
Invasive Annual Grass	Wyoming Big Sagebrush	A	≥10	≤1:1	N/A	N/A
		A-C Transitional	≥10	3:1—1:1	N/A	N/A
		B	<10	≤1:1	N/A	N/A
		B-D Transitional	<10	3:1—1:1	N/A	N/A
		C	≥10	>3:1	N/A	N/A
		D	~0	>3:1	N/A	N/A
Invasive A. Grass/ Conifer Expansion	Wyoming / Mountain Big Sagebrush	A	≥10	≤1:1	N/A	0
		B	<10	≤1:1	N/A	0
		C	<10	≤1:1	N/A	≤10
		D	~0	>1:1	N/A	>10
		E	~0	>1:1	N/A	>10
Conifer Expansion	Mountain Big Sagebrush	A	≥10	N/A	≥5	0
		A-C Transitional	≥10	N/A	≥5	≤10
		B	<10	N/A	≥5	0
		B-D Transitional	<10	N/A	≥5	≤10
		C	≥10	N/A	≥5	>10
		D	~0	N/A	≥5	>10
		E	~0	N/A	<5	>10

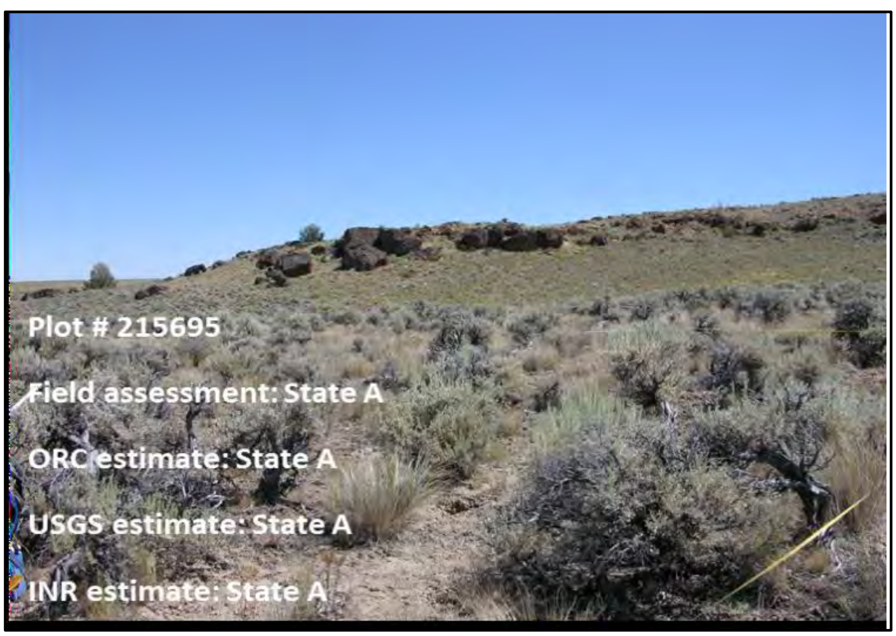
Appendix 2: Photographic Samples for Further Reference

Visual inspection of plot photos, particularly of areas of disagreement provides some additional context and anecdotal insight into the value and limitations of the threat-based framework and use of remotely-sensed data to recognize habitat conditions. The authors believe that these may help inform additional areas for advancement of remote sensing methods as well as forewarn end-users of potential problems with using remote data. Several examples follow.

Strong Agreement: Habitat Condition A

As noted in the results, all remote sensing methods detected habitat condition A reliably (with 63-70% accuracy). If we assume that these sites are relatively stable, it is possible that inter-annual variation of vegetation characteristics and disparate years of remote sensing data collection would have less effect on the recognition of these.



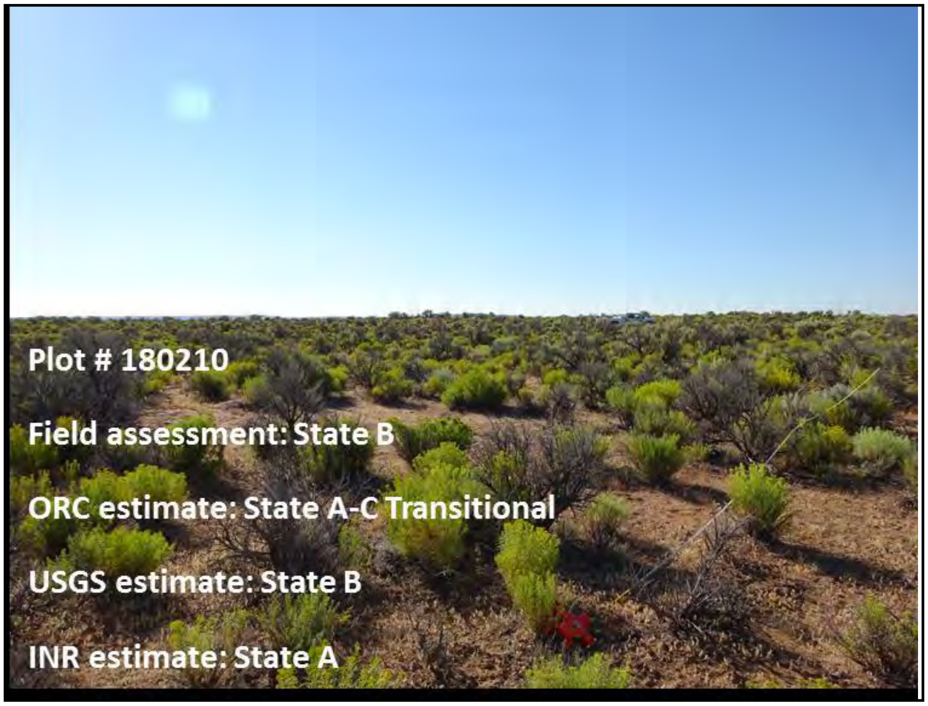


Strong Agreement: Juniper Cover Habitat conditions

As noted in the results, INR and ORC methods were highly effective at assigned juniper cover classes. Though the sample size of juniper plots was limited, the authors of this report are confident that these remote sensing methods will consistently detect and recognize habitat conditions with juniper encroachment.

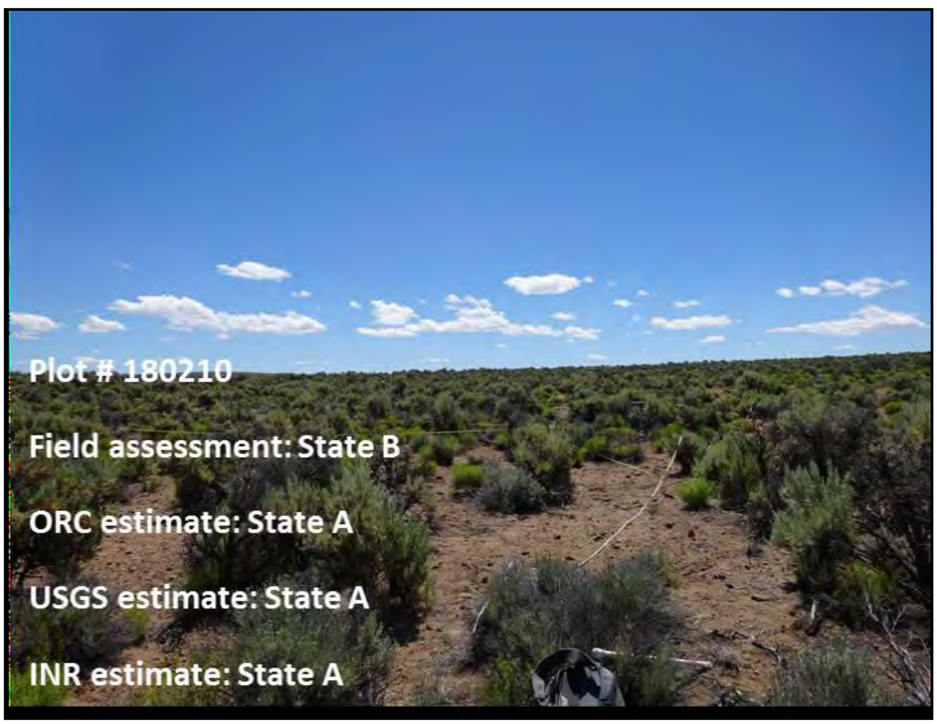
Disagreement: Shrub Type

We observed several instances of disagreement that is attributable to methodological challenges. Though the ecological framework of the threat-based models is centered on sagebrush (the dominant shrub species in this ecosystem), remotely detected species differences among similarly statured shrubs in the same landscape (e.g., rabbitbrush) is difficult and may lead to differences between remotely-sensed estimations and field observations of habitat condition.



Sampling Challenges

The authors of this report recognize that sample size is always a challenge when characterizing vegetation communities. Furthermore, visual observation and expert opinion sometimes are not easily reconciled with raw data. The following example demonstrates a case in which expert opinion would have classified this plot as habitat condition C, yet neither field data nor any remotely sensed data set agree with that conclusion. We draw attention to this example to illustrate the point that there are inherent flaws in any method attempting to characterize plant communities that vary significantly through space and time.



Rangeland Practices in the Western Sagebrush Steppe: Published Scientific Literature

Manager's Guide 2

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Introduction

The western sagebrush steppe is a biome that varies tremendously over time and space (West 1999; Svejcar et al. 2017). There are many plant communities within the biome and these communities are influenced by a variety of abiotic and biotic factors. The variable climate, a variety of invasive species, changes in atmospheric CO_2, changes in disturbance history, and other human activities all influence vegetation dynamics.

In Guide 1, we outlined the value of simple mental models for providing a lens through which a wide variety of stakeholders can view current and future vegetation conditions. Lack of a common vision can derail conservation efforts, especially at the scale required to improve greater sage-grouse (GRSG) habitat. The simplicity of basic mental models allows development of a strategy which can be articulated in a half page of writing. Toward the end of Guide 1 we suggested that structured decision-making (SDM) as presented by Tulloch et al. (2015) could be combined with mental models to develop a framework for improving GRSG habitat. The steps of SDM are:

1) Define clear, quantifiable objectives and constraints relative to the problem;

2) Identify a set of alternative management actions;

3) Evaluate the potential effects of management actions as related to initial objectives;

4) Address uncertainty (which may result from either temporal and spatial variability, or lack of knowledge);

5) Assess trade-offs and make a decision.

The mental model presented as Figure 6 in Guide 1 helps develop a generalized strategy with measurable objectives as presented in Table 5. The next steps are to develop the tactics necessary to achieve the objectives. Tactics revolve around rangeland management practices that can be used to improve GRSG habitat by reducing two primary threats: invasive annual grass and conifer expansion.

There has been an increasing emphasis on quantifying the impacts of conservation practices. A leading effort was the Conservation Effects Assessment Project (CEAP). The impetus for this effort was a substantial increase in conservation funding in the 2002 Farm Bill (Duriancik et al. 2008). The CEAP effort was separated into several components, one of which was the Rangeland CEAP Synthesis (USDA-NRCS 2011). The synthesis was a comprehensive literature review focusing on the following NRCS conservation practices: prescribed grazing, prescribed burning, brush management, range planting, riparian herbaceous cover, upland wildlife habitat management, and herbaceous weed control. The authors concluded that “Although these analyses collectively indicate that NRCS investments in conservation programs are sound, it was not possible to determine the magnitude or trend of conservation benefits originating from these investments because of the paucity of information documenting conservation benefits”. Because there was limited information documenting the effects of the various conservation practices as they were applied, the Rangeland CEAP focused on a broad-scale review of the scientific literature. In developing the tactics for our GRSG habitat conservation strategy we will also focus on the scientific literature. However, our literature base will be much more geographically specific. For this guide, we will focus on grazing, fire, herbicide, mechanical control, seeding, and other miscellaneous practices in the western sagebrush steppe.

Complex vs. Simple Problems

The very act of setting goals and objectives implies a problem; that “problem” being defined as the extent and nature of the distance between current and future desired condition of the resource. It is important to realize the type of problem we are dealing with and what the implications of the nature of the problem are to management, planning, and actions. Simple problems are those problems that have solutions which are invariant in space and time. For these problems, generalized solutions have broad management utility. An example of a simple problem would be a fuel reduction treatment in which woody sagebrush fuels are reduced using a brush beating technique (assuming invasive species are not an issue). The results of brush beating are likely to be both successful and predictably so in space and time to the extent that treating 10 acres is synonymous with reducing the size of the problem by 10 acres. The SageCon Habitat Quantification Technical Team (2015) presents another tool for solving rangeland management problems in a simpler way via conservation mitigation tables by threat model (see Appendix 6).

Complex problems are those problems for which the nature of the problem, and by extension, appropriate management actions, will vary depending on where you are and when you are there (i.e., space and time—Boyd and Svejcar 2009); most sage-grouse habitat management problems in the western half of the species range fall into this category. With complex problems, generalized solutions do not have broad management utility. Even a treatment as seemingly straight-forward as cutting juniper can vary greatly in response depending on phase of encroachment, type of site, understory characteristics, and weather in the initial years following cutting.

The Sage-SHARE database allows us to group articles into categories based on variables such as precipitation, elevation, dominant plant species, or vegetation threat, and determine if conservation practices have similar effects across categories (simple problems) or must be applied within a specific area to be effective (complex problems).

Sage-SHARE Database

Any literature review covering broad topics can be a challenge. Where does one draw the boundaries on the articles to be included? How broadly does one evaluate the response variables, since they tend to vary among studies? Even simple vegetation measurements can cover a wide variety of variables. Plant response variables include cover, density, frequency, biomass, tiller counts, leaf and area index, all of which can be measured using a variety of techniques. Because of the landscape complexity of the western sagebrush steppe, we may want to separate articles based on site conditions or location, such as elevation, precipitation zone, aspect, and so on. To evaluate the literature on rangeland management practices in the western sagebrush steppe, we developed a relational Microsoft Access® database. The

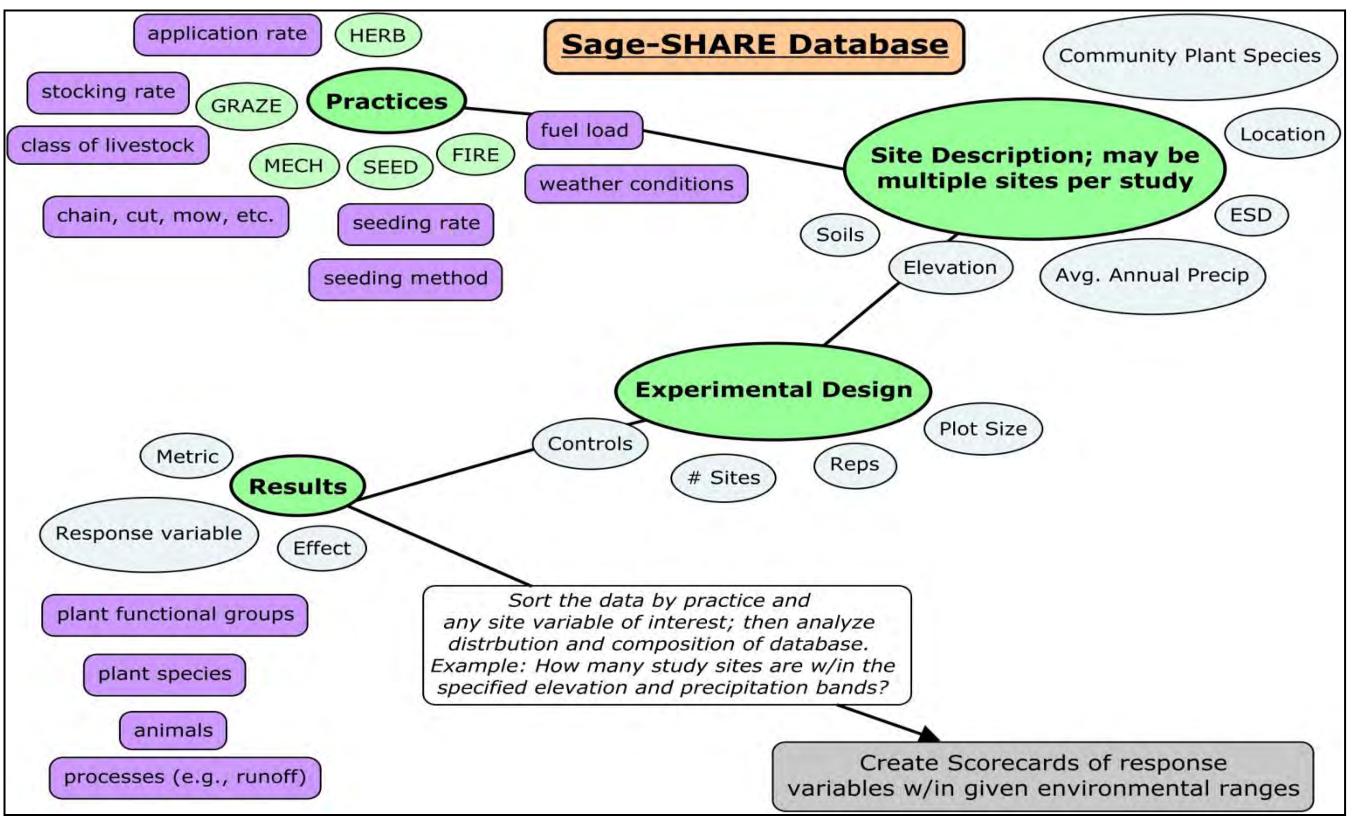

Figure 1. Conceptual map of Sage-SHARE database.

database was named Sage-Steppe Habitat Restoration or Sage-SHARE, and was populated with published scientific literature. The conceptual basis for the Sage-SHARE database appears in Figure 1. One advantage of a relational database is that it can be queried based on one or multiple factors. A query could be done for rangeland seeding on north aspects between 4000 and 5000 ft in elevation. Sage-SHARE is broken into two parts or functions; one for data entry and another for data queries (apart from the ability to design one's own queries).

Each study entered into the database has fields to populate within site description, experimental design, and results. Within the site description, fields include five key rangeland management treatments: fire, grazing, seeding, mechanical treatment, and herbicide application. No data interpretation was made while entering sources. Simple, built-in queries can be run from the main page, such as filtering studies by targeted plant species, elevation, or desired result. However, more complex queries were necessary and designed to more efficiently analyze the data. The data entered into Sage-SHARE was first catalogued on "EndNote Web", which allows for a versatile and license-free mechanism from which to manage the literature library. Microsoft Access 2007® or newer is required to support opening or editing the database. A screen shot of the entry portal for the database is shown in Figure 2. Data entry screens are shown in Figure 3 (site description data), and Figure 4 (results input page). The database comes with a complete user manual (Connell and Holman 2016), and the structure is such that it could be used for a variety of purposes. For example, the database could be populated with landowner or manager experiences rather than scientific literature.

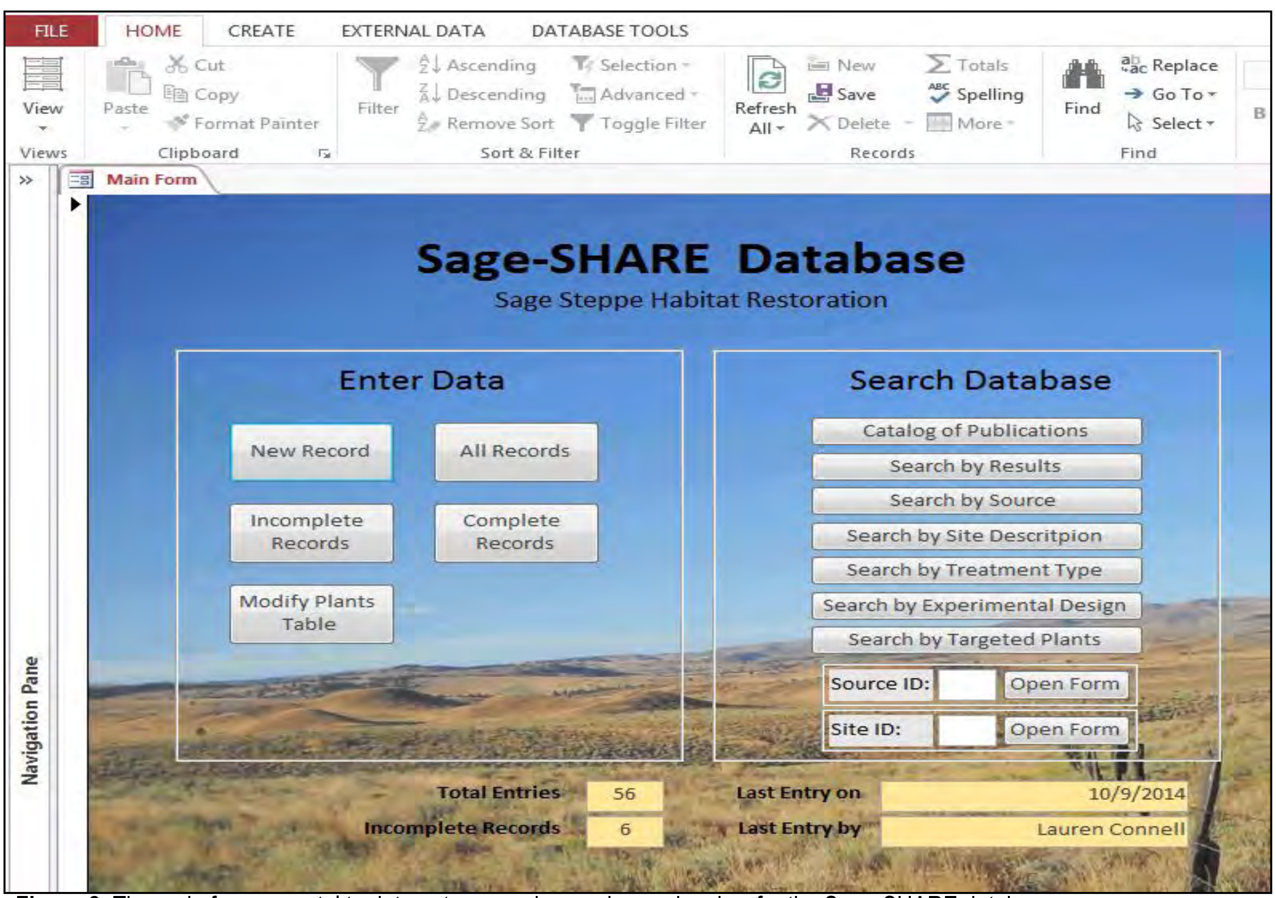

Figure 2. The main form or portal to data entry, querying, and record review for the Sage-SHARE database.

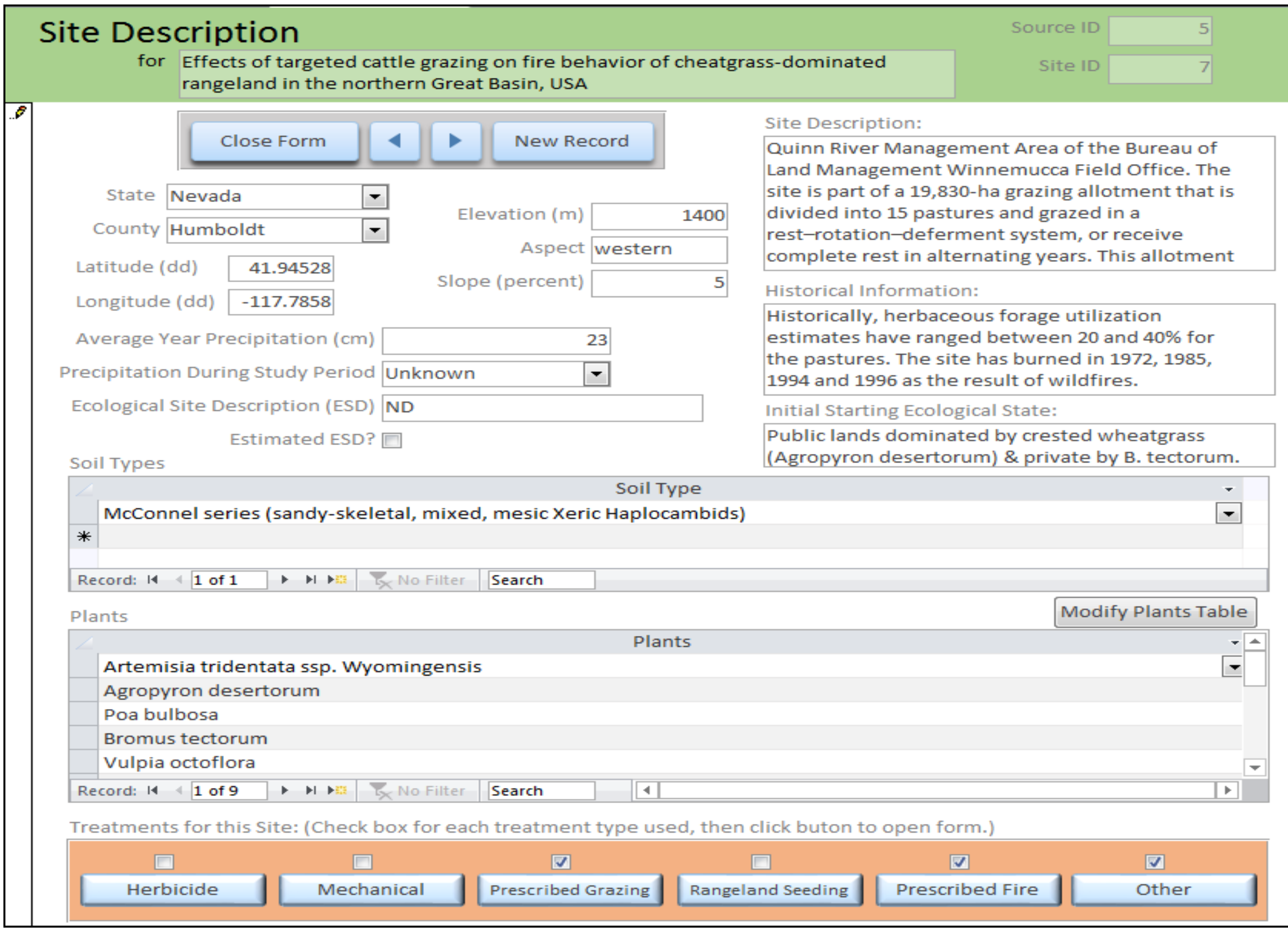

Figure 3. Site Description form.

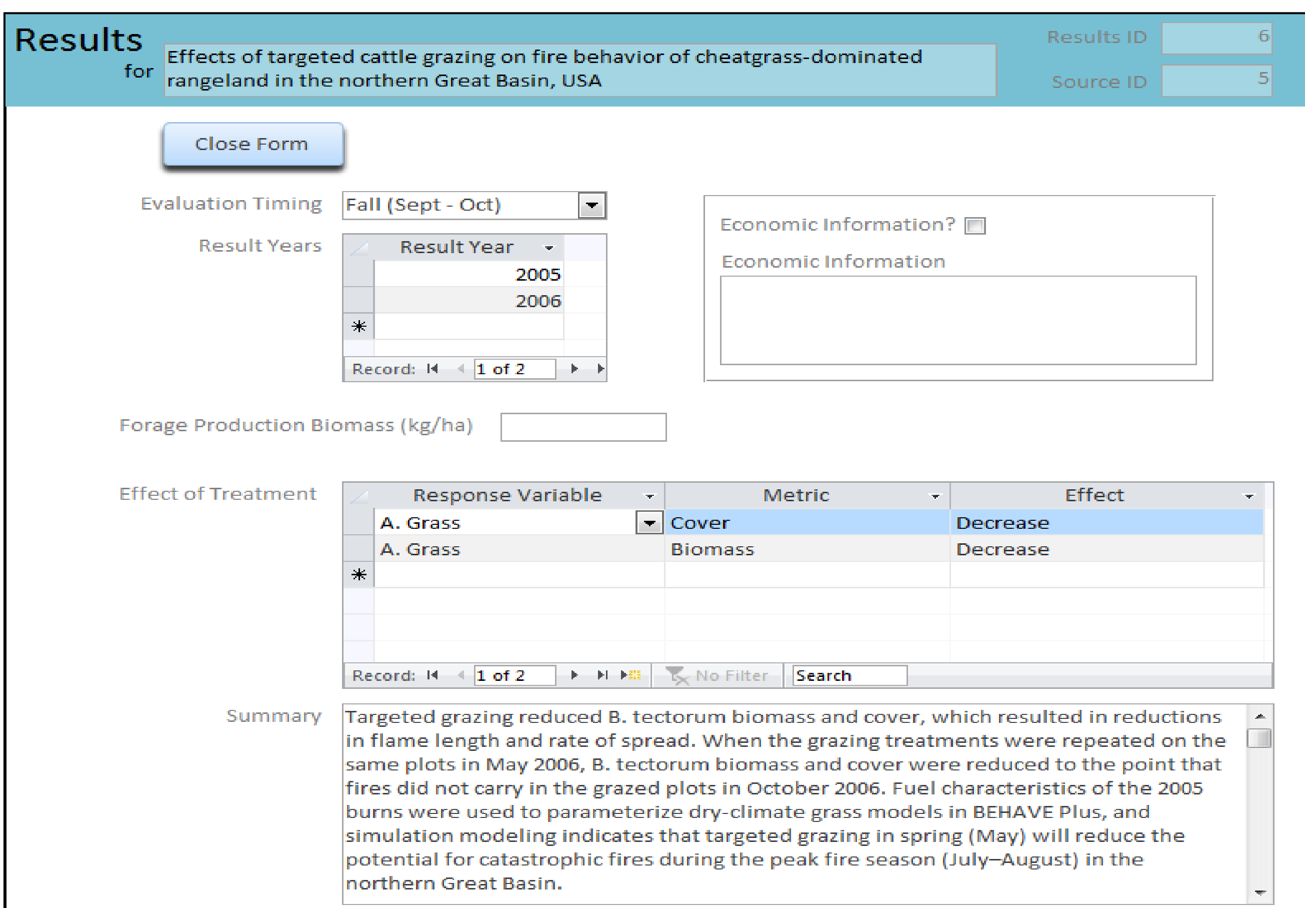

Figure 4. Results form.

Database Details and Challenges

We were not aware of any similar efforts when we started on the development of Sage-SHARE. There have been efforts to use meta-analysis in applied ecology (e.g., Stewart 2010), but meta-analysis is the application of statistical methods to the synthesis of multiple studies, usually on a specific topic. The database effort would involve more studies and more topics than typically included in meta-analysis, and the statistical approach usually focuses on answering a few specific questions rather than sorting articles into various categories. One of our goals was to develop a system that would allow for sorting needs that may not be apparent at this time such as an interest in finding all seeding studies conducted on south slopes in the 10-12” precipitation zone.

Given the diversity of research approaches, journal formats, and author preferences, we knew it was going to be a challenge to develop a database which was consistent across fields of entry and the papers span over a half century of time. Over that period, presenting study locations has progressed from county and state to general latitude and longitude to precise GPS –derived coordinates such as decimal degrees. If a study was conducted without access to long-term weather data, we now have the ability to predict climate from modeling efforts such as PRISM (www.prism.oregonstate.edu). Studies produced from the early part of the period of record would not have had access to that sort of modeling. We worked to extract information as consistently as possible and to fill in as many input variables as possible. We found that interpretation of study design and sites has an impact on query results. Studies with multiple treatments cannot be wholly captured in a results section where determinations on treatments had a positive, negative or no effect. A summary interpreted by the person inputting each journal article is available in a “comments” box. However, any data or results captured in a comments box cannot be queried. When queried for positive, negative or no effect—one study will often be in all queries. When research involved invasive species, there could be an array of results including a treatment effect on invasive species and also on desired plant species. This necessitates those using the database to review specific entries.

Once research was entered, we learned quickly there is large variation in sites across the area of interest including elevation, climate, disturbance regime, and dominant plant community. This is both the beauty and the difficulty in a searchable database. We can begin to see if patterns exist across the variability but it is also difficult but necessary to categorize the data into groups multiple times from varying angles.

We noticed inconsistency in reporting research results over time and journals. Reporting research methodology in peer reviewed ecological journals was highly variable especially for describing sites and response variables. Different protocols were used to measure and report data, especially for studies taken from decades ago. This led to the many “unknown” and “other” data that was grouped together for the sake of simplicity. More detailed site description requirements by scientific journals would increase the efficiency of literature analysis.

New research publications can be added to the Sage-SHARE database. The procedures for adding articles are outlined in the Sage-SHARE Database User’s Manual (Connell and Holman 2016). For the purpose of this exercise we used only refereed scientific publications and focused on the primary management practices used in the western sagebrush steppe. There are sure to be publications that were inadvertently left out, but they can be added as necessary. An extensive literature search was conducted to acquire the articles presented in the database. The list below describes the process used to identify articles.

Sources used to Identify Publications:

1) OSU Library Keyword Searches (Table 1)

2) Range Science Information System (RSIS): http://arc.lib.montana.edu/range-science/titles.php

3) BLM Library>Greater Sage-grouse>Subject Guide>Literature: https://www.blm.gov/wo/st/en/info/blm-library/research/subject-guides/greater_sage-grouse_subj_guide/gsg_lit.html

4) http://www.sagestep.org/index.html

5) USDA General Technical Report RMRS-GTR-308: A Review of Fire Effects on Vegetation and Soils in the Great Basin Region: Response and Ecological Site Characteristics (references were utilized as source of publications for data entry)

\6) Sotoyome Resource Conservation District: Grazing Handbook: http://www.carangeland.org/images/GrazingHandbook.pdf(references were utilized as source of publications for data entry)

7) Conservation Biology Institute: Recommended Reading for Grasslands Symposium: http://d2k78bk4kdhbpr.cloudfront.net/media/content/files/RecommendedReading.pdf(references were utilized as source of publications for data entry)

8) Sage-Grouse Habitat in Idaho: A Practical Guide for Land Owners and Managers: http://www.sagegrouseinitiative.com/wp-content/uploads/2013/08/SGI_FieldGuides-Idaho.pdf(references were utilized as source of publications for data entry)

9) Nevada Wildlife Federation: Enhancing Sage Grouse Habitat - A Nevada Landowner's Guide A Northwest Nevada Sage Grouse Working Group publication: http://www.nvwf.org/pdfs/grouseguide.pdf(references were utilized as source of publications for data entry)

10) USDA NRCS Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps ISBN 978-0-9849499-0-08 (references were utilized as source of publications for data entry).

Now that we have described the database, it is important to recall why the database was initially developed – to help design the tactics portion of the GRSG Habitat Conservation Strategy. The two primary vegetation threats to GRSG habitat are invasive annual grasses and conifers. The tactics will generally be to control annual grasses and conifers and to promote healthy vegetation which provides GRSG habitat. Our environment is both temporally and spatially variable (as described in Guide 1) and conservation practices may be classified as either simple or complex, a topic we discuss in the next section.

Response Variables

The input page of the database is consistent among articles (Figure 4), but there are a wide variety of response variables among the various articles. Most of the responses measured relate to vegetation characteristics, and how these characteristics were affected by a specific practice. A summary of response variables summed across all articles in the database is presented in Table 2. There are over 1800 response variable entries, but some fall into multiple categories. The primary functional groups of interest for determining if a practice might affect habitat condition are: perennial grass, annual grass, shrub, and tree. Forbs are also an important consideration for GRSG habitat, but they are not a primary driver of habitat condition as defined in Figure 2 of Guide 1. Forbs were included in the scorecard ratings presented later.

Table 1. List of Keywords.

Artemisia tridentata	Pinyon-juniper	Sage-grouse response
Medusahead	Sagebrush	Weed control
Juniperus occidentalis	Rangeland seeding	Bromus tectorum
Vaseyana	Cheatgrass	Prescribed grazing
Prescribed fire	Sage-steppe	Seeding
Herbicide	Juniperus occidentalis	Rangeland management
Western juniper	Mountain big sagebrush	Sage-grouse habitat
Taeniatherum caputmedusae		

General Database Summary

Earlier in this guide we introduced the concept of simple vs. complex problems, and suggested that GRSG habitat issues were often complex, where one solution does not work equally across the landscape. One value of the database approach of Sage-SHARE is that articles can be broken into landscape categories. Our first analysis did not separate out practices; rather we simply put all the articles into three broad elevation ranges and five precipitation zones (Table 3). The elevation ranges roughly corresponded to the threat-based models where "low" was less than 4000 ft, the "middle" or mixed range was 4000-5500 ft, and "high" was above 5500 ft. The precipitation zones (PZ) correspond to those commonly used by NRCS to define ESDs (for example, Loamy 10-12 PZ).

The elevation of individual sites was generally provided within an article. When no elevation was provided, latitude/longitude for the site was used to estimate elevation with PRISM. The same general approach was taken for precipitation. When annual precipitation was not provided in an article, PRISM was used to estimate that parameter. Roughly 30 site elevations and precipitation zones were estimated. The acronym PRISM stands for Parameter-elevation Relationships on Independent Slopes Model. Details can be found at www.prism.oregonstate.edu or in Daly et al. (2008). The elevation analysis shows that the lower elevation (<4000 ft) is under-represented in number of articles relative to the mid and upper elevation categories (Table 3). Articles are well distributed among the five PZs with a slight under representation in the <10 PZ (Table 4).

Table 2. Database response variables and associated number of entries made. Key terms are in yellow.

Response Variable	#
A. Grass	256
P. Grass	384
Forb	241
Shrub	306
Tree	37
All plant	115
Ammonium	3
Animal/Insect/ Bird	84
Bare Ground	30
Fire	5
Forage	7
Grass	38
Herbaceous	36
Invasive	65
Litter	12
Model Prediction	1
Nutrient/Energy Exchange	51
Runoff	4
Seed/Seedling/ Collection	35
Soil/Biological	89
Standing Crop	6
Weed	22
Total	**1827**

Distribution of Practices

The next step was to add practices to the sorting criteria. For example, we wanted to determine if some practices were over- or under-represented in particular elevation and precipitation categories. For this analysis we also sorted by subspecies of big sagebrush (specifically Wyoming and/or Mountain big sagebrush), and by threat. In the case of threats, we had to use presence or absence of a threat species – annual grasses and/or conifers. There may be cases where annuals are mentioned in a study but are not really a threat to dominate. And conversely, there may be cases where conifers are a threat, but are not mentioned as a portion of study site flora. There was a high level of uncertainty in classifying articles (or sites) based on either sub-species of big sagebrush or threat model. When we separated based on subspecies, the number of unknown sites ranged from 28% for mechanical treatments to almost 59% for seeding. In the case of threat models, the level of unknowns ranged from 37% for mechanical treatments to 72% for grazing. In contrast, when we separated sites based on precipitation, unknowns ranged from less than 7% for mechanical treatments to 18% for fire.

We chose not to present results for the separations based on subspecies or threat model because the large proportion of unknowns could skew results. The results for elevation and precipitation (Table 4) are in line with the general summary of practices from Table 4.

There tends to be an under-representation of all practices at <4000 ft, with numbers for mechanical and grazing studies being particularly low. Fire studies also appear in much higher numbers at the upper two elevations, where they constitute higher numbers of study sites compared to other practices. No matter how the studies are parsed, there are many more studies on fire relative to other practices. The general order of practices by total number of study sites is fire> grazing > seeding > mechanical> herbicide. If we look only at number of articles the order is similar, except herbicide and mechanical reverse. The mechanical treatment studies include more total study sites than the herbicide treatment studies.

Limitations of Database Analysis

The last section provided a macro-view of the distribution of studies and study sites, sorted by elevation or PZ and individual practice. We attempted several additional sorting criteria other than elevation and PZ, but found too many unknowns to make the results useful. The next step in the process was to see if we could use the database to summarize the effects of practices on the major plant functional groups of interest.

As expected, there were many mixed results and any questions arose. Before we present those results it might be useful to discuss why we should expect a good deal of variation in our assessment of practice effects. One major consistency issue for our practices analysis is the variety of individual treatments within each practice. For example: 1) fire can be wildfire or prescribed fire and in either case, can occur under a wide variety of conditions, 2) grazing can be a multitude of treatments with variation in stocking rates, season of use, duration, and period of rest following grazing, 3) seeding can include a variety of seedbed preparation methods, seeded species, seeding methods and rates, and post-seeding treatments, 4) herbicide treatments are often designed to include a variety of rates (some of which are intended to be too low to be effective) and there are multiple herbicides with different target species and different modes of action, and 5) mechanical treatments come in many forms from chainsaw cutting of conifers to brush beating of shrubs.

Most mechanical treatments are targeted to control individual species or functional groups. But, impacts

Table 3. General summary of Sage-SHARE articles sorted by (A) elevation (ft) and (B) precipitation (in) categories. Values represent number of sites within each category. Studies may have multiple sites as shown in Table 4.

(A)

Elevation	Sites
Unknown	110
< 4000	57
4000-5500	218
> 5500	194
Total	**579**

(B)

Avg. Annual Precipitation	Sites
Unknown	115
< 10	137
10-12	110
12-16	151
> 16	66
Total	**579**

Table 4. Reference Count by (A) Elevation (ft) and (B) Precipitation Zone (in).

	Method *Total Entries*	Fire *243*	Grazing *140*	Seeding *133*	Mech *116*	Herb *108*
	Unknown	41	23	15	13	13
	< 4,000	18	10	20	8	16
	4,000-5,500	86	58	54	43	43
(A)	**> 5,5000**	98	49	44	52	36
	Unknown	44	24	19	8	17
	< 10	51	46	28	20	28
	10-12	51	25	28	30	23
	12-16	82	22	37	49	21
(B)	**> 16**	15	23	21	9	19

can vary over time. There are examples where reductions are temporary and the eventual outcome of a treatment is higher rather than lower levels for a target species.

A second major issue is that starting vegetation condition for individual studies within a practice can be very different. For example, fire can occur in good condition plant communities where native bunchgrass density is at peak levels, or it can occur in sagebrush with a depleted understory. In the first case, fire may cause a decrease in native bunchgrasses, but the community may still be resistant to annual invasion in spite of the decrease. In the second example, removal of sagebrush canopy may open the community to native bunchgrass recruitment, but still favor annual grasses. Thus, in the first example, a decrease in native bunchgrass density could still result in a stable native community. In the second example we could see an increase in native bunchgrasses but still be under threat of annual dominance. Starting point matters a great deal in the interpretation of practices.

> *The distinction in starting point is not necessarily easy to identify, even with the sorting tools available within the Sage-SHARE database.*

Another complicating factor in the interpretation of practices is the duration of individual studies. An example of this would be conifer cutting projects. The initial response is clearly a reduction in conifer cover and density; however, if there are many small trees on the site, removal of large trees may "release" the small trees from competition, and several decades later (or sooner) there may be an increase in density and cover of conifers. In this example, response may depend on post-treatment sampling interval, or time since treatment.

Other issues with consistency may stem from how data was reported (by authors) and how data was entered into the database. There does not appear to be a streamlined protocol for rangeland ecology studies as there is in other fields such as medicine and psychology, and this leads to variable information reporting across studies. For example, one study might give more specific site characteristics such as slope or aspect while another does not. Another example might be a study reporting on the reduction of annual grasses by cover while another study measures density or biomass. This kind of inconsistency makes it difficult to compare results quickly. The variation in reporting also means there is a chance different people entering articles into the database could make slightly different interpretations. We tried to minimize variation in interpretation of results and only use the author's conclusions. It is also important to note that multiple metrics for one study may be included in the scorecard tabulation (e.g., annual grass reduction measured by cover, density, and biomass in the same study would count as three marks under the decrease column).

And finally, we could not end this section without addressing the issues of variable site characteristics and yearly weather as factors which complicate practices interpretation. We broke down study sites by elevation and PZ, but that does not account for factors such as slope, aspect, soil type, or weather during the study (as opposed to longer term climate for a site). These factors interact to influence species composition as well. Annuals, in particular, can vary tremendously from year-to-year. We know that south-facing sites are more susceptible to annual dominance after fire than are north-facing sites and there are many other examples of site-specific or year-specific effects on the success or failure of particular practices.

We felt it was important to provide this discussion before presenting the scorecard summaries of the effects of practices on functional groups because of the lack of consistency in many of the responses. Given the variation in the nature of the studies and the issues covered in this section, those inconsistencies or mixed results should not be a surprise. After the results are presented, we will provide suggestions as to how the Sage-SHARE database can be used efficiently to sort through the multitude of responses.

Practices Scorecards

Now that we have shown the response variables associated with the articles and the means by which we separate articles into categories, the next step was to rank practices based on how they affect the major vegetation functional groups.

The Access database was built with the following responses possible for entering response variables into the results: negative, positive, increase, decrease, mixed, and none. Any functional group marked with a negative or decrease went into the "-" column; any marked with a positive or increase went into the "+" column; none were classified with "0"; and mixed stayed as "mixed". Mixed is separate from none because it indicates that there were both positive and negative results as opposed to no results. It is important to remember that the "+" and "-" designations are based on an increase or decrease in values, not whether the increase or decrease was good or bad. For example, we would take a decrease in cheatgrass cover to be positive, but for purposes of entering data and for the scorecards, that would be entered as decrease and go in the "-" column.

Scorecard Summary

In this section we analyze each practice's effect on the functional groups annual grass, forb, perennial grass, shrub, and tree. We again used the sorting functions within the database to separate articles by elevation and precipitation zone. The distribution of practices has already been discussed (Table 3), so we will not repeat those findings but will instead focus on how a practice influences the major functional groups and whether that influence varies by elevation or PZ.

Fire

Whether separated by elevation or PZ, the bulk of the functional groups responses to fire fell in the mixed category (Table 5). In other words, depending on specific circumstances (year, individual species, etc.), most studies showed both positive and negative effects on individual functional groups. This tendency was particularly evident for annual and perennial grasses. Annual grasses were more likely to be negatively impacted by fire in the <10 PZ relative to higher PZ categories (Table 5b). This result was a little counter-intuitive, and may have to do with initial high levels of annual grasses in the <10 PZ. At the mid-elevation (4000-5500 ft) and 10-12 PZ there was a tendency for perennial grasses to be favored by fire. The functional groups with the most impact from fire were forbs and shrubs. Forbs tended to be favored by fire at higher elevation and PZ sites. Forbs are especially favored in the 12-16 PZ (Table 5b). Shrubs tended to be negatively impacted by fire across elevations and PZ's. This is a case where differential species responses can cause mixed results. Most sagebrush species in the western sagebrush steppe are non-sprouters, and thus negatively impacted by fire, whereas rabbitbrush is a sprouter and can be favored by fire. Trees are underrepresented as a functional group, especially at lower elevations and PZs. Since conifers (and aspen) are more common at higher elevations and PZs, such an outcome is to be expected. A more detailed discussion of fire effects appears in Appendix 1.

Grazing

In general, there were fewer mixed results for the grazing scorecard compared to the fire scorecard (Table 6). Grazing had variable effects on annual grasses, the exception being in the <10 PZ, where the tendency was for grazing to reduce annuals. Annual grasses tended to be under-represented compared to other functional groups in the grazing scorecard. This is surprising given the ubiquitous nature of cheatgrass in particular.

Grazing tended to have neutral or positive effects on perennial grasses, with the positive effect being more pronounced at higher elevations and PZs, especially the >16 PZ. Forbs had more mixed responses to grazing than other functional groups, at least on a proportional basis. For results which did not fall into the mixed category, forbs had a variable response to grazing. Trees are generally not represented in the grazing literature (we found only one exception). A more detailed discussion of grazing effects can be found in Appendix 2.

Seeding

Perennial grass was the functional group with the highest representation across study sites (Table 7). Since over 70% of seeding studies include perennial grass as either all or part of the seeding mix, this result is not a surprise. There were a substantial number of mixed scores, which may be a result of seeding a variety of species, using a variety of techniques and seeding rates, or seeding in multiple years. Success in one year and failure in another would yield mixed scores. We would have expected a higher success rate at higher elevations and PZs (compared to lower and dryer sites), but that result was not evident from the scorecard summary (Table 7). A more detailed seeding practices summary can be found in Appendix 3.

Mechanical

The largest proportion of mechanical treatment studies was in the 12-16 PZ and thus the two higher elevation categories (Table 8). Overall, the functional group responses tended to be variable, with many entries in the mixed category. In the <10 PZ there was a substantial number of negative entries for shrubs. Many of the treatments in this PZ were likely focused on shrub control, so this result should not be surprising. There were also negative entries for shrubs in the 10-12 and 12-16 PZ, but results were more variable in these zones. A detailed analysis of mechanical treatments can be found in Appendix 4.

Herbicide

Annual grasses were the focus of many studies listed in this category (Table 9). The majority of these studies yielded mixed results for effects on annual grasses. This result may stem from the fact that many herbicide studies use multiple rates, some of which are effective while some are intended to be too light to be effective. This is a clear case where further investigation into individual studies is necessary. There are also studies where annual grasses may not be the target species (2,4-D studies for example), but annual grass abundance may still be reported. The scorecards report all functional groups reported in a study, regardless of which group was targeted by the specific herbicide. A detailed analysis of herbicide treatments can be found in Appendix 5.

Cautions about Scorecards

We talked previously about some of the reasons it was very difficult to develop a consistent summary of functional group responses to the various practices. The large number of mixed responses in some of the scorecards should reinforce that point. While the exercise of developing scorecards was useful, it clearly does not alleviate the need to review the scientific articles (or at least summaries of the articles). In the next section we will outline a streamlined approach to using the Sage-SHARE database to extract more detailed information from the individual publications.

Using the Sage-Share Database: Alternatives to the Scorecard Approach

The scorecard approach outlined in the last section will have limited utility in the analysis of scientific literature related to rangeland management practices. However, the exercise was necessary to discover that information. This limitation simply means that a more in-depth analysis of individual articles will be necessary. In this section we propose two options for using the database to help separate articles into logical groupings and then to use ecological knowledge to summarize the articles within a group. In both cases, it will be necessary to use the Sage-SHARE database to do the sorting. The *Sage-SHARE User's Manual* can be found at www.oregonstate.edu/dept/EOARC/ under "Resources" to the right of the front page, and then under "Sage-SHARE". As an example, one might want to sort by seeding and PZ 10-12. An option would be to put the information from each individual article into an Excel spreadsheet. The spreadsheet could be structured to show the following in one row: source ID (from the database), citation, treatment, elevation, precipitation, dominant shrub, perennial grasses, abstract, and summary. This is an approach we have used to organize articles for review. The capture and transfer of information from Access to Excel can be complicated. We placed a detailed set of instructions on the EOARC website, next to the Sage-SHARE user's manual referenced earlier in this section. The more "old-school" approach is to print out the abstract or entire article and have them available for further sorting and analysis. There could be manual sorts based on soil type, seeding mix, and so on.

Table 5. Fire Scorecard by (A) Elevation (ft) and (B) Precipitation Zone (in).

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass	1		1	14	16
Forb	1		2	2	5
P. Grass		2		3	5
Shrub		4	2	8	14
Tree		2		1	3
<4000					
A. Grass		5	1	9	15
Forb	2	2	3	4	11
P. Grass		7	9	12	28
Shrub		8	1	7	16
Tree				2	2
4000-5500					
A. Grass	1	10	7	32	50
Forb	7	5	10	13	35
P. Grass	4	9	18	49	80
Shrub		11	5	25	41
Tree			1	6	7
>5500					
A. Grass		3	2	43	48
Forb	10	3	24	18	55
P. Grass	2	7	8	38	55
Shrub		18	7	45	70
Tree		2	1	13	16

(B)

PZ	0	-	+	Mixed	Total
Unknown					
A. Grass	1	2	1	15	19
Forb		1	2	1	4
P. Grass		4	1	6	11
Shrub		4	1	3	8
Tree		1		2	3
<10					
A. Grass		10	2	25	37
Forb	1	4	3	6	14
P. Grass		10	8	16	34
Shrub		15	2	16	33
Tree				6	6
10-12					
A. Grass	1	2	1	24	28
Forb	4	4	8	9	25
P. Grass	2	1	11	21	35
Shrub		6	4	24	34
Tree		2		6	8
12-16					
A. Grass		4	7	26	37
Forb	13	1	23	16	53
P. Grass	4	10	10	48	72
Shrub		13	8	40	61
Tree		1	2	6	9
>16					
A. Grass				8	8
Forb	2		3	5	10
P. Grass			5	11	16
Shrub		3		2	5
Tree				2	2

Table 6. Grazing Scorecard by (A) Elevation (ft) and (B) Precipitation Zone (in).

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass		1	1	3	5
Forb		2	3	2	7
P. Grass	2	2	1	6	11
Shrub				3	3
<4000					
A. Grass		1		1	2
Forb		1			1
P. Grass	2	3	2	1	8
Shrub			2	1	3
4000-5500					
A. Grass	2	3	2	9	16
Forb	2	3	3	12	20
P. Grass	10	5	21	11	47
Shrub	2	3	23	7	35
Tree				1	1
>5500					
A. Grass		3	1		4
Forb	2	1	1	13	17
P. Grass	8	1	10	6	25
Shrub	3	3	15	7	28

(B)

PZ	0	-	+	Mixed	Total
Unknown					
A. Grass		1	1	1	3
Forb		3	1	2	6
P. Grass		1		5	6
Shrub			1	2	3
<10					
A. Grass		5		6	11
Forb		1		9	10
P. Grass	14	5	11	3	33
Shrub			24	3	27
10-12					
A. Grass	2	1	2	2	7
Forb	3	1	2	3	9
P. Grass	4		6	10	20
Shrub	4	2	1	6	13
Tree				1	1
12-16					
A. Grass			1	2	3
Forb			2	7	9
P. Grass	2	2	5	3	12
Shrub	1		8	3	12
>16					
A. Grass		1		2	3
Forb	1	2	2	6	11
P. Grass	2	3	12	3	20
Shrub		4	6	4	14

Table 7. Seeding Scorecard by (A) Elevation (ft) and (B) Precipitation Zone (in).

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass	1				1
Forb				1	1
P. Grass		1		1	2
Shrub			2	4	6
<4000					
A. Grass		3	1	7	11
Forb	2		2	2	6
P. Grass			7	14	21
Shrub		1	2	2	5
4000-5500					
A. Grass		1	6	14	21
Forb			1	7	8
P. Grass	2	5	8	30	45
Shrub	2	4		10	16
Tree				1	1
>5500					
A. Grass		1	3	6	10
Forb	1	5		9	15
P. Grass	2	2	4	20	28
Shrub	3	6		11	20
Tree			5		5

(B)

PZ	0	-	+	Mixed	Total
Unknown					
A. Grass	1				1
P. Grass		1		1	2
Shrub		1	1	1	3
<10					
A. Grass		1	2	6	9
Forb			2	7	9
P. Grass		1	2	9	12
Shrub		2	3	7	12
10-12					
A. Grass		2	2	3	7
Forb	1			3	4
P. Grass	2		14	14	30
Shrub	4	1		10	15
12-16					
A. Grass		2	6	12	20
Forb	2	5	1	6	14
P. Grass	1	6	2	24	33
Shrub	1	6		9	16
Tree			5		5
>16					
A. Grass				6	6
Forb				3	3
P. Grass	1		1	17	19
Shrub		1			1
Tree				1	1

Table 8. Mechanical Scorecard by (A) Elevation (ft) and (B) Precipitation Zone (in).

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
Forb			2		2
P. Grass			2	1	3
Shrub		5		8	13
Tree		1			1
<4000					
A. Grass		2		4	6
Forb		2	1	2	5
P. Grass		2		3	5
Shrub		2	1	6	9
Tree				2	2
4000-5500					
A. Grass	1	3	7	18	29
Forb	5	2	12	19	38
P. Grass	3	7	7	28	45
Shrub		11	4	19	34
Tree				5	5
>5500					
A. Grass	3	3	2	22	30
Forb	3	9	12	14	38
P. Grass		5	4	27	36
Shrub		10	7	36	53
Tree		2	6	9	17

(B)

PZ	0	-	+	Mixed	Total
Unknown					
Forb			1		1
P. Grass				2	2
Shrub		1	1	1	3
Tree		1			1
<10					
A. Grass	1	4	3	9	17
Forb	3	4	9	7	23
P. Grass	3	5	5	9	22
Shrub		12		13	25
Tree				5	5
10-12					
A. Grass		3		12	15
Forb		1	4	8	13
P. Grass		1	3	11	15
Shrub		4	5	24	33
Tree		1		4	5
12-16					
A. Grass	3	1	6	19	29
Forb	5	8	10	15	38
P. Grass		8	3	30	41
Shrub		11	6	29	46
Tree		1	6	6	13
>16					
A. Grass				4	4
Forb			3	5	8
P. Grass			2	7	9
Shrub				2	2
Tree				1	1

Table 9. Herbicide Scorecard by (A) Elevation (ft) and (B) Precipitation Zone (in).

(A)

Elevation	0	-	+	Mixed	Total
Unknown					
A. Grass				1	1
Forb			2	1	3
P. Grass			1	1	2
Shrub	1	3		5	9
Tree				1	1
<4000					
A. Grass		12		11	23
Forb		5		11	16
P. Grass		2		7	9
Shrub		2	3	1	6
4000-5500					
A. Grass		4	2	23	29
Forb	1	4	1	12	18
P. Grass	2	5	1	11	19
Shrub	1	5	1	5	12
>5500					
A. Grass		2		23	25
Forb		2			2
P. Grass		2	2	6	10
Shrub	1	9		2	12

(B)

PZ	0	-	+	Mixed	Total
Unknown					
A. Grass				1	1
Forb				1	1
Shrub	1			3	4
Tree				1	1
<10					
A. Grass		8		20	28
Forb		7		7	14
P. Grass		5	3	4	12
Shrub		6	3	3	12
10-12					
A. Grass		2	2	14	18
Forb	1	3	1	5	10
P. Grass		3	1	6	10
Shrub	1	5	1	4	11
12-16					
A. Grass		2		12	14
Forb		1	1	3	5
P. Grass	1	1		4	6
Shrub	1	7		3	11
>16					
A. Grass		6		11	17
Forb			1	8	9
P. Grass	1			11	12
Shrub		1			1

Conclusion

Our general conclusions are that the database is a great way to evaluate the distribution of studies, and to find groupings of articles based on parameters of interest. The value of a database as opposed to piles of articles in various categories is that additional sorting can be done with little effort. However, once the sorting is completed, there is no substitute for reading the article or at a minimum the abstract and using knowledge to assess how the articles in a category relate to each other and what value they hold for making management decisions.

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Appendix 1: Fire Treatment Literature Summary

Authors: Chad Boyd, USDA-ARS and Sara Holman, OSU

Introduction

Plant communities of the western sagebrush steppe have evolved with wildfire as a natural disturbance process (for a good review, see Crawford et al. 2004). Historically, fire kept fire-sensitive native juniper populations in check and promoted a diversity of habitat structure and composition at large spatial scales by modulating the relative abundance of perennial grasses and shrubs. Big and low sagebrush species are easily killed by fire, while native perennial grasses are less sensitive and may actually increase post-fire with sufficient moisture; particularly on cool and moist sites typical of higher elevations. Since European arrival, fire regimes have changed dramatically across the region. In juniper prone environments, historic livestock grazing and, later, increased fire suppression effort and effectiveness resulted in dramatically decreased fire frequency; an effect that continues to this day. With a reduced presence of fire, native juniper species have greatly increased across the region, often to the detriment of understory herbaceous and shrub vegetation. At higher elevations, the arrival and proliferation of various exotic annual grass species has promoted more frequent, and in recent years, larger wildfires. These systems are less resilient and resistant to annual grass invasion as compared to more productive and more mesic higher elevation locations. Desired native perennial grasses and shrubs have declined with more frequent fire and are being replaced by non-desired exotic annual species (Mote et al. 2013).

However, prescribed fire is still used as a treatment on sagebrush rangelands to control juniper populations, mitigate fuel build up, and in some cases as a component of integrated management treatments for controlling exotic annual grass abundance. Interpreting the results of prescribed fire treatment is made difficult by the fact that those results are contingent on a number of abiotic and biotic factors including temperature; relative humidity; wind speed; fuel load amount, type, and moisture content; soil type and water holding capacity; plant community type and successional stage; and post-fire climate. Thus, when and where a fire occurs can dramatically affect post-fire vegetation dynamics. Season of burn is also important. For example, medusahead is best controlled by burning in late spring when the seeds are still in the grass canopy and other species' seeds have dropped to the soil surface; this is because flame temperature tends to be hotter in the canopy versus the soil surface—enough to kill the medusahead propagules but not most of the more desirable species (Kyser et al. 2014).

Another factor that complicates interpretation of fire effects on plant communities is the fact that topical literature includes studies on both wildfire and prescribed fire. Wildfires often burn under extreme weather and fuel conditions that differ from conditions typical of prescribed fires. Thus, the effects of wildfire and prescribed fire can be very different. Within the wildfire literature, studies often seek to interpret fire effects by comparing post-fire vegetation in burned areas to that of unburned islands within the fire perimeter. This can be problematic because there is often a fuel load-related reason (e.g., reduced fine fuels) as to why islands of vegetation remain unburned, making the "control" areas less valuable for elucidating wildfire effects.

In recent years, fire as a management treatment has often been viewed in a negative light due to its association with increased annual grass abundance in low and mid elevation sites, and due to the fact that fire-induced sagebrush reduction has negative effects on habitat quality for sagebrush obligate wildlife species such as greater sage-grouse. With regard to the latter, it is important that managers think about the effects of fire on wildlife habitat within the context of ecologically relevant timelines. In other words, while the short-term negative effects of fire on sagebrush are very real, the long-term reduction in conifer populations with fire is undeniable. In fact recent published literature suggests that fire may be a necessary component of juniper control over long time horizons and at large scales, due to increased duration and effectiveness of control relative to more logistically intensive and temporally transient mechanical control practices (Boyd et al. 2017).

Treatment Combinations with Fire

Prescribed burns can be costly, and depending on the condition of the site prior to treatment, may need to be repeated. Prescribed burns may work better if combined with other treatments such as cutting a portion of the juniper population prior to burning on high elevation sites (Sheley and Bates 2008; Bates et al. 2011) or seeding desired species after burning if the site is void of a native perennial seedbank (Roche et al. 2008). Davies and Sheley (2011) and Sheley et al. (2012) showed that fire applied prior to application of imazapic (a soil active pre-emergent herbicide) had the best results for controlling exotic annual grasses and promoting desired species.

Invasive Annual Grass Threat

At low elevations, conditions tend to be warmer and dryer, with increased probability of post-fire annual grass invasion or increase. This is especially true when invasive annuals dominate the pre-fire plant community and desired perennial grasses are in low abundance (habitat condition C or D). Lower elevations are less resilient to disturbances, resulting in a higher chance for annual exotics to take over and/or bare ground to increase following fire (shift to D habitat condition because sagebrush experiences high mortality during fire) (Miller et al. 2014). There is a positive feedback between fire disturbance and cheatgrass cover—cheatgrass is an easily ignited, often continuous and very dry fuel so fire spreads quickly, but does not burn long enough to kill the seeds, creating post-fire conditions that favor cheatgrass germination and growth (Briske 2011). In habitat conditions A or B, fire can result in increased perennial grass cover and density which would lead to a B habitat condition post-burn (Davies et al. 2012). In some instances, post-fire seeding has helped to increase the abundance of desired perennial plant species (Hilty et al. 2004; Davies et al. 2013).

Success of seeded perennial grasses within burned sagebrush communities was higher under shrub subcanopies than in the interspaces (Boyd and Davies 2010). This could be a result of in-creased available nutrients and warmer spring-time temperatures for subcanopy soils that are high in organic matter and often have a blackened appearance following fire (Boyd and Davies 2012). Conversely, perennial bunchgrass mortality during fire is predominantly in locations under sagebrush subcanopies where heat loading is much higher than adjacent interspace locations (Boyd et al. 2015; Hulet et al. 2015).

Invasive Annual Grass/ Conifer Expansion Threat

Within the invasive annual grass/ conifer expansion threat model, fire can be used to control juniper and some weed species. It is similar to the invasive annual threat-based model in that sites with a strong pre-fire annual grass presence are likely to result in an annual-dominated post-fire habitat condition (habitat condition E). Prescribed fire in annual grass prone/conifer systems should be used with extreme caution and only on sites with sufficient pre-burn abundance of desired species, or on sites where post-burn annual grass control and seeding have been planned.

For sites with at least some native perennial cover (habitat conditions A, B, and C), there is a greater chance perennials will recover. In some cases, perennial grasses can ultimately dominate and increase in post-fire cover and density, resulting in habitat condition B even if the first year post-burn showed an increase in annuals (West and Hassan 1985; Hosten and West 1994; Davies et al. 2007; Ellsworth and Kauffman 2013). This seems to occur after three growing seasons following burning and after an initial decline in perennial grasses (Bates and Svejcar 2009; Bates et al. 2011; Miller et al. 2014). Although successful in controlling juniper expansion, recovery to habitat condition A conditions following fire may take decades (West and Yorks 2002; Beck et al. 2009).

In some studies, post-fire herbaceous production increased including root and shoot mass, and culm and seed head count (Young and Miller 1985; Jirik and Bunting 1994; Wrobleski and Kauffman 2003; Ellsworth and Kauffman 2010). Ratzlaff and Anderson (1995) found sufficient post-fire vegetation recovery to stabilize soils in the absence of seeding. Nitrogen availability and mineralization may increase significantly post-fire (Ellsworth 2006; Blank et al. 2007; Davies and Bates 2008; Goergen and Chambers 2012), which is often associated with short-term post-fire increases in nitrophilic annual plant species.

Conifer Expansion Threat

In higher elevation locations with conifer expansion threat, but with little threat from annual species, fire is less risky due to slightly cooler and wetter soil conditions that limit survival of fall-germinated cheatgrass seedlings and promote productivity of desired perennial plants. In these environments, fire is considered an effective means of controlling juniper expansion. One study demonstrated that sagebrush mortality during fire increased with juniper density (Barney and Frischknecht 1974). Others found that forage production, quality, and insect abundance—specifically ants and beetles—increased for a short period after fire (Cook et al. 1994; Nelle et al. 2000). However, sage-grouse use of recently burned areas may decrease in association with decreased post-fire sagebrush abundance (Benson et al. 1991; Byrne 2002). Soil nutrients were also found to be higher after burning at high elevations (Chambers et al. 2007; Rau et al. 2008).

A few long-term studies showed that more than 20 years was needed for sagebrush to recover enough to be considered preferred sage-grouse nesting habitat or meet habitat condition A requirements (McDowell 2000; Nelle et al. 2000). Not only was sagebrush severely reduced in terms of cover and density, but the seedbanks were also mostly eliminated as a result of fire, so sites were most likely left in habitat condition B and would require seeding (Allen et al. 2008).

As with the low and mid elevation studies, the pre-burn plant species assemblage strongly influences post-burn species composition (Wright and Chambers 2002; Seefeldt et al. 2007; Condon et al. 2011; Bates et al. 2014). Thus, caution should be exercised with fire in habitat condition E sites due to the potential for post-burn increases in non-desired species and/or soil erosion in the absence of sufficient post-fire vegetation ground cover (Rau et al. 2005; Pierson et al. 2014). Habitat conditions C and D, however, may benefit from prescribed burning and revert to habitat condition B conditions in the short term or habitat condition A with sagebrush recovery/seeding over a longer period.

Conclusions and Further Research

Overall, fire treatments appear to be successful in terms of promoting native perennial grass abundance as long as sufficient perennial plant numbers are present pre-fire (Chambers et al. 2014; Davies et al. 2014; Pyke et al. 2014). This generalization should be modified in accordance with the extent of perennial grass mortality during fire, which is influenced by fire weather conditions and fuel load composition and structure. Fire tends to result in increased soil nutrient availability, which can promote increases in annual plant species (Hobbs and Schimel 1984; Rau et al. 2007). Season of burn can impact both the degree of control of invasive annuals as well as juniper control. Sagebrush mortality is generally high regardless of fire season.

Climate variability can promote inconsistent burning results between studies (Jessop and Anderson 2007; Ziegenhagen and Miller 2009; Rau et al. 2014). It is recommended that fire be used at higher elevations where the climate is cooler and moister, especially as a means of controlling juniper. Fire is not recommended to control annual grasses, especially at lower elevations, unless it is used as a pre-treatment for herbicide-based annual grass control. Although Blank et al. (1994), Davies et al. (2008), and Diamond et al. (2012) showed success in controlling cheatgrass, this was only short term. When combined with imazapic, burning appears to control medusahead and allows for perennial grasses to return, particularly when followed up by seeding of desired species (Davies 2010).

Many studies monitor post-burn effects through three years while few look at the impacts of fire beyond a decade. More long-term studies are required to learn how fire affects the return of desired species versus exotic invasive species and the successional timeline for sagebrush.

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Appendix 2: Grazing Treatment Literature Summary

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Introduction

Grazing is a complex issue from both a process and an impact point of view. We will outline: 1) a brief history of grazing literature, 2) some of the approaches to grazing management and thus grazing research, 3) interacting factors in the analysis of grazing (e.g., timing, intensity, and duration), and 4) review the results from our database in terms of grazing and the major vegetation threats to GRSG habitat.

First, we need to be clear on the limitations placed on the Sage-SHARE database with regard to grazing. The database is restricted to readily available peer-reviewed scientific literature, and is focused on sagebrush steppe plant communities. There is a great deal of information contained in conference proceedings, agency or university publications and other “grey” literature. We do not mean to imply that such information is not useful, but the database was restricted to scientific journal publications. There is also a good deal of grazing literature on riparian systems in the western sagebrush steppe available, but our focus was on the upland sagebrush steppe communities.

Much of the very early work brought attention to the negative impacts of heavy, unmanaged grazing during the Euro-American settlement period in the western sagebrush steppe (e.g., Griffith 1902). Much of this material was published in government documents and was observational rather than scientific in nature. With the passage of the Taylor Grazing Act in 1934, there was more focus on defining animal production and sustainable grazing, although many studies did not meet modern standards for experimental design and statistical analysis. Jones (2000) points out that there is still room for improvement in the design and analysis of grazing studies. We will address that issue later in this discussion. There is clearly overlap, but the main focus of the grazing literature evolved from 1) demonstrating impacts of uncontrolled grazing to 2) minimizing negative grazing impacts and maximizing animal production, 3) sustainable grazing management, and finally 4) targeted grazing to meet vegetation objectives.

There are peer-reviewed publications that have highlighted the negative impacts of grazing (e.g., Fleischner 1994; Beschta et al. 2012), although a portion of that discussion is focused on riparian systems.

One of the problems with separating negative impacts from neutral or positive impacts has to do with the complex nature of grazing and the fact that many grazing treatments are poorly defined. There are many “exclosure” studies that involve comparison of grazed vs. un-grazed treatments, although grazing level in the grazed—ungrazed comparison is often not defined. Stocking rate is clearly an important consideration in evaluating grazing effects, but so are timing, duration of grazing, and class of livestock (e.g., cattle, sheep, or goats). Since many of these terms have been formally defined, we will include a few general definitions here for those not familiar with the terminology. These and many more grazing-related terms can be found in Allen et al. (2011):

- *Deferment*: the postponement or delay of grazing to achieve a specific management objective;
- *Rest*: to leave an area un-grazed for a specific period of time (e.g., year, season, etc.);
- *Rest period*: the length of time that a specific area is not stocked between stocking periods;
- *Stocking period*: the length of time that grazing livestock or wildlife occupy a specific area;
- *Stocking season or grazing season*: the time during which grazing can be practiced;
- *Stocking rate or stocking density*: the relationship between number of animals and area of land where density is generally an instantaneous value (number of animals per acre right now) and rate has a time element (number of animals per acre for a month as an example);

- *Carrying capacity*: the maximum stocking rate that will allow a target level of animal performance without deterioration of the land; and
- Grazing pressure: the relationship between animal live weight and forage mass per unit area for a specific land unit at a point in time.

There are examples in the literature where the various parameters interact. For example, research by Ganskopp et al. (1999) shows that timing of grazing is critical in terms of influence on young bitterbrush plants. Grazing before grasses produce seed heads favors growth of bitterbrush (cattle graze grass that competes with the shrubs), but once grasses produce seed heads they become less palatable and cattle tend to graze the bitterbrush. In this example, grazing can have either positive or negative effects on bitterbrush growth depending on timing. The older literature demonstrated clearly that heavy, season-long grazing tends to have very negative effects on sagebrush steppe vegetation (Griffiths 1902; Anderson and Holte 1981; Brotherson and Brother-son 1981).

There are some practical and logistical problems with grazing studies. In the relatively arid sagebrush steppe, replicated grazing studies require substantial land area in addition to human and monetary resources. For example, if rangeland produces 500 pounds of forage per acre, and utilization is set at 50%, an acre would produce about 250 pounds of usable forage. If cattle intake rates were about 25 pounds of forage per head per day, an acre would support one animal for 10 days. If four animals were to graze an area for a month, each replicated pasture would need to be 12 acres. Experimental research requires replication. Given the variable nature of the sagebrush steppe, a minimum of four replications is desirable, thus, one grazing treatment would require 48 acres. In addition, fencing, water delivery systems, livestock handling and so on would be required. And this is only one grazing treatment. We would also need some sort of control treatment. For these reasons, studies with multiple levels of stocking rate, timing, and duration can become logistically challenging very quickly. There are creative approaches to grazing research which may involve observational studies, artificial defoliation (clipping or mowing), or grazing few animals for short periods. However, to truly understand the effects of grazing, studies with standard experimental designs are an important part of the mix.

Grazing Treatment

Successful grazing for sagebrush habitat conservation and restoration depends on the type of live-stock used, timing and duration of grazing, stocking rate, disturbance regime, and other management methods used in conjunction with grazing. The disturbance regime plays a major role—for example, wildfire changes ecosystem dynamics, so post-fire parcels should be treated differently than unburned areas. Because grazing methods can vary widely from site to site in addition to varied plant composition, soils, and precipitation, it is difficult to conduct repeat studies involving grazing treatments. Risks of grazing include soil compaction, trampling of desired species, negative interactions with native wildlife, and shifting herbaceous species composition.

Grazing Options*

- Stocking rates: Levels vary depending on available forage, terrain and grazing distribution, timing, and other variables such as riparian areas within pastures;
- Continuous vs. seasonal: Continuous grazing may involve grazing during the entire growing period or entire year depending on the area whereas seasonal grazing is defined as grazing an area during a specific period or season (stocking season);
- Rotation: Moving livestock from one parcel to another (Howery et al. 2000).

*Rest period, stocking period, stocking season, and grazing pressure vary in grazing studies as discussed earlier.

Depending on the environment, preferred management outcome, and desired level of involvement, a variety of grazing strategies can be applied. There has been an active debate over the years about the value of rotational grazing systems relative to continuous grazing (Heady 1961; Briske et al. 2008).

Combinations with Grazing

The most common treatments studied with grazing were prescribed fire and herbicide. Grazing post-fire requires extra attention because of the delicate state of the ecosystem. Mechanical treatments to control juniper combined with grazing can stimulate herbaceous seed production, but rest is needed to allow perennial grasses to re-establish (Bates 2005). Mowing in addition to grazing may reduce standing dead material, making more forage available to other wildlife in the spring (Taylor et al. 2004). Johnson et al. (1980) evaluated the effects of grazing and sagebrush control on erosion potential across a variety of sites.

Invasive Annual Grass Threat

Few references were collected regarding grazing at low elevations where invasive annual grass poses the largest threat. Furthermore, the studies mainly focus on forage quality as opposed to reducing invasive annual grasses.

Of the sources that did concentrate on annual grass reduction, one study found that perennial grass and shrub recovery occurred similarly with moderate or no grazing after several decades (Courtois et al. 2004). Several of the studies focused on grazing effects on forage quality (Pitt 1986) or grazing distribution (Ganskopp and Vavra 1987). Another study found that site location affected medusahead more than intensity or timing of defoliation; specifically, harsh, clayey soils favored the annual grass, but those sites yielded 50% less annual grass density when perennial grass defoliation occurred in the fall when compared to spring (Sheley et al. 2008).

Invasive Annual Grass/ Conifer Expansion Threat

There was a group of articles in this threat model category that described vegetation recovery from the heavy grazing of the early to mid- 1900's (Robertson 1971; Anderson and Holte 1981; Brotherson and Brotherson 1981). A second set of studies evaluated exclosures set up after the Taylor Grazing Act in 1934 (Sneva et al. 1984; Courtois et al. 2004). The Courtois et al. (2004) article provides a good example of mixed results. They found plant cover tended to be higher in exclosures (un-grazed) and density was higher in grazed treatments.

Many results were mixed for studies conducted within the invasive annual grass/ juniper expansion threat model which had the largest number of articles falling within the defined elevation boundaries of 4000-5500 ft. Ample resting time after disturbance (fire, cutting, etc.) may be required for recovery of low condition rangeland, but high condition rangeland may recover quickly after disturbance with or without grazing (Bates et al. 2009). Grazing may reduce wildfire risk by reducing fuel loads (Davies et al. 2009, 2010; Diamond et al. 2009).

Moderate fall grazing may have a less negative impact on perennial grasses compared to spring grazing (Britton et al. 1990; Bork et al. 1998). Light to moderate grazing had limited impact on post-fire vegetation dynamics compared to no grazing, but may reduce seed production (Bates et al. 2009). As mentioned previously, grazing can have either negative or positive effects on bitterbrush growth depending on timing of grazing (Ganskopp et al. 1999). There are cases where grazing can provide a benefit to wildlife habitat by increasing shrub abundance (Austin and Urness 1995, 1998; Armstrong 2007). In some cases, grazing can disturb the ecosystem via trampling, soil degradation, and reduction in cryptogamic crusts or plant species composition shifts (Memmott 1998; West and Yorks 2002; Reisner et al. 2013).

The discussion to this point mainly refers to cattle grazing, but we should note that sheep have been shown to effectively control spotted knapweed (Olson et al. 1997; Thrift et al. 2008). Conversely, goat grazing was not effective at controlling small juniper trees on good condition rangeland (Fajemesin et al. 1996). Pederson et al. (2003) modeled interactions among sheep grazing, fire, and sage grouse populations.

Conifer Expansion Threat

Again, results within and among studies varied, which could be due to different management methods as well as variation in site characteristics and study objectives. A number of the studies included had multiple sites and represented multiple threat models. For example, Johnson et al. (1980) had sites within all three of the elevation bands we used to define the threat models.

During the 1980's and 1990's there was a good deal of attention directed toward "grazing systems". In several cases, the studies involved large grazing allotments that included both mid and high elevation sites (e.g., Eckert and Spencer 1987; Eckert and Spencer 1987; Yeo et al. 1993). In one case, the grazing system work was entirely at higher elevations (Laycock and Conrad 1981).

Studies focused on grazing did not tend to mention juniper control and mainly focused on grass species composition and cover.

Conclusions and Further Research

There are some general conclusions that could be drawn for any of the threat-based models. Grazing can reduce fuel loads and thus fire risk (Evans 1986; Davies et al. 2009, 2010; Diamond et al. 2009). In some instances, it stimulates growth and seed production, but in other instances—usually at higher stocking rates—overwhelms and reduces grass growth and seed production (Bates 2005; Bates et al. 2009). If forage is exhausted at a lower elevation, cattle may move around and up in elevation, and wind up competing with other grazers (Yeo et al. 1993).

Because of the temporal and spatial variation of the western sagebrush steppe (e.g., Svejcar et al. 2017) and the complex nature of grazing, generalizations are difficult. There is general consensus that heavy, continuous grazing is not suitable in this arid vegetation type, but few other consistent rules emerge from the literature. This is an area of management that requires flexibility. Research has been scattered, and a renewed effort focused on sustainable and outcome-oriented (targeted) grazing is warranted.

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Appendix 3: Seeding Treatment Literature Summary

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Introduction

Seeding is used to increase plant community diversity and richness, increase forage supply, fill ecological space previously occupied by undesirable plant species (i.e., invasive exotic plants), or revegetate following disturbance (e.g., fire). Outcomes of seeding treatments are often difficult to predict and success rate depends on many factors, including climatic variables, past disturbances, seedbed preparation, interaction with other treatments (e.g., fire, herbicide application), seeding rate, plant materials selection, and seeding method.

In the context of sagebrush habitat management for ecological function and sage-grouse habitat in the western sagebrush steppe, seeding typically occurs in four scenarios: 1) seeding perennial grasses and forbs to recover degraded plant communities that are or are at risk of exotic annual grass invasion (i.e., habitat condition C and D); 2) planting sagebrush to accelerate or facilitate the recovery of otherwise intact plant communities (i.e., habitat condition B) often following wildfire; 3) rehabilitation of understory plant communities following removal of juniper in juniper-encroached habitat (i.e., habitat condition E); and 4) establishing forb species in any of the aforementioned scenarios. The organization of this seeding review was driven by the relatively limited scope of published research in our project area.

Seeding in Annual Grass-Prone Plant Communities

Successful seeding in annual grass-prone—or already annual grass-dominated—sagebrush steppe plant communities has significant implications for conservation of at-risk species such as sage-grouse (Wisdom et al. 2011; Murphy et al. 2013), public firefighting expenses (Epanchin-Niell et al. 2009; Taylor et al. 2013), and viability of rural western economies (Brunson and Tanaka 2011). Unfortunately, published evidence suggests that seeding on many rangelands such as semi-arid sagebrush steppe can be challenging (Pyke et al. 2013), and estimates of success may be inflated by disproportionate reporting of seeding outcomes from periods of above average precipitation (i.e., conditions more conducive to establishment), and under-reporting of negative results (Hardegree et al. 2016). Establishment of deep-rooted perennial grasses (DRPG) has been identified as a key factor in preventing (Davies et al. 2010) and/or suppressing invasive annual grasses (Davies 2008; James et al. 2008).

Evidence from our project area suggests presence of significant temporal and spatial heterogeneity, which can be associated with seeding outcomes (Hardegree et al. 2016; Rajagopalan and Lall 1998; Svejcar et al. 2017). Temporal heterogeneity is frequently expressed via inter-annual differences in precipitation (total annual or crop year), with below average precipitation years being identified as a factor explaining seeding success, or lack thereof (see Duniway et al. 2015). Intra-annual timing of seeding is a source of temporal variation that has not been exhaustively reviewed for our project area, though Eiswerth and Shonkwiler (2006) provide evidence that seedings in north-central Nevada are more likely to be successful if implemented between October and January. Microclimatic weather patterns may affect seeded species directly via modulation of the seedbed hydrothermal environment (Rawlins et al. 2012, Hardegree et al. 2013, 2016) or via its effect on competing species, such as invasive annual grasses which are very adept at capitalizing on available soil resources (Mangla et al. 2011). Recent evidence suggests that survival of a seeded native DRPG may interact with month of planting within the late fall to early winter timeframe and year (Boyd and James 2013).

Spatial heterogeneity affects seeding outcomes at several scales. At the broadest spatial scales, patterns of elevation and precipitation zone appear to strongly affect DRPG seeding outcomes in post-fire seeding scenarios (Koniak 1983; Knutson et al. 2014). Within seeding project areas, topography (Kulpla et al. 2012), spatial heterogeneity of soils, and soil surface characteristics which may be naturally occurring or a result of previous anthropogenic disturbance (Morris et al. 2014), alter seed placement in addition to hydrothermal environment and outcomes of seeding DRPGs. Though out of our study area, Chambers' (2000) thorough evaluation of soil characteristics on seed entrapment, emergence, and survival suggests that seeding outcomes of several common native bunchgrass species are altered by the effect of soil capacity to retain moisture (via texture, large holes, or mulch) during the period following germination and prior to emergence. Boyd and Davies (2012) similarly concluded that soil particle size as it affects available soil moisture influenced DRPG seeding outcomes in southeast Oregon.

Additionally, these authors quantified the large magnitude of spatial heterogeneity of seeding outcomes. Spatial patterning of pre-fire vegetation, particularly big sagebrush, could also alter seeding outcomes. Boyd and Davies (2010) noted that DRPG seedlings were more abundant and vigorous in the sub-canopy of burned big sagebrush, which was attributed to small-scale differences in soil temperature and color (Boyd and Davies 2012). Eiswerth et al. (2009) found that seeding of grasses and forbs was more successful on burned sites that had a shrub component pre-fire and suggested that seeding immediately after a fire may represent a closing window of opportunity to capitalize on favorable soil conditions.

Successfully seeding desired species into annual-grass invaded sagebrush sites typically requires controlling the annual grasses to allow seeded species to survive (Klomp and Hull 1972; Young 1992; Davies 2010; Nafus and Davies 2014). Though numerous approaches have been attempted, the most common control methods applied are prescribed burning, pre-emergent herbicide application or a combination of both. Prescribed fire and herbicides individually have been used with moderate success to suppress invasive annual grasses temporarily (Young et al. 1972; Davies and Sheley 2011; Kyser et al. 2013), but the most effective treatment combinations identified thus far tend to be, in sequence, burning, application of pre-emergent herbicide, one year fallow, and seeding (Davies 2010; Davies et al. 2014, 2015).

However, in the preceding examples, a highly competitive non-native DRPG (crested wheatgrass) was the most effectively seeded species while encouraging the highest rate of reduction in invasive annual grasses in a direct comparison (Davies et al. 2015). At sites in central and eastern Oregon, Sheley et al. (2007) burned, sprayed, and seeded in the same year with positive results for some species while other species were negatively affected by herbicide application. Seeding immediately following herbicide application has yielded mixed results. Sheley et al. (2012) successfully seeded a non-native DRPG into medusahead-dominated sites in central Oregon, with less success using native species. Their results also showed some patterns of spatial heterogeneity with an interaction between the application of pre-emergent herbicide and seeding outcomes depending on site and herbicide application rate.

Though outside our study area, Morris et al. (2009) also reported that spraying pre-emergent herbicide and seeding in the same year could be effective in west-central Utah sagebrush habitat depending on herbicide application rate. Davies and Bates (2014) evaluated the efficacy of mowing in Wyoming sagebrush plant communities with a sagebrush overstory and depleted understory at risk of invasive annual grasses prior to seeding without success.

Experimental evidence pertaining to setting seeding rates, despite being a key decision in any seeding project, is limited in our study area. In a review of seeding projects in northern Nevada, Eiswerth and Shonkwiler (2006) suggest that seeding rate increases the density of seeded non-native DRPGs, up to a point of diminishing returns in the range of 20-27.5 PLS ft-2. Evidence from annual-grass dominated sites in central Oregon suggested variable response per seeding rate, depending on site and herbicide treatments (Sheley et al. 2012). Field tests by Sheley and Bates (2008) showed that the higher densities of seeded species were achieved with higher seeding rates. However, these data were collected from more mesic sites at higher elevations in southeast Oregon/southwest Idaho that may be more conducive to seeding survival.

Seeding Methods

Direct comparisons of the most common seeding methods—broadcast and drill-seeding—are few in general and in our study area specifically. Drill seeding is generally believed to be more effective than broadcast seeding, given the more favorable seedbed environment provided by seeding to a desirable depth (Hardegree et al. 2016). Several of the most comprehensive evaluations of multiple seeding projects in our study area (Eiswerth and Shonkwiler 2006; Eiswerth et al. 2009) do not provide direct comparisons or are limited to a single method. A wide-ranging review of historic post-fire seeding projects (Knutson et al. 2014) does evaluate common environmental and response variables among disparate broadcast and drill seeding treatments. Generally, cover of seeded native and non-native DRPGs increased with seeding and responded positively with increasing elevation and precipitation zone. Notably, native DRPGs drill seeded alone may perform better than when seeded in conjunction with highly competitive non-native DRPGs, such as crested wheatgrass. This outcome is consistent with site-specific evidence from southeast Oregon (Nafus et al. 2015). Kyser et al. (2013) compared broadcast seeding alone with broadcast seeding followed by a raking treatment to increase seed-to-soil contact on northern California study sites dominated by invasive annual grasses. Neither method effectively established seeded perennial species. An important consideration when co-paring seeding methods is the unintended consequences of physical disturbance associated with drill seeding (Pierson et al. 2007), though these concerns may be dependent on specific drill technologies used and site characteristics (Hardegree et al. 2016).

Non-native DRPGs, such as crested wheatgrass, have been used extensively in sagebrush habitat within our study area to achieve management objectives including, but not limited to preventing soil erosion, replacing or precluding invasive annual grasses, and improving forage for livestock production (Davies et al. 2011). Crested wheatgrass' ease of establishing from seed relative to many native DRPGs and persistence of stands (Robins et al. 2013) is advantageous in scenarios where invasive annual grasses are a risk or already dominate on a site. Nonetheless, seeding non-native DRPGs are not immune to failure under challenging conditions (Roboker and Schirman 1976).

Results from a broad review of seeding outcomes in the Great Basin by Knutson et al. (2014) suggest that seed mixes including both native and non-native DRPGs ultimately lead to plant communities dominated by non-native DRPGs, which is supported by experimental evidence of seeded crested wheatgrass recruiting and establishing more consistently than seeded native species (Nafus et al. 2015). More recently, some published literature has examined the viability of an assisted succession strategy in crested wheatgrass or cheatgrass monocultures (Cox and Anderson 2004; Fansler and Mangold 2011). Several rates of herbicide application or disking failed to suppress crested wheatgrass or improve native seeding establishment in two different years in southeastern Oregon (Fansler and Mangold 2011). Seeding native species into established crested wheatgrass stands outside our study area in Utah also had little success recruiting seeded native species, though drought conditions may have contributed to this outcome (Hulet et al. 2010). Evaluating native species for specific characteristics, such as those exhibited by crested wheatgrass (e.g., seedling vigor, root extension, etc.), that contribute to successful recruitment via seeding may provide insight into the choices that increase native seeding outcomes (Jones 1998; Leger 2008).

Establishing Sagebrush from Seed

Published research from our study area suggests that establishing sagebrush from seed is challenging, though results are variable in space and time. The outcomes are similar for DRPGs, though even less research is available from which to draw conclusions. A report by Lysne and Pellant (2004) evaluated 35 post-fire sagebrush seeding projects in Idaho sagebrush steppe habitat. They found that sagebrush cover and density did not differ significantly from untreated areas, though seeding did result in some patches of effective shrub recruitment. Evidence of temporal variation in sagebrush recruitment following wildfire from existing seed banks suggests that sagebrush seeding outcomes may also be highly dependent on year effects (Ziegenhagen and Miller 2009).

Establishing sagebrush on sites dominated by annual grasses or non-native DRPGs may be especially difficult without effectively reducing competing species. A recent evaluation of restoration treatments in frequently burned sagebrush habitat with abundant cheatgrass in Idaho found no evidence of drill seeded sagebrush emergence (Brabec et al. 2015). Davies et al. (2013) attempted to establish sagebrush via broadcast seeding on sites dominated by crested wheatgrass. Even with attempts to suppress crested wheatgrass dominance via herbicide application, sagebrush recruitment from seeding was absent at low levels of crested wheatgrass control and extremely low at the highest levels of crested wheatgrass control. Two studies outside of our study area reaffirm the hypothesis that sagebrush establishment in annual grass-dominated sites is unlikely. Kyser et al. (2013) attempted to establish sagebrush via broadcast seeding into annual grass-dominated sites in California without any success. Owen et al. (2011) similarly failed to establish sagebrush via drill seeding following herbicide treatments.

Understory Restoration Following Juniper Removal Treatments

In sagebrush habitat degraded by juniper or other conifer encroachment, seeding may be attempted in order to recruit desired species into the understory following juniper removal via fire or mechanical removal. Distribution of encroaching conifer tends to correspond with increased site potential, elevation, and/or precipitation zone. As such, research results from our study area suggest that seeding may be less prone to failure.

Following prescribed fire in southwest Idaho to reduce juniper cover, Sheley and Bates (2008) successfully broadcast seeded three native bunchgrasses and one forb—yarrow—but failed to establish two other forbs via seeding (arrow leaf balsamroot and Lewis flax). Davies et al. (2014) successfully established several native and non-native perennial grasses and sagebrush by broadcast seeding after prescribed fire applied to reduce juniper cover, but were unsuccessful with one native forb—Lewis flax—and one non-native forb—alfalfa.

After mechanical treatments to remove juniper in central Oregon, broadcast seeding two native grasses and one native forb was marginally successful (Kerns and Day 2014), though it is important to note that invasive annual grasses were abundant. Although outside of our study area, Young et al. (2013) successfully established a native DRPG following juniper mastication, though cheatgrass was also favored.

Seeding Forbs in Sagebrush Steppe Habitat

Field studies evaluating outcomes of seeding forbs are relatively limited in our study area. Research on seeding in sagebrush habitat in our study area published through approximately the mid-1970’s predominately related to establishment on non-native DRPGs, particularly crested wheatgrass, as a means to improve livestock forage on degraded sites. More recent interest in seeding forbs into degraded sagebrush steppe habitat is overwhelmed by mixed results, particularly on more arid potential Wyoming sagebrush plant communities relative to more mesic potential mountain sagebrush communities. In their review of seeding in northern Nevada, Eiswerth et al. (2009) concluded that seeding of forbs, as well as grasses, into burned sites dominated by annual grasses previously devoid of sagebrush was less successful. Davies et al. (2013) incorporated a native forb—Lewis flax—into a drilled seed mix with native and non-native perennial grasses, but failed to increase forb density on burned sites dominated by medusahead.

On more mesic sites, there is stronger evidence for successful seeding outcomes with forb species. Wirth and Pyke (2003) looked at the efficacy of hand-seeding three forbs important to sage-grouse including two species of hawksbeard and woolypod milkvetch on relatively mesic burned and unburned sites in southeast Oregon. Hawksbeard seedling survival was higher in burned areas and on burned sagebrush sub-canopies, similar to results seen in DRPGs by Boyd and Davies (2010). Survival of emerged milkvetch was moderate, but overall success was constrained by very low rates of emergence (~10%) on all sites. In southwest Idaho, yarrow and flax were successfully established via broadcast seeding following prescribed fire to reduce juniper cover (Sheley and Bates 2008), but arrowleaf balsamroot was not. However, on fairly similar sites in southeast Oregon burned for juniper control, Davies et al. (2014) reported very poor establishment of Lewis Flax and a non-native forb—alfalfa—via broadcast seeding. It is not possible to discern if this discrepancy was a site difference or year effect. Although out of our study area, Nyami et al. (2011) increased native forb cover following herbicide treatment and mulching and broadcast seeding two native forbs—yarrow and Eaton penstemon—on annual grass invaded and dominated sites in mesic Palouse prairie.

Although DRPGs can compete with invasive annual grasses as mature plants, difficulties associated with establishing them from seed have led to some evaluation of alternative approaches. For example, using ruderal or early-seral species (grasses and/or forbs) may have applicability given their initially higher survival rate in the presence of invasive annual grasses (Leger et al. 2014; Uselman et al. 2014, 2015). These studies show promising results, however, viability of this type of strategy remains to be determined, particularly if it is encumbered by logistical or economic challenges to implement at large scales (i.e., seed supply volume, cost).

Conclusions and Future Research

Seeding research in our study area frequently lacks replication across important gradients of spatial and temporal heterogeneity. Relatively few studies test treatments across spatial variation, such as topography or soil textures. Even fewer evaluate the efficacy of seeding treatments among years, despite abundant empirical and anecdotal evidence that inter-annual and intra-annual variation in weather has a significant effect. Nonetheless, several patterns emerge:

- Seeding outcomes generally improve with an increasing gradient of site potential
- Reduction of competition from invasive species is usually necessary, though seeding is still frequently unsuccessful in these scenarios.
- The majority of published literature in our study area is focused on large bunchgrass seeding; relatively few studies evaluate the efficacy of other functional plant groups.
- In light of numerous challenges associated with seeding in degraded sagebrush habitat, several recommendations emerge from our review and sage advice from other authors:
- Characterization of suitable (and unsuitable) soil hydrothermal environments for seedling survival and establishment that is paired with spatial and temporal predictions will empower land managers to more strategically deploy limited resources.
- Identification of demographic barriers to seedling survival and establishment can lead to development of novel seeding approaches and technologies specifically designed to overcome these barriers.
- Identification of plant materials and traits that prioritize seedling survival and establishment, in lieu of other commonly prioritized traits such as mature plant biomass production, could better inform seed mixes that land managers choose to employ.
- Decision-support frameworks explicitly designed to inform seeding decisions that incorporate understanding of critical patterns of spatial and temporal variation are recommended.

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Appendix 4: Mechanical Treatment Literature Summary

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Introduction

Mechanical treatment of western sagebrush steppe, either as a standalone practice or integrated with other treatments, is used to address conifer encroachment in mid to high elevation habitats, reduce exotic plant populations, and thin or eliminate shrubs to increase production of herbaceous plants, improve habitat for some wildlife species, and increase fire suppression options.

Success of mechanical treatment depends on a variety of factors including timing of treatment, plant phase and height, terrain, ecological status and site potential. With that said, to be effective, treatments usually need to be carried out frequently over long periods of time (Briske 2011) and/or combined with other treatments. Therefore, mechanical treatment is frequently used in combination with other management practices such as seeding, herbicide application, and prescribed fire. As such, a large share of the available scientific literature pertaining to mechanical treatment of sagebrush rangeland reports findings of studies where mechanical treatment was integrated with other management practices.

Invasive Annual Grass Threat

Exotic Plant Management

Broad scale application of mechanical treatments, such as mowing or tilling, are generally not recommended practices in sagebrush/perennial herbaceous dominated (habitat condition A) plant communities within sage-grouse priority areas because of the negative effect on shrubs and associated wildlife habitat characteristics (Mueggler and Blaisdell 1958; Davies et al. 2009, 2011, 2012a; Derner et al. 2014; Pyke et al. 2014). However, mechanical treatment is one of several tools available for management of exotic species when present, and has potential use as a conservation measure to help shift a degraded annual grass dominated community (habitat condition D) to a perennial bunchgrass dominated community (habitat condition B). Common practices for mechanically controlling invasive weeds such as knapweed, yellow star thistle or cheatgrass include mowing, rototilling and chaining. Mechanical methods kill or reduce unwanted species through a physical disturbance that potentially increases site and resource availability for establishment of desired species.

However, of the limited studies available that consider mechanical treatment independently of other management methods, mowing or rototilling alone had a negative or no effect on desirable herbaceous species (Thomsen et al. 1996; Sheley et al. 2009; Hirsch-Schantz et al. 2014). Restoration of degraded exotic annual grass dominated rangelands (habitat condition D) requires additional, integrated management practices, such as herbicide treatments, altering water regimes or seeding. Sheley et al. (2005) found that rototilling decreased the density of knapweed and increased perennial grass densities, but only when combined with seeding desirable species at heavy rates. Sheley et al. (2009) found tillage only decreased exotics when combined with watering and seeding treatments. When integrated with seeding, mechanical disturbances can increase site availability, leading to higher densities of seeded species than in sites with aerial seeding alone (Ott et al. 2003). Only one known study examines the effects of mechanical treatment for exotic plant management at elevations less than 4,000 feet (Sheley et al. 2009). This limits our ability to draw conclusions on the effectiveness of mechanical treatments on rangelands where the threat of annual grass invasion is greatest.

Shrub Reduction

Sagebrush reduction, through a variety of methods, has been employed to achieve the objectives of increasing understory herbaceous vegetation and providing a mosaic of habitats (Beck and Mitchell 2000; Connelly et al. 2000; Chambers et al. 2014). At low elevation sites, mowing may be effective at stimulating desired perennial grass production because it reduces competition from non-sprouting shrubs and is less disruptive than burning (Pyke et al. 2014). The mechanical removal of sagebrush in plant communities where the native understory remains intact has been reported to generate increases in perennial herbaceous vegetation (Mueggler and Blaisdell 1958; Dahlgren et al. 2006; Inouye 2006; Bechtold and Inouye 2007; Davies et al. 2012a, 2012c; Derner et al. 2014). However, shifting priorities regarding management of low elevation sagebrush steppe over the past decades complicate the interpretation of mechanical treatments on plant communities. Early literature focused primarily on increasing forage production for cattle, treating sagebrush as an undesirable species, and focusing on the cheapest, most effective sagebrush eradication methods that increased forage (Pechanec 1954; Hyder and Sneva 1956; Frischknecht and Bleak 1957; Mueggler and Blaisdell 1958; Frischknecht 1963; Hedrick et al. 1966; Tausch and Tueller 1977; Blaisdell et al. 1982; Wambolt and Payne 1986).

Priorities have since shifted to include a more comprehensive set of objectives for the sagebrush steppe ecosystem, with increasing emphasis on wildlife habitat values. Earlier work prioritizing forage improvement demonstrated short term increases in forage and herbaceous production after eliminating shrubs from presumed habitat condition A plant communities (Mueggler and Blaisdell 1958; Frischknecht 1963). These studies effectively suggest that shrub removal transitioned the plant community from habitat condition A to B. However, complicating interpretation of sagebrush removal studies is the inclusion of desirable exotics such as crested wheatgrass, which may respond quite differently following disturbance associated with mechanical treatments (e.g., Frischknecht and Bleak 1957; Frischknecht 1963; Ralphs and Busby 1979; Cluff et al. 1983). Many studies indicate more nuanced and mixed results, but only report findings after two to three years. In such a short time frame, it is unclear whether the trajectory of the site was moving towards a habitat condition D dominated by annual grasses, or a habitat condition B with dominated by perennial herbs (Hyder and Sneva 1956; Hedrick et al. 1966; Prevey et al. 2010; Davies et al. 2011; Chambers et al. 2014).

One potential utility for shrub reduction is the restoration of habitat condition C communities consisting of high sagebrush cover with a depleted herbaceous understory. Due to the likelihood of habitat condition C sites converting to annual grasslands post fire, their restoration prior to burning is critical to prevent further losses of wildlife habitat. Broadcast seeding native species rarely succeeds (James et al. 2011; Hardegree et al. 2016), and dense sagebrush inhibits drill seeding native species. Thus, shrub reduction efforts may be necessary to restore habitat condition C communities. However, very little is known about the efficacy of shrub reduction for restoring degraded sagebrush habitat, as only one study examines shrub reduction and reseeding in sagebrush communities with a depleted understory. Davies and Bates (2014) tested mowing of sagebrush, and mowing followed by seeding in an attempt to recover habitat condition C rangelands, but seeding proved unsuccessful in re-establishing a perennial bunchgrass understory and the disturbance caused by mowing potentially decreased the resistance of the site to invasive exotics. Studies seeding the heartier crested wheatgrass post sagebrush removal have also failed to prevent conversion to weed dominated landscapes at some sites (Frischknecht and Bleak 1957). The few studies available for evaluating the efficacy of mechanical shrub reduction in degraded Wyoming big sagebrush habitat suggest treatments are generally ineffective when combined with seeding treatments (Hedrick et al. 1966; Davies et al. 2012b; Davies and Bates 2014).

Unfortunately, the paucity of long term mechanical shrub removal studies in the literature makes it difficult to draw sweeping conclusions. The few existing long-term studies on the matter indicate a neutral effect or reduction in perennial bunchgrasses with chaining, plowing or roto-cutting treatments (Tausch and Tueller 1977; Wambolt and Payne 1986). More recent work examining the removal of sagebrush by hand found a positive response of perennial grasses to brush removal after eight years (Inouye 2006; Bechtold and Inouye 2007). One possible explanation is that disturbance caused by mowing or tilling might increase nutrient cycling, whereas Bechtold and Inouye (2007) found that hand cutting sagebrush led to no difference in soil N content. This could explain the discrepancy between long term studies of mechanical removal of sagebrush. The perennial grasses in hand cut sites may have benefited from a reduction in competition without incurring the costs of increased disturbance caused by tilling or mowing.

While the primary threat to western sagebrush steppe at low elevations is assumed to be post-disturbance conversion to annual grasses, research examining the efficacy of mechanical treatments for addressing annual grasses below 4000 ft. is severely lacking. We found only a handful of studies that included low elevation sites. Two large-scale, multi-site studies examining mechanical shrub removal included at least one site at low elevation (Chambers et al. 2014; Pyke et al. 2014) but did not report findings on a site basis. Additionally, while conifer encroachment has generally been considered a high elevation concern, the inclusion of a low elevation site in a recent study indicates sage-steppe areas as low as 2600-2900 ft might be under threat from conifer encroachment, and could benefit from mechanical treatment. Miller et al. (2014) observed an increasing trend in both native and exotic grass cover at a basin big sagebrush site exhibiting encroachment by western juniper at an elevation range of 2600-2900 ft.

Invasive Annual Grass/ Conifer Expansion Threat

Exotic Plant Management

The few existing studies examining mechanical control of exotic plants focus on a limited number of invasive species and integrate other control methods into treatments. Current literature exists only for knapweed, yellow star thistle and cheatgrass examining the effects of mowing, rototilling and chaining. The literature is devoid of any research examining the effects of mechanical control of other noxious weeds such as *Ventenata* or medusahead. While most of the available work has been done at mid elevations, it was performed in weed dominated rangelands where the primary threat is conversion to annual grasses, and was included in the discussion above. To the best of our knowledge, no studies have examined the use of mechanical exotic plant management in sites experiencing the dual threat of annual grasses and conifer encroachment.

Shrub Reduction

Shrub removal studies undertaken at mid elevations (4000-5500 ft) rarely produced positive habitat changes (Hyder and Sneva 1956; Hedrick et al. 1966; Prevey et al. 2010; Davies et al. 2011; Chambers et al. 2014; Davies and Bates 2014). Of those that did report herbaceous increases one included exotics in the result (Hedrick et al. 1966). Thus, positive effects cannot be disentangled. Other studies were performed in productive sites with deep soils (Inouye 2006; Bechtold and Inouye 2007), and only one study indicated a short term increase in herbaceous matter (Pyke et al. 2014). Similar to the lower elevation sites, sagebrush removal or reduction in intact rangeland is not recommended. Mowing was damaging to shrubs, and twenty years were required for recovery (Davies et al. 2009; Hess and Beck 2012). There was not much success in areas more affected by invasive annual grass (habitat condition C or condition A trending C) where mechanical treatment tended to be mowing or tillage (Prevey et al. 2010; Davies et al. 2011; Davies et al. 2012b), and the integration of herbicide into chemical treatments resulted in variable levels of exotics and herbaceous biomass with responses dependent upon differing moisture regimes (Rau et al. 2014). Similar to prescribed burning, the species on the site before treatment appeared to make a difference as to what species recovered and eventually dominated. Following mowing treatments with the seeding of native species may not be enough to overcome these priority effects (Davies and Bates 2014).

Conifer Reduction

Goals for conifer removal include improving habitat for sensitive wildlife species (Connelly et al. 2000), decreasing runoff and soil erosion (Pierson et al. 2007), alleviating competitive pressures exerted on perennial species by conifers, and reducing fuel loads and increasing fuel mosaics in order to decrease fire severity (Chambers et al. 2014). At mid to high elevations, trees are often the target species since conifer encroachment leads to site abandonment by many sagebrush obligate wildlife species (Connelly et al. 2000; Knick et al. 2013, 2014). One study showed that after cutting in a presumed C-habitat condition site, abundance of birds and other wildlife that prefer sagebrush increased (Crow and Van Riper 2010). At mid-elevation sites, cutting phase II and phase III juniper in presumed habitat condition C or D sites generally resulted in short term positive results for perennial grasses and total herbaceous plants due to their existence in the interspace (Bates et al. 1998; Bates 2005; Chambers et al. 2014; Miller et al. 2014; Roundy et al. 2014), with at least one study demonstrating increases in perennial grasses 13 years post treatment (Bates et al. 2007b). However, increases in annuals were a concern at some sites, particularly under mastication (Chambers et al. 2014; Roundy et al. 2014).

Cabling, chaining, and mastication, while not as severe of a disturbance as fire, might be seen as advantageous in areas experiencing both the invasive annual grass and conifer encroachment threats, but the inability of these methods to totally control the conifer threat (Miller et al. 2005), and the risk of benefitting annual grasses from increased disturbance leads to neutral or negative results not justifiable of the cost incurred (Tausch and Tueller 1977; Cline et al. 2010). No known studies examine the effect of mechanical treatment on juniper sites with understories dominated by annual grasses (habitat condition E), but cutting without integrating seeding or weed Management would likely not yield a positive transition for the site, and annual grasses will presumably continue to dominate.

Whether or not tree debris should be left following cutting depends on the threat of increased wildfire intensity due to increased fuel loads, as well as potential subcanopy species and their seed dispersal method—some species do well with residual debris cover while others fail. Bates et al. (1998) found that juniper debris decreased species common to interspaces and benefited plants characteristic of duff zones consisting of primarily wind dispersed species. The post-cutting release of water and nutrients associated with remaining conifer debris likely affects the competitive dynamics between conifers and native perennials; mechanical removal of conifers has resulted in increased soil moisture beneath debris (Bates et al.1998), better nutrient exchange (Bates et al. 2007a), increased inorganic nitrogen (Young et al. 2013), and increased infiltration (of rainwater) (Cline et al. 2010; Pierson et al. 2014). Mastication treatments produce considerable debris, and can lead to increased soil moisture and inorganic nitrogen resulting in increases of both annual and perennial grasses (Young et al. 2013). Thus, seeding in conjunction with mastication treatments to prevent conversion to annual grass dominated habitat types (habitat condition E) is recommended (Young et al. 2013).

Mechanical conifer removal decreases canopy fuels, but increases surface fuels in the form of downed debris or through an increase in shrub biomass (Young et al. 2015). If wildfire were to burn through a mechanically treated area with high surface fuels, fire severity near the surface would likely be greater, potentially reducing shrub and herbaceous cover (Roundy et al. 2014), resulting in a possible shift to a habitat condition E. This heightened risk of increased fire severity and mortality of favorable species has resulted in managers conducting prescribed burns following mechanical treatments in order to shift the community towards a sagebrush perennial community (habitat condition A) and away from conversion to a degraded exotic annual grass dominated community (habitat condition E) (Bates and Svejcar 2009; Bates et al. 2011; O'Connor et al. 2013). However, studies integrating cutting and fire treatments either lacked a good control for cutting (Ralphs and Busby 1979; Bates et al. 2011; O'Connor et al. 2013), used cutting as the control treatment (Bates and Svejcar 2009), were used only as pretreatments (Sheley and Bates 2008), or exhibited mixed results (Bates et al. 2011). Bates et al. (2011) burned a juniper woodland after cutting 25-50% of late successional trees, resulting in perennial grass recovery by the third year. Similarly, Bates and Svejcar (2009) compared cut and winter burning treatments to cut treatments, and found that winter prescribed fire one or two years after cutting led to significant increases in perennial grass cover and density, and significantly less cheatgrass than the cut-unburned treatments. However, when cutting is coupled with fire, the trajectory of the plant community can be uncertain and often depends on the phase of the juniper stand, the season of burn, and the native vegetation present in the understory.

Conifer Expansion Threat

Conifer Encroachment

Cutting phase II or phase III juniper has generally shown positive results with an increase in herbaceous plant biomass or density when some native perennial cover remains intact, particularly in the short term (Bates et al. 1998; Bates 2005; Pierson et al. 2007; Chambers et al. 2014; Miller et al. 2014; Roundy et al. 2014). Mechanical removal of juniper in phase II woodlands (habitat condition C) has in some cases yielded increases in shrubs and herbaceous perennials, resulting in communities trending towards habitat condition A (Davies et al. 2012a; Chambers et al. 2014; Roundy et al. 2014). Similarly, the cutting of juniper in phase III woodlands with at least some presence of native perennials (habitat condition D) almost always results in some perennial recovery, especially in the short term when cutting with chainsaws (Bates et al. 1998; Bates 2005; Bates et al. 2007b; Pierson et al. 2007). Mechanical removal by cutting exhibits positive transitions from phase I or II juniper to plant communities potentially trending towards habitat condition A communities (Chambers et al. 2014; Miller et al. 2014; Roundy et al. 2014), and phase III juniper to habitat condition B communities (Bates et al. 1998, 2007b; Bates 2005), when desirable species are present at the site. However, downed debris post-cutting can make reseeding efforts difficult in sites devoid of native perennials, and can present dangerous fuel conditions for several years post treatment (Young et al. 2015).

While cutting is less destructive to native understory plants than heavy machinery treatments (such as shredding or chaining), and safer than prescribed burns (Tausch and Tueller 1977), the treatment effect can be short lived when small juniper seedlings are missed, requiring multiple treatments to prevent a transition back to woodlands (Miller et al. 2005; O'Connor et al. 2013). Mechanical treatments that do not remove all trees in an area will at best only achieve a short term positive response from native vegetation with treatment effects lasting fewer than 15 years (e.g., Tausch and Tueller 1977).

All shredding, cabling, chaining and mastication studies with known elevations were performed at high elevations greater than 5500 ft. Cabling, chaining or shredding rarely had positive effects on the subcanopy plant community, proving it ineffective at shifting the plant community from a presumed habitat condition C or D to a more favorable A or B in the few studies available (Tausch and Tueller 1977; Cline et al. 2010; Young et al. 2013). However, clear cut conclusions cannot be drawn from the limited literature examining mechanical removal of junipers with heavy machinery; studies either incorporated seeding into the treatment (Tausch and Tueller 1977; Clary 1988; Young et al. 2013), reported no vegetation metrics (Pierson et al. 2014), pooled all mechanical results together (Roundy et al. 2014), or were done jointly with prescribed burns (Ralphs and Busby 1979; Clary 1988). Studies comparing the effectiveness of the various mechanical treatments as standalone restoration techniques are needed to make inferences about their ability to act as conservation measures for improving rangelands encroached by conifers (improving habitat conditions C or D).

There are no studies at high elevations (>5500 ft) that examine the effects of cutting conifers alone. Of the literature available, either no controls were used in the study (O'Connor et al. 2013; Bates et al. 2014) or mechanical treatments were integrated with fire (Bates et al. 2014; Knick et al. 2014). Bates et al. (2014) felled about one third of trees prior to burning phase II and phase III juniper stands. Phase II woodlands responded positively with an increase in herbaceous plants, shifting from a presumed habitat condition C community to a habitat condition B, while the late successional (phase III) woodlands experienced high fire severity, shifting from a presumed habitat condition D juniper woodland towards a degraded exotic grass habitat condition (E). Thus, our ability to summarize the effectiveness of mechanical treatments in affecting positive change on high elevation conifer encroached sites is severely limited given the scarcity of relevant literature.

Conclusions and Further Research

Cutting to reduce the threat of conifer dominance in western sagebrush steppe generally elicited a positive response regardless of elevation and phase of encroachment if the plant community included adequate perennial understory vegetation prior to treatment implementation. Cutting in phase I and II woodlands and phase III woodlands generally resulted in habitat condition A and B, respectively, or a trend toward these conditions The trajectory of plant community response, however, is much less predictable when cutting was conducted in plant communities with degraded understories or when it was combined with fire treatments. Most studies have been limited to documenting only short term plant community responses, which constrains our understanding of the long term economic and ecological implications of using cutting as a standalone or integrated practice for reducing conifer dominance.

The utility of mechanical shrub reduction treatments is much more tenuous with most previous research indicating either negative or mixed plant community responses when the practice was used as a standalone or integrated restoration treatment. Early mechanical shrub reduction research was focused on increasing rangeland productivity and was generally successful in bolstering short term herbaceous plant production when conducted within intact plant communities. Effectively, these treatments transitioned plant communities from habitat condition A to B. Although limited, contemporary research is beginning to focus on the integration of mechanical shrub treatments with other practices (e.g., seeding) to recover perennial herbaceous vegetation in degraded habitat condition C sagebrush plant communities. To date, results associated with this approach have not been promising, but more research is needed to better understand its potential. Recovery of degraded habitat condition C sagebrush plant communities remains a formidable challenge for rangeland restoration specialists that likely warrants increased research attention.

Most research involving mechanical practices in western sagebrush steppe has been focused on native woody plant (i.e., shrubs and juniper) reduction, with little attention being directed at the efficacy of such treatments for management of exotic plants. The available literature suggests mechanical treatments are largely ineffective for knapweed, yellow star thistle or cheatgrass unless paired with other practices such as seeding, herbicide or watering. These three species represent a very small sampling of the exotic plants which have invaded rangelands in the Great Basin. Vast research opportunities exist for evaluating management of exotics through mechanical practices and their integration with other restoration practices such as grazing, herbicide application, fire, and seeding. Furthermore, including mechanical practices as standalone treatments will help us better understand the nuances of range management at various elevation gradients.

Our ability to draw synthesized conclusions from the wealth of rangeland scientific literature is significantly limited by the short term nature of many studies, as well as missing or lacking data regarding site characteristics, methods used, and applicable threats. Consistent publishing of specific site and treatment details will allow managers to determine clearer trends and make more detailed generalizations regarding the application of specific conservation measures to unique sites, thus improving the usability of the available scientific literature. More evidence is also needed to show in which specific cases (state prior to treatment, target and desired species, timing, etc.) mechanical treatments at low and mid elevations both stimulate desired perennial grasses and reduce undesirable species. Specifically, including more detailed vegetation data beyond presence/absence (e.g., percent cover, density) prior to treatment application would allow for a better understanding of a site's trend before and after treatment. High variability in climatic conditions year to year can confound short term studies, and while many studies report findings within a couple of years, few examine the effects of mechanical treatment longer than a decade. Finally, reporting study site elevation and threats outside of the immediate study area would also help classify studies into the appropriate model and allow for better understanding of treatments across the landscape.

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Appendix 5: Herbicide Treatment Literature Summary

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Introduction

Invasive annual grass threat to western sagebrush steppe remains one of the foremost concerns in maintaining functional ecosystem services as well as for restoring degraded systems. While there are a number of invasive weed species of concern, the literature points to invasive annual grass infestations as a chronic and widespread concern in these landscapes. When invasive annual grasses fill in open niches in the sagebrush plant community, the fine, dry content of the grasses creates continuous fuel for fire, which can consume hundreds of thousands of acres of sagebrush habitat. The resulting landscape often is reduced to a near monoculture of annual grasses and a landscape that is subject to frequent fire return intervals. This degraded landscape contributes to the decline of sagebrush-dependent wildlife species such as greater sage-grouse (Davies and Johnson 2008).

Herbicides continue to be an important management tool. However, it is clear that they must be used in an integrated plan and generally, the use of herbicides is just one part of restoring a degraded landscape. There are various options for herbicide application methods, types, and rates. Options include aerial applications that are often a necessity in rugged and expansive landscapes in the western sagebrush steppe. Ground application from a vehicle or backpack sprayer for smaller spot treatments is also extensively used. Some studies indicate specific formulations of herbicides may have greater activity depending on site conditions. Herbicide research often focuses establishing optimum rates to determine the most successful rates depending on the scale and severity of the infestation and the potential for herbicide injury of desired species. Many factors impact herbicide activity and selectivity, including soil type and weather patterns. Chemical composition of herbicides also offers alternative Management decisions with regard to application timing (e.g., before seedlings are established (pre-emergence) or after (post-emergence). The use of herbicide comes with risks such as drift that can affect desirable species outside of the plot, potential water contamination, toxicity to humans and wildlife/livestock, and if used repeatedly over time the target species can become resistant to treatment.

When used alone, herbicide is best fit for small-scale infestations, and will most likely need to be reapplied over time (Briske 2011) but ideally herbicides are best used in a systems approach program. On landscapes degraded by invasive plants, repairing ecological processes is critical to correcting the cause of the invasion rather than continuously or periodically treating the symptoms (Sheley and Krueger-Mangold 2003) as is often the case when herbicides are applied. The value of this database is that it offers managers the ability to review herbicide research conducted in the sagebrush steppe primarily in the northern Great Basin to assist in developing an integrated best management practices plan for managing or preventing annual grass infestations.

It should be cautioned that herbicide research often utilizes non-labeled or non-commercially available (at the time of the research) herbicide in experiments. Managers should be advised of this fact when reviewing the literature and be sure management plans utilize herbicides that are labeled for the specific applications.

Common Herbicides and Rates

Herbicides can be non-selective (affect all species) vs. selective (target specific species), or meant to use in pre-emergence (fall timing) or post-emergence (spring timing) plant stages. The main herbicides used in more recent studies are:

- *Imazapic* – soil active, so it continues to work as seedlings emerge; effectiveness varies in different soils; generally needs some moisture to infiltrate soil; more successful in cooler climates since warmer temperatures break it down faster; safer to use on established perennial grasses and shrubs; and appears to be a more expensive herbicide option.

- *Glyphosate* – non-selective; not soil active so anything emerging after application will not be affected; and less costly (Kyser et al. 2014).
- *2, 4-D* is also relatively common, but appears to have been used more in older studies before the turn of the century when there was somewhat of a focus on controlling sagebrush to improve forage for cattle and sheep (Blaisdell and Mueggler 1956; Mueggler and Blaisdell 1958; Whisenant 1987).

According to the literature collected, imazapic is used most frequently and tends to be the most effective in reducing invasive annual grasses with generally minimal negative impacts on established native perennial grasses. Intermediate application rates of around 70 grams (g) active ingredient (a.i.) per hectare (ha) with spot treatment were more successful at all elevations (Davies 2010; Elseroad and Rudd 2011; Davies et al. 2012). Although higher rates of up to 140 g a.i./ha or more effectively reduce invasive annuals, they also appear to have a negative impact on native perennial vegetation and shrubs (Cluff et al. 1983; Morris et al. 2009; Owen et al. 2011). Low rates were not successful in reducing target species.

Glyphosate was also more effective at intermediate rates of between 1 to 2 kg a.i./ha (Sheley et al. 2005; Kyser et al. 2012a). It is best applied in spring after seedlings have emerged (Kyser et al. 2014). Since glyphosate is non-selective and not active in the soil, application windows tend to be in the time after annual grasses have emerged but while perennial grasses remain in dormancy.

Other herbicides used that seem to be less common and/or less effective include *rimsulfuron* (pre-emergence herbicide) and *aminopyralid* (pre-emergence, selective). These were often reported better as spot treatments on invasive annual grasses. For example, aminopyralid used at a rate of 245 g/ha on medusahead was an effective control (Kyser et al. 2012b).

The majority of studies carried out herbicide treatments in the fall seasons with the next most common timing in the spring. This is most likely because many of the studies took place in cooler rangeland climates and required more selective treatment, resulting in the use of the pre-emergence herbicide imazapic. It is well known that invasive annual grasses typically germinate early with precipitation events in the fall but after native perennial grasses are dormant. The best timing partially depends on timing of the target species' reproduction or growth stages. Timing in accordance with weather variables can also affect the success of an herbicide treatment. For example, imazapic is more successful prior to light precipitation because the moisture helps move it down into the soil (Kyser et al. 2014). The literature has also found that soil type can affect the success rate of herbicides. Hirsch et al. (2012) and Morris et al. (2009) found that salt desert shrub soils can tolerate higher herbicide rates whereas sagebrush habitat soils do best with intermediate application rates. Kyser et al. (2014) indicates this is mainly applicable in the case of imazapic.

Combining Other Practices with Herbicide

Herbicide use alone tends not to be cost effective since treatments—especially on a large scale—are expensive compared to output of rangeland. Additionally, herbicide applications are not efficacious alone as annual grass infestations are often a symptom of underlying ecological problems in the plant community. Herbicide treatments are often used in combination with other practices such as prescribed burning or seeding of desired species to improve outcomes on landscape scales. The most effective combination depends on a number of variables in these heterogeneous landscapes including timing of treatment, target species, elevation, soil type, and climatic variation. Davies and Sheley (2011) found that burning prior to herbicide treatment in the spring was more successful in controlling medusahead and increasing native perennial vegetation compared with fall burning or control treatments. Grazing combined with herbicides can be effective, but the type needs to be appropriate for wildlife and livestock.

Invasive Annual Grass Threat

In the invasive annual grass threat-based model, success occurred more frequently where some desired perennial species already existed (habitat conditions B/C) with spot treatments also being effective (most likely in habitat conditions A or B or after original treatment of an entire plot) (Nyamai et al. 2011; Sheley et al. 2012). The combination of herbicide and seeding was more successful where desired species were absent (habitat conditions C/D) (Sheley et al. 2012). Combining a prescribed burn, herbicide, and seeding is effective depending on timing, but is also the least cost-effective and poses the most risk since burning can result in increased annual invasive grass cover. As with any elevation or project, land managers need to return to monitor progress and most likely spot treat over time.

Invasive Annual Grass/ Conifer Expansion Threat

Similar to the lower elevation, success with herbicide treatment alone in the invasive annual grass/ juniper expansion threat-based model resulted in areas where native perennials already existed (Pokorny et al. 2005). Herbicide in combination with seeding (Sheley et al. 2005; Sheley et al. 2007) or grazing (Sneva 1972; Evans and Young 1978; Whitson and Koch 1998) were effective at increasing perennial cover in some studies in E habitat conditions. Unlike the lower elevation where it tends to be slightly warmer and dryer, using herbicide after prescribed burning was more effective (Chambers et al. 2007; Davies 2010). This could be due to the fact that some junipers exist in this model (habitat conditions C, D, and E) and may be effectively controlled with fire.

Conifer Expansion Threat

At high elevation sites (juniper expansion threat-based model), dense sagebrush canopy cover was one issue for which herbicide was used in an attempt to thin the canopy and reduce competition to encourage perennial grass growth (Olson and Whitson 2002). It was more successful when combined with other treatments like seeding to increase grass emergence rates (Morris et al. 2009). Another study looking at whether or not controlling annual grasses with a herbicide treatment enhanced the ability of shrubs to establish found little effect of the herbicide on more mature seedlings or shrubs (Owen et al. 2011). McAdoo et al. (2013) found that transplanting shrub seedlings after applying herbicide increased success substantially (habitat condition E sites). Few studies occurred where conifer encroachment was a problem, indicating most studies were conducted within habitat conditions A or B. Since herbicide would not effectively control juniper species on its own, prescribed burning (in a C habitat condition) or cutting (in a D/E habitat condition) is recommended to control conifer encroachment.

Conclusions and Further Research

Herbicide research indicates variability among treatments when evaluating control of invasive annual grasses and herbicide injury potential to desired species, particularly for lower elevations where invasive annual grasses are the primary threat. Often they are more successful when combined with seeding after application in a C/D habitat condition in the annual invasive grass model or in a D/E habitat condition in the annual invasive grass/ juniper expansion model. Spot treatments may effectively be used on invasive annual grasses in any of the habitat conditions, but would be most successful where there is already existing native perennial vegetation.

Success appears to also depend greatly on climate and soil conditions. Generally, higher elevations are moister and some studies indicated that the soils are higher in nutrient availability (Blank et al. 2007; Chambers et al. 2007). Conservation efforts may fail where they might otherwise be successful due to drought conditions, heatwaves, etc. as they did in Mangla et al. (2011), Owen et al. (2011), and Kyser et al. (2013).

More research needs to be completed on a long-term scale to determine whether or not herbicide applications are effective over longer durations than one to three years. It is recommended to continue monitoring a site after application and use spot treatments as needed. Additionally, herbicide applications as one component of a systems approach to enhance western sagebrush steppe habitat would continue to provide valuable information to fill in knowledge gaps.

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Appendix 6: Habitat Quantification Tool Mitigation Methods

Table 14. Practices Recommended to Improve Habitat Conditions for Annual Invasive Grass Threat Model.

Practices to Change to Desirable States										
Habitat Condition	**Desired outcome**	**Practices to Implement**	**Uncertainty**	**Risk**	**Likelihood of state change**	**Time to state change**	**Avoided loss (habitat)**	**Measure of Success**	**Cost**	**Comments**
B	A	Time/ Sage-brush planting	M	Wildfire	M	Long	N/A	Increase shrub cover	$$	
C	A	Shrub reduc-tion/Control annuals/ Revegetate	H	Moving to state D	M	Moderate	H	Increase peren-nial bunchgrass density	$	High uncertainty, difficult to protect from fire
C	A	Improve grazing management of desired plants	M		M	Moderate-Long	H	Increase peren-nial bunchgrass density	$	
D	B	Control annu-als/ Revegetate	L		L	Moderate	N/A, D is non -habitat	Increase peren-nial bunchgrass density	$$$	High uncertainty, native seeding success is reliably poor, may include pre-scribed fire for site prep
D	B	Control annu-als/Revegetate using intro-duced species such as Crested Wheatgrass	L	Wildfire	M	Moderate	N/A, D is non -habitat	Increase peren-nial bunchgrass density	$$	Crested wheatgrass seed-ing success is more relia-ble, may include prescribed fire for site prep
B	A	Protect from high severity wildfire (fuel breaks)	H	Wildfire	M	Long	M	Increase shrub cover	$	High uncertainty, difficult to protect from fire

Table 15. Practices Recommended to Improve Habitat Conditions at Annual Invasive Grass / Conifer Expansion Threat Model.

Practices to Change to Desirable Habitat Conditions										
Habitat Condi-tion	**Desired Condi-tion**	**Practices to Implement**	**Uncertainty**	**Risk**	**Likelihood of state change**	**Time to state change**	**Avoided loss (habitat)**	**Measure of success**	**Cost**	**Comments**
B	A	Time, Sagebrush planting	M		L	Moderate	N/A	Increase shrub cover	$$	
B	A	Time, Protect from wildfire	L	Conver-sion to C	H	Moderate	M	Increase shrub cover	$	
C	A	Cutting/ Mechanical juniper removal	L		H	Immediate	N/A, non-habitat as C	Decrease Juni-per density/	$$	
D	B	Cutting/Mechanical juniper removal/ Revegetate under-story	M	Conver-sion to E	M	Moderate	N/A, non-habitat as D	Decrease Juni-per density/ cover	$$$	
E or D	B	Cutting/ Mechanical juniper removal/ Control annuals/ Revegetate with native per-	H		M	Moderate	N/A, non-habitat as D	Increase per-ennial buchgrass den-	$$$	
E or D	B	Cutting/Mechanical juniper removal/ Control annuals/ Revegetate with introduced perennial species such as crested wheatgrass	L		H	Moderate	N/A , non-habitat as D	Increase per-ennial buchgrass den-sity	$$	Fire risk reduc-tion strategy

Table 16. Practices Recommended to Improve Habitat Conditions at Conifer Expansion Threat Model.

Practices to Change to Desirable States										
Habitat Condi-tion	**Desired Condition**	**Practices to Implement**	**Uncertainty**	**Risk**	**Likelihood of state change**	**Time to state change**	**Avoided loss (sage-grouse habitat)**	**Measure of Success**	**Cost**	**Comments**
B	A	Sagebrush seeding	L	Moderate	M		N/A	Increase shrub cover	$$	
B	A	Time/ Potect from fire	L	Increase in Juniper cov-er	H	Moderate - long	M	Increase shrub cover	$	Success depends on seed bank and prox-imity to seed sources
C	A	Prescribed fire with mosaic effects	L	Decrease shrub cover	H	Immediate	N/A , non-habitat as C	Decreased juni-per, increase mosaic habi-tats	$$	
C	B	Prescribed fire with homoge-nous effects	L		H	Immediate	N/A, non-habitat as C	Decreased juni-per	$$	
C	A	Cutting/ Mechanical juniper removal	L		H	Immediate	N/A, non-habitat as C	Decreased juni-per	$$	
D	B	Prescribed fire	M		M	Immediate	N/A, non-habitat as D	Decreased juni-per	$$	Depends on percent juniper kill and burn
D	B	Cutting/ Mechanical juniper removal / Understory restora-tion	L		H	Immediate	N/A, non-habitat as D	Decreased juni-per	$$$	
E	B	Cutting/ Mechanical juniper removal / Understory restora-tion	M		M	Moderate - long	N/A, non-habitat as E	Decreased juni-per	$$$	depends on pretreat BG density